AN INTERDISCIPLINARY STUDY ON SCOTTISH SALTMARSH BLUE CARBON: DATA UNCERTAINTY AND VALUES IN POLICY DESIGN

Simone Riegel

A Thesis Submitted for the Degree of PhD
at the
University of St Andrews

2023

Identifiers to use to cite or link to this thesis:

DOI: https://doi.org/10.17630/sta/444
http://hdl.handle.net/10023/27576

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Simone Riegel

University of
St Andrews

This thesis is submitted in partial fulfilment for the degree of

Doctor of Philosophy (PhD)

at the University of St Andrews

September 2022

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ABSTRACT

The potential of saltmarshes to store carbon has recently been gaining increasing interest in the Scottish policy arena, particularly in the face of the recently declared climate emergency. Yet, while there are first estimates of saltmarshes' overall carbon storage capacity, there is still significant uncertainty concerning the average soil depth of Scottish saltmarshes, and thus their total organic carbon stock. Moreover, other aspects, such as the value of the carbon storage ecosystem service and how it could be incorporated into Scottish policy, are under-researched. This thesis therefore takes a holistic and interdisciplinary approach connecting natural science, economics, and social science to investigate the potential of Scottish saltmarshes for climate change mitigation. A scenario approach is used to analyse the potential organic carbon stocks according to different average saltmarsh depths to reduce the uncertainty regarding the total Scottish saltmarsh stocks. A choice experiment was then conducted to investigate the Scottish public's preferences and willingness to pay for the improvement of saltmarsh ecosystem services, particularly the carbon storage service. Furthermore, the experiment tests the significance of the influence of information provision on individuals' preferences and willingness to pay. Lastly, this thesis presents an in-depth study on blue carbon policy integration based on expert-interviews to close the link between science and policy. This work suggests that even though climate change is a pressing issue, Scottish saltmarsh climate change mitigation contributions are comparatively minor and that other saltmarsh ecosystem services must not be disregarded to facilitate a prioritisation of the carbon storage service. In terms of policy integration, this means that it may be beneficial to integrate saltmarshes and their carbon storage service into the Scottish Marine Spatial Planning framework rather than climate change mitigation policy specifically.

ACKNOWLEDGEMENTS

It's been quite a journey and I'm immensely thankful to the people who have helped and supported me along the way.

First and foremost, I would like to thank my supervisors, Dr Tim Stojanovic, Dr Laure Kuhfuss, and Prof David Paterson who believed in me and my project and were always available with advice, guidance and reassuring words. It's hard to express in words how much your support means to me. Your support, feedback, and encouragement helped and pushed me to deliver the best thesis I possibly could. Tim, thank you so much for agreeing to take over as main supervisor last year when I couldn't see another way forward. Laure, thank you very much for always being there and bearing with me while I learned as much about environmental valuation as I could during this project. Your patience was as endless as my questions, and you went above and beyond. Dave, I couldn't have finished the PhD without your help and advice and I'm incredibly thankful that you agreed to come on board. I would also like to thank Prof William Austin for introducing me to the topic of blue carbon and his help in devising the project. Bill, thank you for believing in my project when I went to you for advice about applying for a PhD and all your support in making it a reality during the first three years.

My sincere thanks go also to Dr Craig Smeaton who was always there with advice and answers to my questions about saltmarshes and Dr Will Hiles for his support in the lab; Colin Cameron who helped me with my numerous computer struggles and made sure I had access to the software I needed; Dr Antje Brown who acted as a mentor and always had an open ear; Dr Ian Lawson who replied to all of my questions about cluster analysis; and Dr Jo Hale and Prof Keith Bennett in their roles as Wellbeing Officer and Head of School for their unwavering support when I had to make one of the hardest decisions during my entire PhD journey.

Special thanks also to everyone who helped me get through the lows and celebrate the highs of the PhD. I appreciate each and every one of you and would not have made it this far without you. To my family, who were always just one call away with endless encouragement. To Seb, who was an incredible flatmate and friend and his amazing wife Lucia; the Assistant Warden life had many challenges, and I was so lucky to have a great friend to tackle them with. To Judi, Kevin, Stephanie, and Will who were the best Assistant Warden friends I could have asked for. I miss all of the fun moments we had together. To Kirsty, Corallie, Hanna, Wilmer, and Solène, my time in St Andrews wouldn't have been the same without your friendship. From the office, to fieldtrips, and to the pub, it was a great time and boost to morale. And last but definitely not least to the SERG team who adopted me for last 9 months of my PhD, the moral support and coffee breaks were much needed and appreciated.

FUNDING

This project was supported with funding for travel and research by The University of St Andrews (School of Geography and Sustainable Development) and the SBCF (Scottish Blue Carbon Forum)/Scottish Government. Further funding was provided by the University of St Andrews, SAGES (the Scottish Alliance for Geoscience, Environment, and Society), the Roberson Postgraduate scholarship, and the Thomas and Margaret Roddan Trust. The funding and support are gratefully acknowledged.

RESEARCH DATA/DIGITAL OUTPUTS ACCESS STATEMENT

Research data underpinning this thesis are subject to a permanent embargo as explained in the Declarations section.

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LIST OF ACRONYMS AND ABBREVIATIONS

AIC	Akaike Information Criterion
ASC	Alternative Specific Constant
AWE	Approximate Weight of Evidence Criterion
BaU	Business as Usual
BEIS	Department for Business, Energy & Industrial Strategy
BIC	Bayesian Information Criterion
BLRT	Bootstrapped Likelihood Ratio Test
CAIC	Consistent Akaike Information Criterion
CH	Caliński-Harabasz Index
CICES	Common International Classification of Ecosystem Service
CO_2	Carbon Dioxide
DCE	Discrete Choice Experiment
DECC	Department of Energy and Climate Change
DEFRA	Department for Environment, Food & Rural Affairs
EA	Element Analyser
EM	Environmental Management
ES	Ecosystem Services
FEGS	Final Ecosystem Goods and Services Classification System
GHG	Greenhouse Gas
GMNL	Generalised Multinomial Logit Model
H_2O	Water
HCl	Hydrochloric Acid
IIA	Independence of Irrelevant Alternatives
IID	Independently and Identically Gumbel Distributed
IPCC	Intergovernmental Panel on Climate Change
IUCN	International Union for Conservation of Nature
LCA	Latent Class Analysis
LCL	Latent Class Logit Model
MASTS	Marine Alliance for Science and Technology for Scotland
MEA	Millennium Ecosystem Assessment
MNL	Multinomial Logit Model
MPA	Marine Protected Area
MSP	Marine Spatial Planning
N_2	Nitrogen
NbS	Nature-based Solutions
NEP scale	New Environmental Paradigm
NERC C-SIDE	Natural Environment Research Council - Carbon Storage in Intertidal Environments project
NESCS	National Ecosystem Services Classification System
NMP	National Marine Plan
OC	Organic Carbon
P1	Participant 1
P2	Participant 2

P3	Participant 3
P4	Participant 4
P5	Participant 5
P6	Participant 6
P7	Participant 7
P8	Participant 8
PV	Present Value
RCP	Representative Concentration Pathways
RMP	Regional Marine Plan
RSPB	Royal Society for the Protection of Birds
SABIC	Sample-size Adjusted Bayesian Information Criterion
SBCF	Scottish Blue Carbon Forum
SCC	Social Cost of Carbon
SEFARI	Scottish Environment, Food and Agriculture Research Institutes
SNH	Scottish Natural Heritage
SNP	Scottish National Party
SO_2	Sulphur Dioxide
SPC	Shadow Price of Carbon
SSSI	Sites of Special Scientific Interest
TDC	Thermal Conductivity Detector
TEEB	Economics of Ecosystems and Biodiversity Project
UKCP	UK Climate Projections
UNFCCC	United Nations Framework Convention on Climate Change
VLMR-LRT	Vuong-Lo-Mendell-Rubin Adjusted Likelihood Ratio Test
WTP	Willingness to Pay

Units of Measurement

cm	Centimetres
ha	Hectares
kg	Kilograms
m	Metres
m^{-2}	Square Metres
m^{-3}	Cubic Metres
ppm	Parts per Million
t	Tonnes
tCO_2	Tonnes Carbon Dioxide
tCO_2e	Tonnes Carbon Dioxide Equivalent

1 INTRODUCTION AND LITERATURE REVIEW

The goal of this thesis is to connect blue carbon science with policy by analysing the carbon storage potential of Scottish saltmarshes, their economic value, and how these habitats can be integrated into Scottish policy. Reducing net carbon emissions is a highly relevant and pressing issue, which requires the consideration of a variety of solutions including preventing the release of carbon or even increasing the carbon storage of natural ecosystems. This thesis thus contributes to the development and implementation of one aspect of addressing the challenge of climate change. The Scottish context provides an excellent framework for evaluating this due to the Scottish government's ambitious emission reduction targets and significant interest in exploring blue carbon as a nature-based solution for climate change mitigation and adaptation (SBCF, 2022a; Scottish Government, 2022). As illustrated in Figure 1.1, this chapter introduces key concepts that are important for this thesis. It first defines and introduces blue carbon habitats as nature-based solutions for climate change mitigation and adaptation before going into more depth regarding saltmarshes and their carbon storage function. The concepts of natural capital and ecosystem services are subsequently introduced to establish the connection to economic valuation for market and non-market goods. The importance of valuation for decision-makers and policy development is explained, which provides the connection to the following section that introduces the Scottish policy context. The final section elaborates in more depth on the research rationale and explains the structure of the thesis.

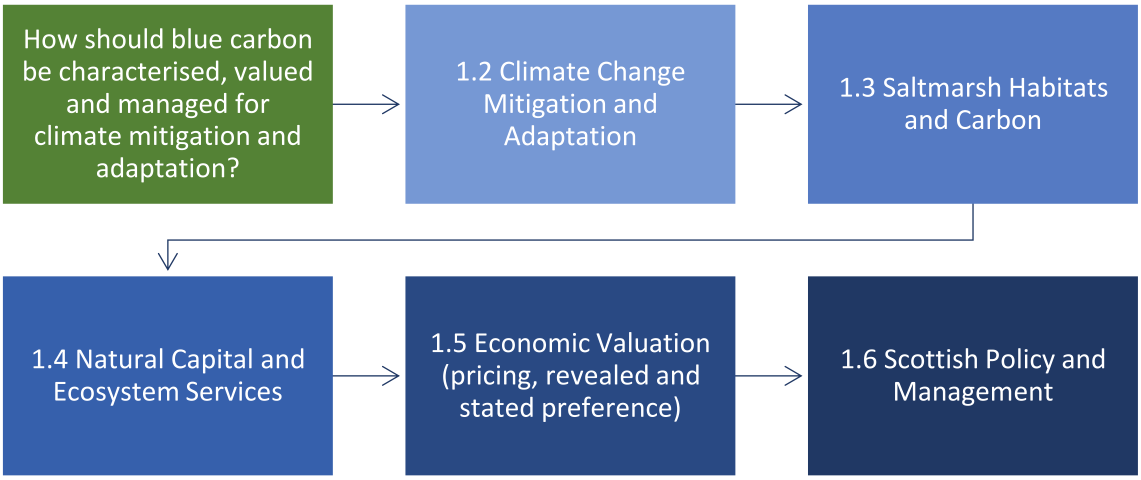


Figure 1.1: Key concepts and introduction structure.

1.1 Introduction

It is well-established that increasing atmospheric greenhouse gas (GHG) concentrations are correlated with increasing global temperature. Long-term ice core records show that global carbon dioxide (CO_2) concentrations are rising to unprecedented levels and at unprecedented speed (Gulev *et al.*, 2021; IPCC, 2018; Lüthi *et al.*, 2008). Since 2000, global CO_2 concentration has been rising by about 220 ppm (parts per million) per decade, up to 10 times faster than observed in the past 800,000 years and since 1970, the global average temperature reached the increase rate of 1.7°C per century compared to the baseline rate of 0.01°C decrease per century over the past 7,000 years. This change is largely human-driven and exceeds any observed rates of change to the Earth System driven by natural environmental forces. While there are natural factors that affect global temperature, since 2000 the level of human-induced warming has been indistinguishable from total observed warming that includes natural factors (IPCC, 2018).

Increasing mean global temperature has many adverse impacts including an increase of extreme weather events such as droughts and floods, global mean sea level rise, the magnitude of which depends on future emission pathways, and impacts on biodiversity and ecosystems such as species loss and extinction. The Intergovernmental Panel on Climate Change's (IPCC) sixth assessment report, which is the most recent, highlights that human-induced climate change already affects weather and climate extremes worldwide and assesses that evidence of observed change in extreme events and particularly their ascription to human influence has strengthened since the previous assessment report in 2014 (IPCC, 2021). According to Stern (2006), the costs that are associated with the impacts of global climate change are much higher than the costs of stabilising the climate. The Paris Agreement that was adopted at COP 21 in Paris on 12 December 2015, entered into force on 4 November 2016. Its aim is to address the challenge of climate change and limit global warming to 1.5°C preferably, but at least to well below 2°C. It is considered a landmark in combating global climate change since it is the first "binding agreement [that] brings all nations into a common cause to undertake ambitious efforts to combat climate change and adapt to its effects" (UNFCCC, 2022a). Since then, the IPCC special reports on the impacts of global warming of 1.5°C above pre-industrial levels and on the ocean and cryosphere in a changing climate have been published in addition to the preparation of the previously mentioned sixth assessment report. All of these IPCC publications reinforce the importance of limiting global climate change (IPCC, 2018; IPCC, 2019; IPCC, 2021) and irrespective of specific targets, together with the Paris Agreement they highlight the importance of countries adopting climate change mitigation and adaptation measures.

1.2 Climate Change Mitigation and Adaptation

Climate change mitigation and adaptation are well established concepts that are respectively defined by the IPCC as "an anthropogenic intervention to reduce the sources or enhance the sinks of greenhouse gases" (2001, 379) and as "adjustment in natural or human systems in response to actual or expected climatic stimuli or their effects, which moderates harm or exploits beneficial opportunities" (2001, 365). The definitions used by the Scottish government only vary slightly in the wording and not substantially in the meaning. Accordingly, climate change mitigation is defined as “reducing the amount of greenhouse gases in the atmosphere and reducing activities which emit greenhouse gases to help slow down or make less severe the impacts of future climate change” and climate change adaptation as “the adjustment in economic, social or natural systems in response to actual or expected climatic change, to limit harmful consequences and exploit beneficial opportunities” (Scottish Government, 2020a). There are a range of tools that can be employed to achieve climate change mitigation and adaptation; one of which is nature-based solutions. Blue carbon ecosystems as nature-based solutions will be explored further in the following subchapter to explain the significance of these habitats for climate change mitigation and adaptation and how they can play a role in adjusting to and limiting climate change.

1.2.1 Blue Carbon Ecosystems as Nature-based Solutions

Definitions of ‘blue carbon’ range quite widely from only including vegetated coastal ecosystems such as mangroves, saltmarshes, and seagrass meadows (Pendleton *et al.*, 2012) to including all “marine and coastal carbon fluxes or stores which can be managed to contribute to GHG [greenhouse gas] mitigation” (Kershaw *et al.*, 2022) and the Scottish Blue Carbon Forum’s (SBCF) definition that encompasses “the carbon captured by all biological metabolic process (e.g., photosynthesis, calcification) and organic material derived from other sources (e.g., terrigenous) that is subsequently deposited and stored as either organic or inorganic carbon in marine sediments” (SBCF, 2022b). If either the definition that includes only vegetated ecosystems or the definition that focuses on ecosystems that can be managed is applied, Scotland has two types of blue carbon habitats, saltmarshes and seagrass (Kershaw *et al.*, 2022). If the wider blue carbon definition that the SBCF embraced is applied, a multitude of additional blue carbon ecosystems exists in Scottish waters, including but not limited to kelp forests and mearl beds (Porter *et al.*, 2020). In the context of policy, the definition that focuses on the manageability of the ecosystems is the most relevant. However, this does not signify that other blue carbon ecosystems are not important but rather that the manageable ones are prioritised, as recommended by Howard *et al.* (2017). The definition that focuses on the manageability of the ecosystems is also the most relevant in the context of this thesis since its overarching aim is to connect science and policy.

Ecosystems, such as blue carbon habitats, that are natural carbon sinks can contribute to climate change mitigation as nature-based solutions (NbS), which are defined by the International Union for Conservation of Nature (IUCN) as “actions to protect, sustainably manage, and restore natural and modified ecosystems that address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits” (IUCN, 2022). According to Luisetti *et al.* (2015), the vegetated blue carbon ecosystems (i.e., mangroves, saltmarshes, and seagrass meadows), capture about 55% of the total carbon sequestered globally by photosynthesis.

The significance of this amount becomes apparent when the global extent of blue carbon ecosystems is put into perspective; they approximately cover about 2% of the earth surface (Wylie *et al.*, 2016) whereas terrestrial forests cover about 31% of the global land area[1] (The World Bank, 2018).[2] Vegetated blue carbon ecosystems are thus disproportionally important in sequestering CO_2. Additionally, all three of the vegetated blue carbon habitats have higher carbon burial rates than terrestrial ecosystems. These higher sequestration and storage rates compensate for the much smaller area, elevating their long-term carbon sequestration contribution to the same level as that of the terrestrial ecosystems (McLeod *et al.*, 2011). Carbon sequestration occurs through photosynthesis by the vegetation; in this process, atmospheric CO_2 is transformed into organic carbon (OC) in the form of vegetative biomass and ultimately buried in the soil (Sheehan *et al.*, 2019). The oxygen status of the soil is the decisive factor that shifts carbon sequestered by blue carbon ecosystems from the short-term carbon cycle (10-100 years) to the long-term carbon cycle (1000 years). Under anoxic soil condition, organic matter decomposition is inhibited and buried biomass decays slowly in the form of peat (van Ardenne *et al.*, 2018; Barbier *et al.*, 2011). According to Barbier *et al.* (2011) this ability to shift carbon into the long-term carbon cycle is a unique feature; usually carbon is turned over faster. The capture and storage process of blue carbon ecosystems can continue for millennia (Duarte *et al.*, 2005; Luisetti *et al.*, 2015), which means that today's blue carbon stock is likely the result of centuries or millennia of carbon sequestration and storage.

This efficiency of carbon sequestration is significant in the context of climate change since global warming is caused by the increased concentration of GHG in the atmosphere and carbon sequestration can contribute to reducing atmospheric CO_2; CO_2 is the most important GHG as it is responsible for about 74% of the amplified GHG effect (Houghton, 2015). According to Duarte *et al.* (2013), protecting and conserving natural carbon sinks is one of the easiest solutions to reduce GHG emissions by avoiding emissions caused by the loss of these sinks while also being one of the cheapest and safest solutions. Furthermore, as illustrated in Figure 1.2, vegetated blue carbon habitats are a global resource as they are distributed across all continents, except for Antarctica. Seagrass and saltmarshes can be found across almost all latitudes, while mangroves are more restricted to the tropics and subtropics with very limited extent in the temperate zone (The Blue Carbon Initiative, 2022).

[1] Number from 2015 (The World Bank, 2018).
[2] 2% of the total earth surface is ca. 10,200,000 km^2 and 31% of the global land area is ca. 46,190,000 km^2 (numbers based on Ritchie and Roser, 2019.).

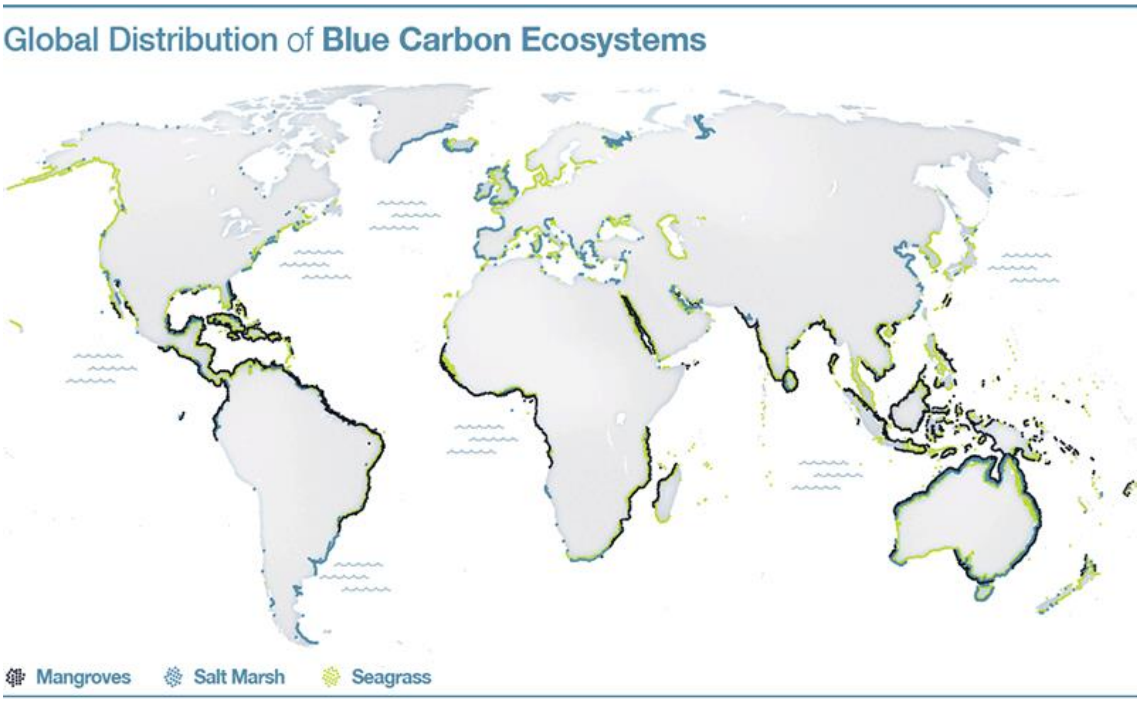


Figure 1.2: The global distribution of blue carbon ecosystems (The Blue Carbon Initiative, 2022).

However, blue carbon ecosystems can shift from being a sink to being a source of carbon emissions due to natural or anthropogenic disturbance. They should hence not be seen as an alternative to emission reduction strategies (McLeod *et al.*, 2011). Rather, they should be seen as a complementary NbS to reduce GHG in the atmosphere.

Moreover, blue carbon habitats can also be a tool for climate change adaptation since ecosystem-based coastline protection is an efficient adaptation tool. Previously built artificial hard structures may not be fit for purpose anymore due to an increase in extreme events caused by climate change in the period since their construction (Duarte *et al.*, 2013). Considerable investments would hence be needed to ensure they are future-proof. Natural ecosystems however can adapt to changes in sea level and thus maintain their protective capacity. This provides many advantages such as lower costs and the benefit of additional ecosystem services, such as nutrient cycling, biodiversity regulation, and food provision (Duarte *et al.*, 2013). Narayan *et al.* (2016) found that saltmarshes in particular can reduce wave height and have significant potential to provide protection for the shoreline. Moreover, the authors found that mangrove and saltmarsh restoration projects to improve shoreline protection can be cost-effective. In the Scottish context, there is evidence for the Solway region and for some loch-head saltmarshes on the west coast that sediment deposition is outpacing relative sea-level rise. Saltmarshes can thus function as shoreline protection and have capabilities to support climate change adaptation (Nature Scot, 2023).

Acknowledging and using NbS for adaptation can also contribute to preventing coastal blue carbon habitat loss and therefore the release of the stored carbon. Schuerch *et al.* (2018) found that large scale wetland loss can be avoided if accommodation space is created to allow for sea-level rise induced wetland migration. Acknowledging and capitalising on the protective benefits wetlands can provide may thus be an impetus for protecting existing and creating new accommodation space. However, this accommodation space competes with other land uses, which may negatively impact the ability to employ blue carbon habitats as NbS (Bradfer-Lawrence *et al.*, 2021). Furthermore, coastal vegetation as NbS for coastline protection may not consistently offer sufficient protection due to seasonal variation of vegetation development. There are a multitude of factors that impact the development of coastal vegetation such as elevation, wave exposure, and currents, and favourable conditions may not be universally present (Duarte, 2013). Narayan *et al.*'s (2016) findings that variations in wave reduction and thus cost-effectiveness depend on several factors including water depth and vegetation height, match this assessment that NbS may not always be sufficient or the best solution.

Debates about the role of blue carbon in climate change mitigation and adaptation centre on its significance, reliability, cost-effectiveness, co-benefits, and inclusion in broader frameworks. Hilmi *et al*. (2021) discuss that mangrove, seagrass, and saltmarsh restoration as an NbS mitigation solution is unlikely to produce mitigation benefits that exceed 2% of the current total CO_2 emissions (IPCC, 2019). Blue carbon NbS are thus clearly not sufficient on their own to address the challenge of climate change. Yet the authors still promote the inclusion of blue carbon habitats into climate change policy and state that the best approach to blue carbon ecosystem management is a synergy between climate change adaptation and mitigation strategies. However, they also make the observation that blue carbon ecosystems tend to be included more frequently into adaptation than mitigation policies. Wedding *et al*. (2021) raise similar points. Accordingly, blue carbon ecosystems have a high potential as NbS for climate change mitigation and adaptation and they highlight that particularly the potential for climate change mitigation has been "recognised as critical" (Wedding *et al*. 2021, 2) in the academic literature (Pendleton *et al*., 2012; Sutton-Grier *et al*., 2014). The authors further promote their inclusion in policy. Yet, Wedding *et al*. (2021) also acknowledge that recognition of NbS benefits provided by coastal ecosystems has been hindered by concerns regarding their reliability and cost-effectiveness (Seddon *et al*., 2020) and that there has been little formal consideration of blue carbon in policy. Moreover, the authors also recognise that blue carbon ecosystem restoration would have additional benefits for climate change adaptation and other ecosystem services, such as biodiversity. This recognition is a connecting factor across publications. Dencer-Brown *et al*. (2022) also highlight this, but in contrast to the concerns regarding cost-effectiveness that were acknowledged in Wedding *et al*. (2021), they stress that robust science shows that blue carbon ecosystems have "exceptional value" (Dencer-Brown *et al*. 2022, 1988). This thesis analyses the policy inclusion of saltmarshes, as blue carbon ecosystems, into Scottish policy, and contributes to this discussion regarding blue carbon ecosystems in climate change mitigation and adaptation policy.

Chapter 3 of the thesis will investigate whether there is public support for employing blue carbon habitats as NbS for climate change mitigation in Scotland. There are indications that this is already the case for coastal protection. Johnston *et al.* (2018) investigated participants' willingness to pay (WTP) for coastal flood risk reduction and found that strategies utilising hard infrastructure that is accompanied by a loss of beach and saltmarsh ecosystem services reduces social welfare even without taking the actual monetary costs for the hard infrastructure into account. The authors summarise that hard infrastructure is valued less than expected while beach and saltmarsh ecosystem services are valued more highly and that individual adaptation approaches are needed since actions that are beneficial for one community may cause negative effects in others. But these results have to be considered with caution since they are based on only one case study with communities based on the U.S. East coast (Johnston *et al*., 2018). A clear preference for NbS over built defence structures for coastline protection could be confirmed in the UK context though and includes a preference for the expansion of total saltmarsh area and for increasing the saltmarsh area with high vegetation (Rendón *et al.*, 2022). These are promising results that may indicate that there is also support for using blue carbon and specifically saltmarshes as NbS for climate change mitigation and adaptation in the Scottish context. The following subchapter provides essential context for this thesis and particularly Chapter 2 by providing a general introduction to saltmarshes, narrowing the topic down further to saltmarshes in Scotland, and explaining why they are efficient carbon sinks.

1.3 Introduction to Saltmarshes

According to Adam (1990), saltmarsh is a subset of the wider category of tidal marsh. The critical factor that distinguishes saltmarshes from other systems is the regular submersion of the marsh by the tides, which is why saltmarsh flora and fauna have both marine and terrestrial characteristics. Yet, overall, Adam (1990) argues that "saltmarsh is best regarded as a highly modified terrestrial ecosystem" since its defining organisms are of terrestrial origin. This applies to saltmarsh vegetation, which can be divided into vegetation zones according to the elevation of the marsh and the amount of submersion they experience since their occurrence is primarily defined by the species' ability to withstand submersion in seawater. The submersion is thus a stress factor (Boorman *et al.*, 2001), which would not be the case in a fully marine ecosystem. Moreover, according to Chapman (1960), even plant species that can tolerate longer submersion require the time during neap tides with reduced or no submersion to enable seedlings to establish a root system that anchors it sufficiently. Saltmarshes can have up to four zones with varying vegetation: (i) pioneer marsh, (ii) low marsh, (iii) middle marsh, and (iv) high marsh. Pioneer marsh is located at the very seaward edge of the marsh, is sparsely vegetated, and reaches out onto adjacent tidal flats. Lower saltmarsh is also located at the seaward edge of the marsh but is part of the main saltmarsh surface and more densely vegetated than the pioneer zone. The middle marsh is frequently covered in small pools and is relatively flat while the high marsh is typically located at the landward edge of the marsh and transitions into fully terrestrial ecosystems (Austin *et al.*, 2021; Haynes, 2016a).

The global extent of saltmarshes is an important factor if these habitats are to be used as NbS for climate change mitigation. Unfortunately, it is also a point of discussion with differing estimates. Greenberg *et al.* (2014, 180) state that "an estimate of 60 000 km^2 seems reasonable" but fail to provide an argument why this specific extent 'seems reasonable'. Howard *et al*., (2017) report the global extent as up to 40 million hectares, although they clarify that only 2.2 million hectares have been verified and that the 40-million-hectare estimate is based on modelling. The verified 2.2 million hectares are based on Chmura *et al.* (2003). Translated into hectares for easier comparison, the extent estimate presented by Greenberg *et al.* (2014) amounts to 6 million hectares, which presents a considerable gap to the extent presented by Howard *et al.* (2017). A further estimate presented by Mcowen *et al.* (2017) that was publicised in a similar timeframe as Howard *et al.* (2017), however, limits mapped saltmarsh extent to just under 5.5 million hectares, which is closer to the estimate presented by Greenberg *et al.* (2014). The authors do point out that there are notable saltmarsh areas that require more spatial data before they can be included in the saltmarsh mapping. Mcowen *et al.* (2017) thus raise the saltmarsh area that can be considered as verified to 5.5 million hectares.

It is widely agreed, however, that saltmarshes are under considerable pressure. Coastal ecosystems are among the most threatened worldwide; 50% of saltmarshes are already either lost or degraded (Barbier *et al.*, 2011). Adam (1990) pointed out the significant pressures on saltmarshes as early as 1990 and also suggested that measures for their protection were insufficient. He emphasised pollution and possible sea-level rise due to global warming as some of these pressures. Sea-level rise itself, however, is not necessarily an existential threat to saltmarshes since healthy saltmarshes can adapt to rising sea-levels as they accrete vertically "until they approach a surface elevation which is in equilibrium with the local tidal frame" (Adnitt *et al.* 2007, C/2). Several saltmarsh plant species are known to increase productivity with moderate increases in flooding duration, which increases organic matter production and thus soil building. There are indications that historically, vertical accretion rates exceeded sea-level rise and models suggest that this may also be the case of a wide range of future sea-level rise scenarios (Kirwan *et al.*, 2016). Valiela *et al.* (2018) agree with these assessments to an extent but maintain that the recent significant acceleration of sea-level rise is problematic. According to the authors, "recent accelerated rates of global sea level rise are unprecedentedly high, and expected to continue accelerating in the future." (Valiela *et al*. 2018, 1149). Sea-level rise, therefore, can become a threat to saltmarshes when it outpaces the vertical accretion ability. It is further exacerbated when saltmarsh migration is limited due to a steep upland topography or when structures have been built landward of the saltmarsh. Moreover, it is also important to call attention to the importance of sediment supply for saltmarsh accretion. Saltmarshes can only grow vertically if there is an influx of sediment that settles on the marsh surface. Currently, global sediment flux to the coast is in decline, which decreases saltmarshes' resilience to sea level rise (Ladd *et al.*, 2019). The combination of sea-level rise and landward migration barriers is called coastal squeeze (Valiela *et al.*, 2018) and combines threats from both the terrestrial and marine sides. Greenberg *et al.* (2014, 184) point out that "even in regions with relatively low population density, saltmarshes are disproportionally affected by development", which emphasises that such significant barriers exist and Lockwood and Maslo's (2014, 6) assessment that "human density near coastlines is increasing at a rate nearly three times faster than comparable rates of growth inland" suggests that the number of these barriers will increase further in the future. This supports the assessment that a natural shift or migration of saltmarshes inland, without active management, is unlikely in highly developed areas due to obstacles, such as infrastructure, and emphasises the requirement for a supporting policy (Doody, 2013; Greenberg *et al.*, 2014).

1.3.1 Saltmarshes in Scotland

Saltmarshes are globally spread across all regions but are most common in temperate areas, such as Scotland (Pendleton *et al.*, 2012). Scotland's Nature Agency, NatureScot[3], commissioned a saltmarsh survey to map the locations of Scottish saltmarshes (Haynes, 2016a; Haynes, 2016b). The report states that 5,840 hectares of saltmarsh were surveyed. It is important to point out that this only includes saltmarshes larger than 3 hectares or longer than 500 metres. It is likely that many saltmarshes in Scotland are smaller than these dimensions and would thus be uncounted. The map below (Figure 1.3) presents the NatureScot saltmarsh location data.

[3] At the time when the saltmarsh survey was commissioned, NatureScot was known as Scottish Natural Heritage (SNH). The agency has since conducted a rebrand.

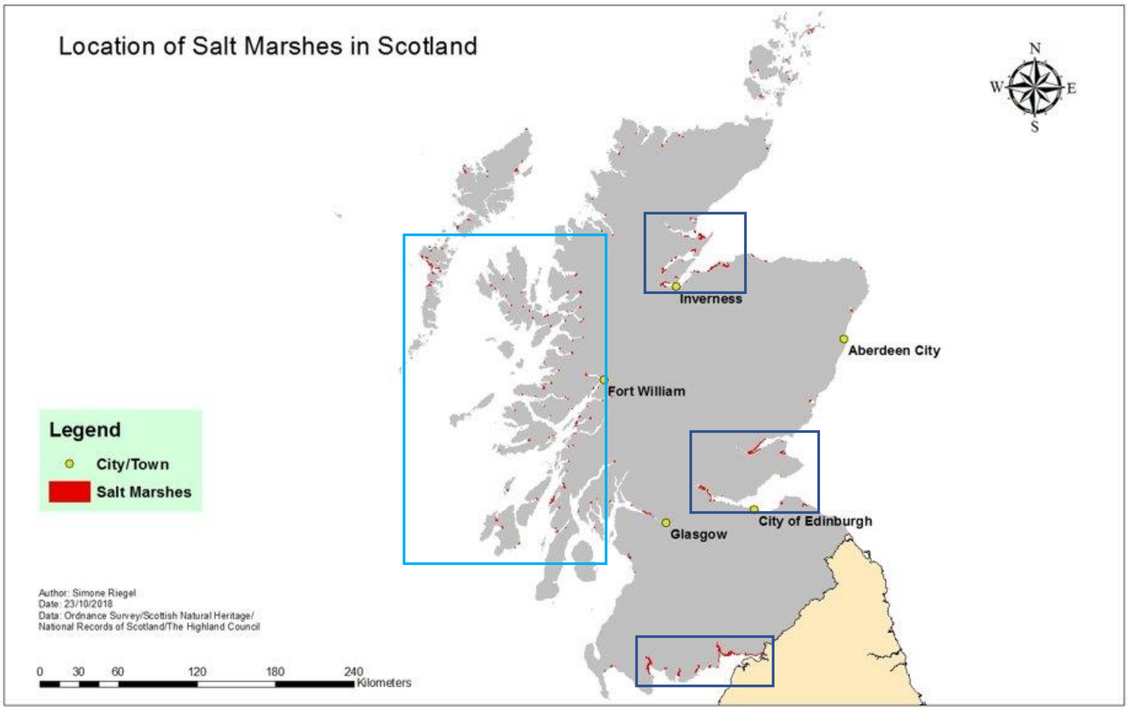


Figure 1.3: Location of saltmarshes in Scotland; based on Haynes (2016b) dataset. Contains OS data © Crown copyright and database right (2021).

It shows that saltmarsh habitats are not limited to one region but are scattered across the whole Scottish coastal area. The dark blue boxes mark concentrated saltmarsh areas primarily located on the north-eastern, eastern, and south-western coast, whereas the lighter blue box marks the area primarily on the western and north-western coast where saltmarshes are more scattered and smaller. Especially in this area, many smaller saltmarshes may not have been included in the study and data due to their size. The more concentrated saltmarsh extent is located in sheltered bay and estuarine areas, which aligns with Barbier *et al.*'s (2011, 178) observation that "extensive saltmarshes (>2 km in width) establish and grow both behind barrier-island systems and along the wave protected shorelines of bays and estuaries". The more scattered and often smaller saltmarshes are located at the most sheltered points of the sea-lochs.

In addition to the exclusion of smaller saltmarshes, there are further limitations regarding the mapping of saltmarshes. The saltmarsh survey authors argue that perched saltmarshes are often not recognised as saltmarshes. They are quite rare in Scotland and form on sea cliffs in the wave splash zone. Due to the saltwater spray, saltmarsh vegetation can grow on the cliffs on shallow sediment (Haynes, 2016a). The difficulties with classifying this environment as a saltmarsh becomes evident when analysing other literature. Adam (1990) clearly describes the same environment as the NatureScot commissioned saltmarsh survey, but he emphasises that this environment is not saltmarsh habitat. This is an important discussion since it affects the total Scottish saltmarsh extent but since this thesis is focussed on carbon storage in saltmarshes, the significance of including or excluding perched saltmarshes is discussed further in the context of saltmarsh carbon storage in the following section.

1.3.2 Saltmarshes and Carbon

The inundation of saltmarshes by the tides is critical to carbon storage as the wet and periodically submerged state of the ecosystems inhibits microbial action, which slows organic decomposition. This enables carbon accumulation and thereby the creation of carbon sinks. Additionally, saltmarshes also accumulate allochthonous carbon from adjacent ecosystems due to sediment influx (Howard *et al.*, 2017). Overall, their annual carbon burial rate ranges from 18 to 1713 g m^{-2} (McLeod *et al.*, 2011); the average carbon burial rate is 218 g m^{-2} per year and therefore significantly higher than the carbon burial rates of terrestrial forests (Figure 1.4, values from McLeod *et al.*, 2011).

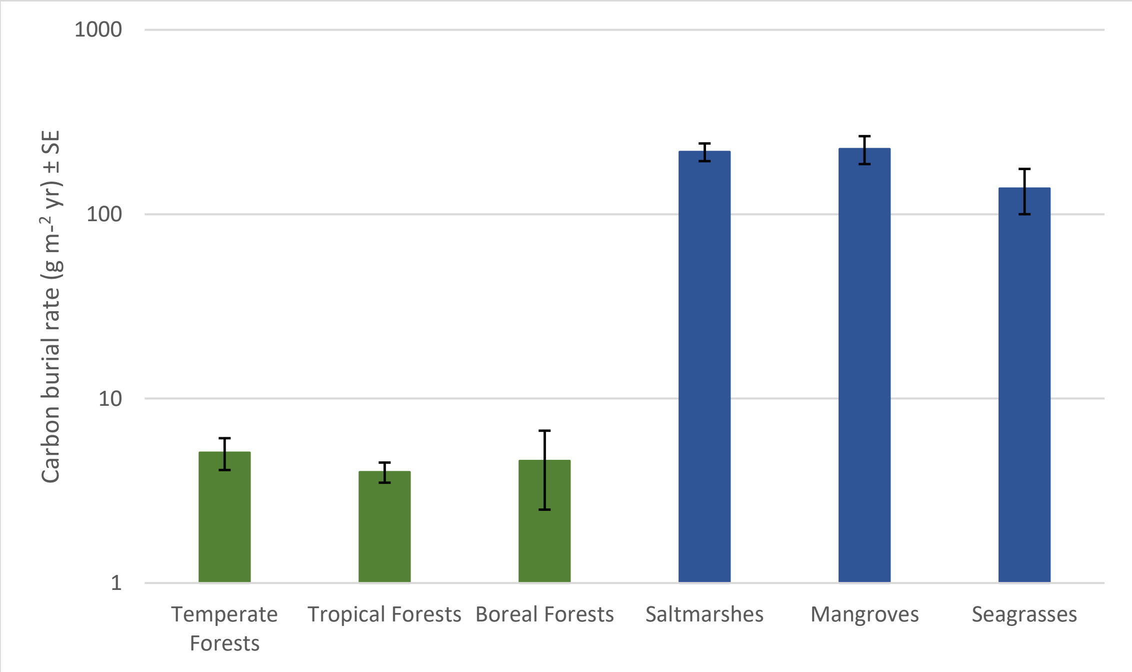


Figure 1.4: Annual mean carbon burial rates (g m^{-2}) presented on a logarithmic scale for blue carbon and terrestrial ecosystems; error bars indicate the standard error. Values from McLeod *et al.* (2011).

Studies focused on the UK have used a range of carbon burial rates that are largely similar to the annual average presented by McLeod *et al.* (2011). Accordingly, rates of 315 g C m^{-2} per year (Burden *et al.*, 2019), 64-219 g C m^{-2} per year (Beaumont *et al.*, 2014), 112.5 g C m^{-2} per year (Lockwood and Drakeford, 2021) and 210 g C m^{-2} per year (Watson *et al.*, 2020) were used. Beaumont *et al.* (2014) did point out though that typical values for UK marshes are 120-150 g C m^{-2} per year, however they proceeded with the wider range above in their study. Furthermore, the estimate used by Burden *et al.* (2019) refers to Northern European marshes and not just UK marshes specifically. Lockwood and Drakeford's (2021) estimate is based on data from the

Blackwater estuary, which was also used as one out of several data sources by Watson *et al.* (2020) and Beaumont *et al.* (2014).

A study by Rogers *et al.* (2019) suggests that sea-level rise further increases the carbon storage rate of saltmarshes. The study fails to address, however, if their observation would still be widely valid under accelerated sea-level rise conditions and possible habitat loss. Their results are also partly based on a case study where existing blue carbon habitats transformed to mangroves and seagrass beds under the influence of rapid sea-level rise (Rogers *et al.*, 2019). This is not possible in higher latitudes where mangroves do not grow. Moreover, the study does not take coastal squeeze into account, which is a major pressure on coastal habitats and prevents saltmarshes and other coastal ecosystems from migrating due to sea-level rise. However, as Chmura (2013) pointed out, to assess the impact of coastal squeeze by determining where inland migration is possible or hindered by elevation or development, LIDAR data is necessary but not available for all coastlines even in developed countries. Hence, coastal squeeze may have been omitted in Rogers *et al.*'s (2019) study due to a lack of data.

According to Howard *et al.* (2017), the mean global estimate of saltmarsh carbon stock is 570 million – 10,360 million mg C based on extent data taken from McLeod *et al.* (2011) and Duarte *et al.* (2013), and saltmarsh biomass and top 1 metre soil carbon data taken from Pendleton *et al.* (2012). There are limitations, however, to carbon stock estimates; the authors clarify that this estimate is likely to be low as most studies only estimate the carbon in the top metre of the soil even though organic-rich soil profiles are known to extend up to several metres deep. In addition to differing sampling depths, studies also use different methods. The IPCC attempted to establish international standards by recommending a three-tiered carbon stock assessment framework for carbon stock inventories and promoting the use of the Tier 3 approach, which requires highly specific carbon stock data and repeated measurements of key carbon stocks to provide estimates of change. For comparison, the Tier 1 approach relies on simplified assumptions and published IPCC default values and may thus have an error rage of ±90% for soil carbon pools. Yet, not all studies can use the Tier 3 approach due to budget restraints or lacking technical resources and capacities (Howard *et al.*, 2014; IPCC, 2014a). Upscaling results from several studies for a larger region therefore can produce significant uncertainties if different tiers were used. Additionally, Adam (1990) expressly cautions about generalising saltmarshes since, as Greenberg *et al.* (2014) state and as mentioned above, there are several factors that influence saltmarshes, but can significantly vary between localities, like the tidal regime. According to Greenberg *et al.* (2014), other factors that cause the variation in saltmarshes, are varying dominant plants and colonising fauna; the frequency of storm disturbance; and human disturbance and usage. It is very unlikely for carbon storage to be unaffected by all these influences. Lewis *et al.* (2017) present results that suggest the susceptibility of saltmarshes to these varying factors. Accordingly, the mean carbon stock differed between sampling points within the same ecosystem as well as across the study regions. A further factor causing variation in saltmarshes could be grazing. Davidson *et al.* (2017) found in their meta-analysis on the impact of grazing on saltmarsh ecosystem properties and services that above-ground plant material is reduced, which can cause diminished wave attenuation and therefore impair coastal protection. Moreover, according to Adam (1990), reduced and changed vegetation causes a reduction in sediment trapping. As mentioned above, some of the carbon stored in saltmarshes originates in other locations and is trapped by saltmarshes. Adam's (1990) observation thus suggests that less carbon may be stored due to the reduction in trapped material due to grazing. Furthermore, the reduction of biomass could also affect the sequestration rate and through this, the storage rate. However, Davidson *et al.*, (2017) only found a reduction of soil carbon from grazing in American saltmarshes and not in European marshes, which highlights geographical differences between marshes and their carbon storage abilities. Harvey *et al.* (2019) confirm these findings in the UK context where they could not detect a relationship between grazing intensity and soil OC. Rather, they found evidence that vegetation showed compensatory responses to grazing impacts such as increased root growth.

All these factors can influence the soil OC in saltmarshes and therefore make the generalisation and upscaling of saltmarsh carbon storage difficult. A further challenge for upscaling the carbon stock of Scottish saltmarshes is perched saltmarshes. Due to lacking or very shallow soil, there is most likely much less carbon stored than the average estimate that would be used for upscaling. The upscaling result would then be too high. The inclusion or exclusion of perched saltmarshes in the overall saltmarsh extent is thus highly relevant when the soil carbon stock is assessed. If saltmarsh extent is used as a variable to determine the saltmarsh soil carbon stock, perched saltmarshes should be excluded due to the absence of significant underlying soil (Austin *et al.*, 2021). This is the approach taken in Chapter 2 of this study and perched saltmarshes were thus excluded.

Reliable upscaled estimates are necessary to promote and develop policies for the conservation of these carbon sinks. Conservation is advisable since saltmarshes are under pressure. Disturbance activities like land conversion and development negatively affect the carbon stores and stored carbon is released into the atmosphere (Wylie *et al.*, 2016). These emissions add further to the already enlarged atmospheric carbon reservoir and the amplified anthropogenic greenhouse effect. Fortunately, the understanding and appreciation of saltmarshes for the benefits they provide has undergone a positive development. While Adam (1990) pointed out over 30 years ago that "for a long time the general public attitude towards wetlands has been best expressed as 'wetlands are wastelands'" and that simply arguing "that wetlands should be conserved, without being able to demonstrate the benefits from such a policy or proposing alternative ways of meeting community needs, is unlikely to succeed" (Adam 1990, 381), more than 25 years later, he stresses that the "old concept of 'wetlands as wasteland' is dead and buried" and that their provision of ecosystem services is firmly established (Adam 2016, 530). Moreover, saltmarshes have since been recognised as among the most productive ecosystems on earth (Barbier *et al.*, 2011; Silliman, 2014). Consequently, the carbon storage ecosystem service of saltmarshes is a promising angle. Global warming is a sensitive topic in the current political climate and has the potential for significant traction as an argument for conservation policy. The following section explores the concept of ecosystem services and natural capital and their significance as the connecting factor between science and policy. This section thus introduces an essential link between the following chapters.

1.4 Natural Capital and Ecosystem Services

As previously mentioned, saltmarshes provide a multitude of ecosystem services (ES). The term 'ecosystem services' first appeared in the academic literature in 1981; however, 'nature's services' were already mentioned as early as 1977 and according to Costanza *et al.* (2017, 2), it is possible to argue that the idea of nature's benefits supporting human wellbeing "is as old as humans themselves." The multidisciplinary concept of ES is an attempt to "incorporate the natural environment into the sphere of human commercial activity" (Muddiman 2019, 2), which makes it possible to link environmental degradation and loss to economics and development and draws from the current knowledges of economics and ecology. The concept of ES is closely connected to the concept of natural capital. Capital is defined as a stock that yields a flow of services over time; in the concept of natural capital, all ecosystems are part of the stock and yield a flow of ES. Ecosystems are thus directly connected to human welfare through the provision of ES. As established by Costanza *et al.* (2017), ES provided by natural capital produce human benefits in interaction with other forms of capital, such as manufactured capital, human capital, and social capital.

The focus on the benefits ES provide to society make the concept inherently anthropocentric, which is further highlighted by the ES categories the Millennium Ecosystem Assessment (MEA) defines. Three of the four categories are focused on benefits society can derive from ES (MEA 2005, 40): (i) provisioning services which refer to "products obtained from ecosystems", (ii) regulating services which are "benefits obtained from the regulation of ecosystem processes", and (iii) cultural services which are nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences". All of the ES included in these categories are based on the fourth category which contains the supporting services which "are necessary for the production of all other ecosystem services"

(MEA 2005, 40). The MEA's classification of ES is, however, not the only classification system; there are a range of different systems whose broader structure is quite similar, but which vary in their details. The MEA's classification originated from a mainly ecological project; a later classification by the Economics of Ecosystems and Biodiversity project (TEEB) took more the economic aspects of ES into account and brought ES to the attention of a broader audience as it was picked up by mass media (Costanza *et al.*, 2014). Other classifications include the Common International Classification of Ecosystem Services (CICES), the Final Ecosystem Goods and Services Classification System (FEGS), and the National Ecosystem Services Classification System (NESCS) (Costanza *et al.* 2017).

Although, as demonstrated above, the concept of ES and its classifications can easily be interpreted as anthropocentric, Costanza *et al.* (2017, 3) do not leave this claim uncontested. The authors argue that "the notion of ecosystem services implies recognition that humans depend for their wellbeing and their very survival on the rest of nature and that *Homo sapiens* is an integral part of the current biosphere." As a biological species, humans use resources of the environment just like other species to ensure their survival and to thrive. Yet, in a later section of their paper in which the authors demonstrate the interactions between different forms of capital to produce ES, they state that "storm protection by coastal wetlands requires built infrastructure, people, and communities to be protected" and that "even 'existence' and other 'non-use values' require people (human capital) and their cultures (social and built capital) to appreciate" (Costanza *et al.* 2017, 5). This human-centric assessment does not fit their earlier argument that the concept of ES is not inherently anthropocentric. Nevertheless, Costanza *et al.* (2017) highlight an important point, which is the interconnectedness of natural systems.

Overall, the ES concept aims to bridge the gap between necessary environmental protection and economic activities that affect the environment, seeking a compromise that takes our dependence on the environment into account but recognises that the use of natural resources benefits society (Muddiman, 2019). Demonstrating the value of ecosystems and their services to society can help support policy decisions that involve trade-offs between factors that affect the ability of natural capital stocks to yield ES to different extents and can thus help society make better decisions (Costanza *et al.*, 2017; Farber *et al.*, 2002). ES can thus be a tool to demonstrate and measure the benefits of saltmarsh management. Moreover, it can promote policy initiatives; many ES, such as clean air and waste treatment, are supplied as public goods without passing through the money economy and often people are not aware of their contribution to human welfare (Costanza *et al.* 1997). In the context of saltmarshes, it needs to be pointed out that, as mentioned in section 1.3.2, the awareness of their benefits has grown (Adam, 2016); yet this does not signify that society is aware of all ES saltmarshes provide, which is one of the reasons why the influence of information on the public's saltmarsh management preferences is investigated further in Chapter 3.

However, at this point it should also be mentioned that there are advocates against ES valuation; they argue that it is impossible to place value on such intangible services and that ecosystems should be protected for moral reasons and valuations are hence not needed. Costanza *et al.* (1997) report this notion and then go on to counter it with the compelling argument that there is no choice regarding the valuation of ecosystems and ES since our decisions regarding ecosystems imply a valuation, although, it may not be expressed in monetary terms; the only choice we have is whether we make valuations explicit. In the ES approach, if possible, values are measured with one standard for which monetary value is proposed as a mixture of direct and indirect market evaluation (Grunewald and Bastian, 2015); however, this does not mean that ES that cannot be monetised are disregarded (MEA, 2005). A further point of criticism is the 'commodification of nature' and that it will not be sufficient and successful in protecting nature. Rather, to promote conservation, a focus should be on ethics, aesthetics, and "instilling a love for nature in more people" (McCauley, 2006). De Groot *et al.* (2017) also warn that caution is necessary to ensure that the ES concept is not misused but emphasise that the benefit of greater awareness of nature's values outweigh the risk.

1.4.1 Saltmarsh Ecosystem Services

The previous sections on saltmarshes already referred to some of the ES saltmarshes provide such as flood protection and the carbon storage capacity. Saltmarshes have a multitude of known ES; in addition to the previously named ones, these include the provision of raw material and food; coastal protection through erosion control; water purification; biodiversity in the form of rare plants, breeding ground for birds, and the maintenance of fisheries by providing a nursery habitat; tourism and recreation through their existence and enabling activities such as wildfowling; and education and research (Barbier *et al.*, 2011; Jones *et al.*, 2011; King and Lester, 1995; Silliman, 2014). This listing of ES is by no means conclusive but demonstrates the wide variety of benefits saltmarshes provide. They may provide even more services that are still unrecognised. The carbon storage ES has only been established relatively recently and added further value to these habitats (Sheehan *et al.*, 2019).

1.5 Economic Valuation

Every economy provides a combination of market and non-market goods, with market goods being those that are traded in a marketplace involving an explicit exchange between sellers and buyers and non-market goods being those that have no or only a limited market. However, if these goods contribute to human wellbeing, they still have economic value (Bateman *et al.*, 2002).

Carbon emissions became a good that is traded in markets in 1997 with the adoption of the Kyoto protocol (UNFCCC, 2022b). Saltmarshes in their entirety, however, are a non-market good since there is no market in which they are traded but they do contribute to human wellbeing. It is thus necessary to explore both the carbon market price and non-market valuation.

Environmental valuation is an important concept that provides a variety of benefits. It is (i) important for measuring the impacts and effects on social wellbeing of a policy or project and (ii) allows for a comparison in a cost-benefit analysis since positive effects and negative effects are expressed in monetary terms (Hanley *et al*., 2013); (iii) in the context of climate change, valuation is used to estimate the impact of climate change in a common metric. Reducing emissions requires sacrifices and monetary valuation can help inform the decision which of these sacrifices are worth it and which not (Tol, 2014); (iv) it has been used in the UK to inform eco-taxes such as the landfill tax, which included determining the tax level and justifying it; and (v) it has been used to determine environmental damages after environmental catastrophes such as the Exxon Valdez oil spill; lastly, (vi) it can be used to improve the measure of welfare by enabling adjustments to national accounting figures. These figures could then also take changes in environmental quality into account such as changes in the level of pollution (Hanley *et al.*, 2013). In the next sections alternative approaches to determine the value of the carbon storage ES are discussed.

1.5.1 The Carbon Price

Up until 2009, the UK used a Shadow Price of Carbon (SPC) as a non-traded carbon price, which is based on the Social Cost of Carbon (SCC). The SCC is a global concept and measures “the full global cost of the damage [an incremental unit of carbon] imposes over the whole of its time in the atmosphere” (DEFRA 2007, 1). It is therefore the marginal damage cost of emissions and is estimated based on studies that assess the total economic damage caused by climate change as reviewed in Tol (2011). The SCC is hence also influenced by the emissions scenario chosen for the assessment of the damage caused by climate change. It is calculated in complex models, such as the DICE model used by Nordhaus (2017), that consider impacts from emissions and economic damages from climate change. The SPC is based on the SCC but can be adjusted to take other country-specific factors into account, such as political will. The Department for Environment, Food & Rural Affairs (DEFRA) based its SPC of £25/tCO_2e in 2007 on the Stern Review’s 2000 SCC of \$30/tCO_2e which is equivalent to £19/tCO_2e, and the emissions scenario that stabilises at 550ppm CO_2e. However, since the SCC rises as each emitted ton of carbon causes more damage than the previous one due to accumulation in the atmosphere, it needs regular updating. Following DEFRA’s updating convention published with the SPC of 2007, the 2020 SPC for the UK would have been £33/tCO_2e (DEFRA, 2007).

However, in 2009 a new approach to non-traded carbon valuation was set out by the Department of Energy and Climate Change (DECC) (DECC, 2009). Since 2009, non-traded carbon values have thus been based on the marginal abatement cost of meeting emission reduction targets and been subjected to regular updates. The latest review of the estimates was undertaken due to changes in international emission reduction targets (i.e., the UK signed the Paris Agreement in 2016); changes in domestic targets, such as the target to achieve net-zero GHG emission by 2050 that was adopted in 2019; and due to new understanding of technology costs and availability (BEIS, 2021a). The update presents carbon prices for every year starting in 2020 until 2050 with a 50% error range. The 2020 central carbon price according to this update was £241 per tonne of CO_2 or CO_2 equivalent (BEIS, 2021a), which represents a considerable difference to the projected 2020 carbon price based on the SCC (i.e., £33/tCO_2e).

The carbon price is important since it determines the price of carbon offset credits, which is becoming an increasingly popular tool for organisations to offset at least a part of their GHG emissions either in a voluntary or regulatory market (Herr *et al.*, 2019). If carbon is offset in voluntary markets, the money paid for the carbon credits is often invested locally into the ecosystem that helped offset it. Blue carbon offset markets are mostly linked to saltmarshes and mangrove ecosystems (Kuwae *et al.*, 2022).

Unfortunately, there is a gap between the actual carbon price and the optimal carbon price that is required to achieve successful climate change mitigation (Klenert *et al.*, 2018; The World Bank, 2021). Klenert *et al.* (2018) propose to close this gap by increasing the public acceptability of carbon pricing. Using the public's WTP as a carbon-pricing instrument can help overcome limitations associated with the carbon price; WTP provides information on public acceptability and also takes cultural and political beliefs into account (Klenert *et al.*, 2018), which is demonstrated respectively by Alberini *et al.* (2018) in their comparative study on households' WTP/avoided tCO_2 in the context of new climate change mitigation policies in Italy and the Czech Republic, and Ziegler (2017) who investigates the determinants of climate change beliefs and attitudes in the U.S., Germany, and China. The use of WTP to determine the value of the saltmarsh carbon storage ES in this thesis and its overall holistic approach consequently contribute to closing this gap between the actual carbon price and the carbon price that is required to achieve climate change mitigation.

The public's WTP can be determined through revealed and stated preference methods. Hanley and Czajkowski (2019) also highlight the usefulness of eliciting the public's WTP, particularly with stated preference methods, and point out that stated preference methods have been approved as part of cost-benefit analysis in public policies in the UK. Moreover, there is a track record of stated preference methods being used in marine policy in the UK, which encompasses saltmarsh habitats (Hanley and Czajkowski, 2019; HM Treasury, 2013). A further advantage of using WTP is that the carbon price can be considered in the context of the whole ecosystem, which as previously established is an objective of this thesis. As previously established, carbon storage is far from the only ES provided by saltmarshes. The value of the carbon storage ES can thus not be considered as being equivalent to the value of the entire habitat.

1.5.2 Valuation of Non-market Goods – Revealed and Stated Preference Methods

Carbon stored in and potentially emitted from natural ecosystems is not included in the UK ETS scheme (BEIS, 2021a) and as explained in the previous section would thus be covered by the UK's non-traded carbon price based on the marginal abatement cost. However, a different approach is needed to value other ES and as established above, using the public's WTP to determine the value of carbon stored in Scottish saltmarshes has several advantages.

WTP is based on the key economic principle that the value of a good is dependent on what a person is willing to give up for it. Economic value is defined over a positive or negative change in the ES and not the entire ES itself. In case of an increase in a good, individuals' maximum WTP to have this increase is measured while in the case of a negative change in the good the maximum WTP to prevent this decrease is measured. However, WTP does not only depend on individuals' preferences but also on their income (Hanley *et al*., 2013). The amount of money that is deducted from the income for the utility level to stay the same as it was before the environmental improvement is called the 'Compensating Surplus'. Without this compensating surplus, there would be an increase in individuals' utility due to the environmental improvement; the improvement would be 'for free'. If more money than the compensating surplus was deducted for the environmental improvement, individuals' utility would decrease. The compensating surplus hence describes the equilibrium where individual's utility stays the same and represents individuals' maximum WTP to obtain the environmental improvement (Markandya, 2005). There has been criticism that WTP is biased in favour of richer households. Yet, without being backed up with the ability to pay, WTP would not be a useful concept. This means that economic value determined through WTP is a function of the income distribution (Hanley *et al.*, 2013).

Revealed preference methods can be used to infer WTP information. These methods use information from markets associated with the good to be valued and are thus restricted to market context and rely on the assumption that decisions made in markets reliably indicate individuals' preferences. However, frequently there is no market directly associated with the environmental good to be valued. In this case, information from markets for proxy private goods needs to be used to infer a value. A private good qualifies as a proxy if it is consumed as a precondition for benefitting from the public good that is to be valued. An example of this would be measuring the travel costs individuals are willing to pay in order to visit the public good (Bateman *et al.*, 2002; Tol, 2014). However, this method cannot provide a maximum value since it can only observe how much individuals would be willing to pay at the minimum. This approach of using a proxy good also does not capture all of the benefits individuals derive from public goods causing a further undervaluation (Bateman *et al.*, 2002) and only provides an estimate of the overall value of a habitat (or the value of one ES if there is a fitting proxy good) rather than the value of the provision of several ES that are provided by a natural ecosystem.

If WTP cannot be inferred from markets, stated preference methods can be used. These methods are not restricted to a market context but rather simulate a hypothetical market for the environmental good in question. Participants respond to questions in questionnaires in a way that simulates their behaviour in the marketplace (Bateman *et al.*, 2002; Hanley *et al.*, 2007). In contrast to revealed preference methods, stated preference methods are able to capture a maximum WTP instead of just the least amount individuals are willing to pay, and they can capture non-use values which cannot be captured with revealed preference methods since non-use values are values that individuals hold for public goods without using them and thus cannot be observed. It is also possible to value a range of ES of one habitat in relation to each other. Moreover, stated preference methods can capture individuals' WTP for a change in a public good instead of just the value of the current existing public good that is inferred from the revealed preference methods, which is a distinct advantage if the purpose of the valuation is to support a management or policy decision (Hanley and Czajkowski, 2019). In the past, data from revealed preference methods was thought to be more reliable and accurate since it observes individuals' actions while stated preference data is based on what individuals say they will do (Willis, 2014). This is called hypothetical bias and measures have been developed to address this issue. Hanley and Czajkowski (2019) maintain that sufficient information is available on 'best practice' to ensure that estimates derived from stated preference methods are valuable to policy. Since the aim of this thesis is to connect science, economic valuation, and policy in the context of saltmarsh management and since there are sufficient measures to manage hypothetical bias, a stated preference method was used in this thesis.

1.5.2.1 Stated Preference Valuation Methods

Stated preference methods in environmental economics focus on the direct impact the environment has on utility. The two main methods are contingent valuation and choice experiments. Both methods use the survey instrument to ask a subset of the public (usually a subset that is affected by the environmental change in question) directly about its WTP for a hypothetical change in environmental quality. (Bennett and Blamey, 2001; Hanley *et al.*, 2013).

The contingent valuation method was developed before the choice experiment method; contingent valuation was first used to a significant extent by environmental economics in the 1970s, while choice experiments were only first applied in the 1990s (Hanley and Czajkowski, 2019). The contingent valuation method values an overall change in a good or ecosystem; respondents are asked whether they are willing to pay a specific amount to achieve a hypothetical environmental goal. They have thus the choice between the status quo and a proposed situation that represents an improvement for which they would have to pay (Bennett and Adamowicz, 2001).

As previously mentioned in section 1.5.2, the validity of stated preference methods, particularly contingent valuation before the emergence of the choice experiment method, has been debated in the past (Hanley and Czajkowski, 2019). Due to the critique towards the contingent valuation method, economists explored and developed other stated preference methods. Two variants of conjoint analysis emerged, the contingent ranking or rating method and the choice experiment method. Conjoint analysis requires respondents to consider alternatives that are described in terms of their attributes. Lancaster's (1966) characteristics of goods approach is the conceptual foundation of this technique. Contingent ranking or rating that requires respondents to rank alternatives was the first to emerge, however, due to theoretical and practical obstacles, including the difficulty of interpersonal comparisons with ranked data and the difficulty of the task itself for respondents if a large number of alternatives needed to be ranked, the choice experiment method that requires respondents to choose between alternatives evolved (Bennett and Blamey, 2001).

Choice experiments have two main advantages over the contingent valuation method that make it the more suitable approach for the economic valuation included in this thesis. Firstly, choice experiments are more efficient since they include more than one choice per respondent and thus more observations can be made with the same sample size. And secondly, unlike contingent valuation studies, choice experiments can value the change in attributes[4] of the policy or project and gather information on trade-offs between all of these attributes. To value the different attributes with contingent valuation, a series of contingent valuation studies would have to be designed for each attribute separately, which is very resource intensive (Bateman *et al.*, 2002; Bennett and Adamowicz, 2001). In addition to these advantages, there are precedents in which choice experiments have been used to value the ES provided by saltmarshes (Birol and Cox, 2007; Interis and Petrolia, 2016; Petrolia *et al.*, 2014). The choice experiment method is described in more detail in Chapter 3.

[4] The different elements or building blocks of a policy or project are called attributes in the context of a choice experiment. In the case of a valuation of a specific ecosystem, the attributes in the choice experiment represent the ecosystem services that are provided by the ecosystem.

1.5.2.2 Discrete Choice Experiments' Theoretical Framework

The analysis of data collected with a choice experiment is based on random utility theory (McFadden, 1973). Utility is synonymous with wellbeing (Bateman *et al.*, 2002) and, under Lancaster's model that underpins choice modelling, is derived from the characteristics a good possesses (Lancaster 1966; Louviere *et al.*, 2000; Willis, 2014). Random utility theory is based on the hypothesis that individuals make choices based on the good's characteristics but that a certain degree of randomness also exists either due to randomness in the participant's preference or incomplete information (Willis, 2014). Utility cannot be observed directly, but if a valid preference elicitation procedure is used, a proportion of this utility can be understood and explained. However, a part of the utility will always remain unexplained; this proportion represents the random component (Louviere, 2001). Based on this fundamental assumption, the marginal utility of a change in a characteristic of the good can be estimated. If a payment attribute is included, it is possible to infer WTP estimates from the observed trade-off between the marginal utility associated with one of the good's characteristics and the marginal utility of income (Atkinson *et al*., 2018; Bateman *et al*., 2002; Louviere *et al*., 2000). The different assumptions that can be made about the distribution of the random component give rise to the different models of choice (Louviere, 2001).

1.5.3 The Effect of Information

The underlying assumption of stated preference methods is that respondents make informed choices (Tienhaara *et al.*, 2021). To ensure that this assumption holds, information on the good that is to be valued with the choice experiment is provided in the survey instrument. It is well established in the academic literature that the kind of information and the amount that is provided can influence respondents' WTP, although different effects have been observed. Sandorf *et al.* (2016) for example observed that explanations about the good that are aided by video material increased WTP, while Imamura *et al.*, (2020) found that solely relying on video information decreased WTP. Munro and Hanley (2001) compared the results of eight studies and found clear variations. Some studies showed a significant difference between groups that were provided with additional information (Bergstrom and Dillman, 1985; Samples *et al.*, 1986), while others showed a difference, but it was not statistically significant or the significance depended on the payment vehicle[5] that was used (Hanley and Munro, 1992; Samples *et al.*, 1986). Munro and Hanley (2001) furthermore present an interesting observation from their comparison. Accordingly, values are more sensitive to new information if the good that is valued does not have a strong use value. There was no significant impact by information in two out of three studies that valued goods with use-values (Bergstrom *et al.*, 1989; Boyle, 1989; Boyle *et al.*, 1990). The other five studies either

[5] The payment vehicle provides the context of the payment attribute; it shows the participants how the payment would be made (e.g., through an increase in a tax or through a donation).

valued only existence values or a combination of existence values and use values. In four out of the five, information had a positive effect on mean WTP (Bergstrom and Dillman, 1985; Samples *et al.*, 1986; Whitehead and Blomquist, 1991). The only exception was the study by Hanley and Munro (1992).

Furthermore, better information can also positively influence WTP (LaRiviere *et al.*, 2014). In their study, Jessoe and Rapson (2014) highlight the importance of clear information on price elasticity in the context of domestic energy usage. They found that clear real-time information on the quantity of electricity being used led to a significantly higher reduction in energy usage during announced high pricing events compared to the second treatment group that was exposed to the same announced high pricing events but without real-time energy use information; this second group still reduced their energy usage but to a significantly lower extent. Other studies, however, found no significant effect of information provision on respondents' behaviour (Boyle *et al.*, 1990). A third possibility is that studies did not observe a significant impact of information on mean WTP but on the variance. Boyle (1989) observed such an effect in his study on trout fishing in Southern Wisconsin. Yet, Shapansky *et al.* (2008) also tested for this effect in their study on the effect of information and respondent involvement on preferences for passive-use values and found no significant difference in the variances (or in the mean WTP). This wide variation in findings highlights the importance of being cautious about generalising the observations of one study.

Czajkowski *et al.* (2016a), Czajkowski *et al.* (2016b), and Needham *et al.* (2018) investigated the effect of information on mean WTP for a good in combination with their pre-existing knowledge of the good. Czajkowski *et al.* (2016a) focus on the scale parameter that indicates the variance of WTP and found that mean and variance of individual-specific scale parameters are sensitive to the information that is provided to respondents. Needham *et al.* (2018) and Czajkowski *et al.* (2016b), however, found that while pre-existing knowledge had a significant impact on respondents' WTP, the additionally provided information and increase in knowledge did not alter the mean WTP or its variance. Needham *et al.*'s (2018) results did indicate though that pre-existing information impacts the mean WTP, which is consistent with previous literature that found that pre-existing knowledge and experience with the good correlates with WTP (Cameron and Englin, 1997; Ekstrand and Loomis, 1998; Tkac, 1998). Generally, respondents know more about goods they have experience with or care about (Czajkowski *et al.*, 2015). Knowledge on the effect of information on the public's WTP adds a further level of support for decision-makers on top of the support the economic valuation itself already provides for environmental management since it provides information on potential communication strategies or measures to increase the acceptability of management strategies. Investigating the effect of information on the public's preferences and WTP for managing saltmarsh carbon storage is thus included in this study.

1.6 The Scottish Context

Since this thesis aims to analyse the best way of integrating saltmarsh blue carbon into Scottish policy and since saltmarsh management falls within the concept of Environmental Management (EM) and more particularly Marine Spatial Planning (MSP) in Scotland, these two concepts will be reviewed in more depth in subchapter 1.6.1 before they are discussed specifically within the Scottish context in section 1.6.2.

1.6.1 Environmental Management

Although EM and MSP are two different concepts and MSP sits within the wider context of EM, there is significant overlap. Planning in general, which includes MSP, contributes to the delivery of effective EM. Moreover, due to the difficulty of limited knowledge of marine ecosystems, which is essential for MSP, the parallel development of EM tools is important to achieve effective management (Smith *et al.*, 2011). Both concepts are thus significant for saltmarsh management in Scotland.

1.6.1.1 What is Environmental Management?

EM is an evolving, broad concept that can be broken down into three broad steps: (i) identifying goals, (ii) establishing if these can be met, and (iii) developing and implementing the means to achieve the goals (Barrow, 1999), but does not have a set definition. Barrow (2005, 15) describes its aim as "meeting and improving provision for human needs and demands on a sustainable basis with minimal damage to Nature" with sustainable development simultaneously as a core concept and the goal of EM. Yet, in the late 1990s, Bryant and Wilson (1998) criticised EM as a technocentric problem-solving concept aiming to provide practical assistance to state officials that did not focus on taking cultural, economic, and political aspects into account and was ignoring potentially valuable contribution by non-state actors such as grass-roots stakeholders, businesses, and NGOs. This approach has been coined 'environmental managerialism' and criticised by Bryant and Wilson (1998) as disconnected from key issues in human-environment interactions. Barrow (2005) argues that a shift from previous top-down approaches to the encouragement of bottom-up approaches promoting consultation and participation can help those challenges. Accordingly, EM emphasises stewardship over exploitation and has the key role of watching out for and warning about critical thresholds. It thus aims for management that ensures a long-term and sustainable use of ecosystems with a participatory and precautionary style based on a holistic approach. However, there remains a need to be more politically informed and overcome institutional fragmentation where actors' knowledge and actions are artificially constrained by narrow directives (Brunyeel, 2009; Reed, 2009; Wilson, 2009). Bennett *et al.* (2018) argue that there is an increasing emphasis on local communities and resource users in EM policies and practice. This approach profits from the fact that local people can play a central role in caring for the environment and have a strong connection to it.

Nevertheless, while EM is undergoing this development, its challenges are growing as well. Bryant and Wilson (1998) already argued in the late 1990s that state and non-state actors were facing increasing problems due to the acceleration and globalisation of environmental challenges. These challenges have become even more pronounced since global pressures have further accelerated and there is a growing recognition that human factors have a large influence on rapid climate change with debates about improving governance or more radical transformative approaches (Barrow, 1999; Barrow, 2005).

1.6.1.2 Key Concepts

Barrow (2005) notes that it is difficult to pin down key issues of EM since it is a concept that is undergoing rapid development, but nevertheless introduces three important concepts of EM: (i) the polluter pays principle, (ii) the precautionary principle, and (iii) sustainable development. The polluter pays principle denotes a shift in attitude; instead of pushing aside the issue of pollution and thinking about it in terms of something to be dealt with at a later time, the focus moves to avoiding pollution in the first place by creating a deterrent to polluting in the form of holding the polluter accountable and making them pay for the pollution they are causing. It is regarded as an effective tool in the context of climate change (Pill, 2022). In addition to fines, the polluter pays principle can also be enforced through licensing in which case entities need to convince the authorities that no difficulties will arise from a development. Risk and impact assessments are useful tools to support this process. This application of the polluter pays principle highlights its close relationship to the precautionary principle.

The goal of the precautionary principle is to prevent environmental harm rather than to react to it after it has been caused and can thus be described as 'institutionalised caution'. It is well-established in international environmental law (Applegate, 2000). Potential harmful impacts of developments should be anticipated by decision-makers who then have the responsibility to make decisions in order to avoid them. The precautionary principle should be used if "(1) the range of possible impacts from one or more uses cannot be predicted, (2) one or more of the outputs or outcomes could have extremely undesirable impacts for future people, and (3) substitutes are not available for the resource to be used." (Mitchell, 2002, 34). Like the licensed approach to the polluter pays principle, a critical component of the precautionary principle is the shift of the burden of proof to the proponents of a proposed development. This acts jointly with the other three core components which are the avoidance of impacts through the exploration of alternatives; the increase of public participation in decision-making; and preventative action under uncertainty. The last core component acknowledges that uncertainty is a reality and signifies the implementation of regulations before scientific certainty is available, which risks costs that may in hindsight not be justified (Barrow, 2005). The careful use of the precautionary principle is hence required to avoid wasting funds. This dilemma of needing to act before knowledge and proof are fully available can be avoided by finding win-win paths. The key to these win-win paths are additional benefits that may still offset costs even if the regulation turns out to be ineffective regarding its actual target (Barrow, 2005; Mitchell, 2002). This is in line with Bodansky (1991) cautioning that applying the precautionary principle does not provide a guarantee that no harmful impacts take place since it is impossible to anticipate all environmental problems that may be caused by a development.

Sustainable development is a versatile concept that can be seen as goal, paradigm shift, or guide for development. As mentioned in section 1.6.1.1, in EM it is not merely a core concept but can also be considered as its objective (Barrow, 2005). There are strong parallels between the purpose of EM and the commonly used definition of sustainable development from the Brundtland Report (1987): "Sustainable development is development that meets the needs of the present without compromising the ability of future generations to meet their own needs." (Brundtland, 1987). Conservation and development are not considered opposing interests since a healthy environment is paramount for healthy communities. Correspondingly, economic viability,[6] environmental protection, and the health and happiness of people are the three goals of sustainable development. EM can support sustainable development by identifying key issues and opportunities but also threats and limits; establishing feasible strategies and boundaries; overseeing stakeholders; as well as coordinating diverse factors such as physical and socio-economic issues (Barrow, 2005).

1.6.1.3 Environmental Management Tools

There are a wide variety of tools for EM that are important for decision- and policymakers since they provide roadmaps, help with anticipating or scoping impacts, provide monitoring capacity, and/or enable the inclusion of environmental impacts into an economic context. Examples include Hazard and Risk Assessment. Increased use of risk assessments in EM is partly linked to the use

[6] Barrow (2005) uses the term 'economic growth' rather than 'economic viability'. However, as summarised by Purvis *et al.* (2019), there is not one clear definition of the economic aspect of sustainable development but rather different interpretations. According to Purvis *et al*. (2019), the UN, for example, pushes a growth-oriented understanding of the economic aspect, while other understandings reject the growth narrative (e.g., Brown *et al*., 1987). 'Economic viability' is thus a better term than 'economic growth' in the context of this thesis as using the term 'economic growth' would entail a clear positioning within this debate. However, it is beyond the scope of this thesis to unravel this debate and a more neutral term to explain this aspect of sustainable development is thus more suitable.

of the precautionary principle (Barrow, 2005). Further tools that are important in the context of this thesis are pilot studies and the scenario approach. Large projects may take the scoping approach further and run a pilot study to reveal opportunities or potential detrimental impacts. However, this requires time and cost, and must be built into the project timetable. Additionally, small scale results may also not scale up well to predict the impact of the full project accurately (Barrow, 2005). But other theorists argue that piloting can be an important component of adaptive management, allowing for experimentation, learning, and testing (Armitage *et al*., 2007; Greenhill *et al*., 2020). Overall, EM needs to be proactive, and it is essential to assess future scenarios. They are "hypothetical sequences of events, constructed for the purpose of focusing attention on causal processes, crucial developments and for providing insight into ongoing situations" (Barrow 2005, 182). They are often determined with the assistance of modelling or brainstorming by a group of experts and are not accurate forecasts but rather an exploration of possible developments and responses. Participatory scenario planning can go beyond the inclusion of only a group of experts and can include communities in the planning process and natural resource management. It can bring together a wide variety of stakeholders and their perceptions on an issue. This constitutes a great advantage but can also introduce participant bias (Miller and Morisette, 2014). A limitation of a purely quantitative scenario approach, however, is thus the lack of such different perspectives. Yet, a focus on the analysis of quantitative data also has the advantage of eliminating human bias and presenting transparent assumptions (European Environment Agency, 2023). A combination of these two approaches is thus considered advantageous (Döll, 2004, Peterson *et al.*, 2003). According to Miller and Morisette (2014), developing scenarios based on a participatory approach usually consists of series of workshops that brings the different stakeholders together. It is thus a resource-intensive method and beyond the scope of this thesis, which instead incorporates a quantitative scenario approach based on available quantitative data in the following chapter.

1.6.1.4 Marine Spatial Planning

Marine Spatial Planning (MSP) sits within the wider context of EM. Its purpose is to manage the spatial and temporal distribution of human uses in the ocean. It is a holistic approach that goes beyond singular sectors with the aim to foster compatibilities, reduce conflicts, and balance conservation and development (Ehler and Douvere, 2009; Frazão Santos *et al.*, 2019). MSP should be conducted in a continuous cyclical fashion that allows for adaptation to changed circumstances and involve relevant actors and stakeholders from government and society (Olsen *et al.*, 2014; Frazão Santos *et al.*, 2019). MSP is based on the premise that it is best to begin planning before issues arise (Ehler, 2012) and is thus a proactive tool with which decision-makers plan management actions that are anticipated to deliver the results required to achieve the desired future state of the marine environment. It is significant in the context of this thesis since it provides a place for saltmarsh management integration into marine policy in addition to climate change mitigation policy.

Although MSP dates as far back as the 1970s, MSP information and expertise was relatively sparse for over two decades until it expanded significantly in the early 2000s (Grip and Blomqvist, 2021; Frazão Santos *et al.*, 2019). It has its roots in conservation, however, according to Frazão Santos *et al.* (2019) the focus has shifted to managing conflicting existing and future maritime uses. Moreover, the authors criticise that contemporary MSP processes focus more on blue economy growth than balancing conservation and development, making conservation less of a priority. This could be problematic for saltmarsh management, but identifying, assessing, and valuing ES can challenge this shift and are thus key for informing environmentally sustainable MSP (Frazão Santos *et al.*, 2019). Furthermore, the impacts of climate change could increase use and environmental conflicts in the marine domain; this presents an evolving challenge which makes keeping track of potential changes in the distribution of marine resources and ocean uses an important step in MSP (Ehler and Douvere, 2009; Frazão Santos *et al.*, 2019). In contrast to marine planning, terrestrial planning has been in place for several decades; these two planning systems now meet in the coastal zone leading to challenges regarding the spatial boundaries of the two planning systems and the management of coastal ecosystems where the terrestrial environment transitions into the marine environment. The Ecosystem Approach could be a useful tool to bridge this border between management systems. It is "a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way" (Convention on Biological Biodiversity, 2004) and also underpins MSP. It is a holistic approach that seeks to balance sustainable use, conservation, and an equitable distribution of the benefits generated by ecosystems and requires adaptive management to account for the incomplete knowledge and dynamic nature of ecosystems, and the uncertainty that is involved in managing these systems (Convention on Biological Biodiversity, 2004; Martino and Kenter, 2021). Limited information on marine ecosystems further inhibits the introduction of MSP and highlights the importance of making use of EM tools in MSP (Smith *et al.*, 2011). This is also the case regarding saltmarshes, which will be demonstrated in the second chapter of this thesis, which uses a scenario approach to address the issue of uncertainty with respect to saltmarsh depth. The boundaries of the MSP system in Scotland are covered in the following section.

1.6.2 Environmental Management and Marine Spatial Planning in Scotland

The Scotland Act of 1998 established a Scottish government and parliament with devolved powers from Westminster over all matters that are not specified as reserved to the UK parliament, which includes environmental policy and thus land-use planning and management. This is important in the context of this thesis since it affects blue carbon policy, providing opportunities to include it at the Scottish policy level. The Scottish government has a track record of adopting high environmental standards (Scotland Act (1998); Warren, 2009), which it confirmed recently by reducing the timescale for reaching the aim of net-zero emissions. In 2019, the Climate Change (Emissions Reduction Targets) (Scotland) Act was passed which establishes the Scottish Government's commitment to reach net-zero emissions by 2045. This Scottish ambition is an important factor in the exploration of blue carbon as an NbS and into inclusion into policy (SBCF, 2022a). In his book on EM in Scotland, Warren (2009) further explores land-use management in Scotland, however, intertidal wetlands are covered under the Marine (Scotland) Act (2010) that introduced a Marine Planning system, which will hence be focused on in this section.[7] The boundary between marine and terrestrial planning in Scotland is the mean high water spring tide line. Any area that is submerged at mean high water spring tide is covered by marine spatial planning (MSP); in the case of estuaries, rivers, or channels, the waters are covered under MSP as far as the tide flows at mean high water spring tide (Marine (Scotland) Act 2010).

Both sustainable development and climate change mitigation and adaptation are included as general duties in the Marine (Scotland) Act (2010). It stipulates that Ministers and public authorities "must act in the way best calculated to further the achievement of sustainable development, including the protection and, where appropriate, enhancement of the health of that area, so far as is consistent with the proper exercise of that function" (Part 2, Section 3) and "must act in the way best calculated to mitigate, and adapt to, climate change so far as is consistent with the purpose of the function concerned" (Part 2, Section 4) regarding sustainable development and mitigation and adaptation respectively. In Part 3 it further sets out the adoption of a National Marine Plan (NMP) and the possibility for adopting Regional Marine Plans (RMP). Preparing an NMP requires Ministers to set economic, social, and marine ecosystem objectives as well as objectives relating to climate change mitigation and adaptation; assess the condition of the area in question at the time of the plan's preparation; and summarise the pressures and impacts of human activities on the region in question. In the case that RMPs are drafted or amended, they have to be in conformity with the NMP in effect and compatible with the RMP of the adjoining region or regions. The Marine (Scotland) Act (2010) also calls for regular reviews of NMPs and RMPs.

The NMP for Scotland was adopted in 2015 and the Marine (Scotland) Act (2010) requires a 5-year timescale for review. However, the NMP covers Scottish inshore waters governed under the Marine (Scotland) Act (2010) and Scottish offshore waters governed under the Marine and

[7] In addition to the Marine (Scotland) Act (2010), marine spatial planning in Scottish waters is also governed by the Marine and Coastal Access Act 2009, which is a UK parliament Act (Marine Scotland, 2021). However, in the Scottish context, the Act applies only to offshore waters, which is not relevant in the context of this thesis (Scottish Government, 2020b).

Coastal Access Act (2009), which have conflicting timescales for review. Since Scottish Government's intention is to keep the NMP as an integrated plan and update it as one document, a shorter 3-year timescale for review was agreed to satisfy the requirements of both underlying Acts (NMP, 2015). Thus, the NMP was first reviewed in 2018. However, in the 2018 review it was decided to forego drafting a new plan or making amendments to the existing one due to the uncertainties around the UK leaving the EU (Scottish Government, 2018a). The second review took place in 2021 and found that the plan remains effective but that work to replace it should commence to ensure that it meets the arising challenges that were identified impacting on the plan. Amongst others, these identified challenges included the changed legislative context of the plan due to the exit from the EU; the effect the climate emergency has on the seas and sectors that rely on it; new emerging industries and technologies; and increasing competition for space and marine resources. However, this recommendation does not necessarily mean that changes will be made to the plan since this is a decision that needs to be made by Ministers, but it will form the foundation for advice to Ministers (Marine Scotland, 2021).

Up to this point (April 2022), the NMP of 2015 remains valid and mentions saltmarshes and the ES they provide in two of the 'General Policies' sections. General Policies apply across all developments, existing and future. General Policy 5 requires developers to act in the way that is best to mitigate and adapt to climate change. Regarding saltmarshes, it calls for reducing pressure and safeguarding ES; if significant harm to the ecosystem cannot be avoided, compensatory habitat creation or enhancement should be considered. Moreover, opportunities for enhancing natural carbon sinks and allowing natural coastal change should also be taken into consideration. The second mention of saltmarshes is in the context of General Policy 8 on coastal processes and flooding, which stipulates that natural processes and features should be utilised for flood risk management and coastal protection; in this context, it encourages managed realignment, which refers to breaching existing sea defences so the land behind them can flood and natural saltmarsh can be restored or created (NMP, 2015; Luisetti *et al.*, 2011).

Slater and Claydon (2020) and Sangiuliano (2019) examined the NMP in detail but took different approaches. Slater and Claydon (2020) examined its effectiveness while Sangiuliano (2019) explored the ES covered by the NMP. Slater and Claydon (2020) find that the NMP needs to set clearer priorities and address ambiguous wording in some policies to improve their directiveness and ability to influence decision making. Moreover, the authors found that the social and economic policies in particular need to be more explicit and directive to be useful within the regulatory process. On the positive side, Slater and Claydon (2020) observed that there seems to be a cooperative and constructive tradition in the approach to adverse impacts in the Scottish marine licensing context and identify this as the reason why very few applications are rejected.

Yet, overall, the authors assessed that the NMP has “released a new forward-looking comprehension of our seas and their future” and that this “is perhaps the greatest achievement of marine planning in the UK” (Slater and Claydon, 2020). Sangiuliano (2019, 51) on the other hand found that Marine Scotland was successful in using an ecosystem approach to planning as is set out in the UK Marine Policy Statement but cautions that there are currently (at the time the article was published, 2019) no benchmarks against which it can be tested whether the NMP addresses the full spectrum of ES sufficiently. He further attests that “the NMP demonstrates remarkable cohesion amongst national and sectoral objectives and policies though the attribution of economy, social, ecosystem, and climate change mitigation and adaptation themes” (Sangiuliano, 2019).

1.6.3 The Scottish Policy Style

The ‘Scottish Policy Style’ is likely to have an influence on blue carbon policy integration and is thus highly important in the context of this thesis. It refers to the Scottish Government’s reputation of making policy following comparatively extensive and inclusive consultation with stakeholders and its willingness to devolve policy implementation to local organisations or authorities (Cairney, 2016a). Keating (2005) and Cairney (2008) already used this term over 15 years ago and less than a decade after devolution. It is thus not a recent concept. However, consultation and forming relationships with pressure participants is also a common response to the limitations of bounded rationality. Within the concept of bounded rationality, it is understood that policymakers have to make decisions with limited resources, which makes it impossible to study all choices and their effects and eliminate all uncertainty and ambiguity. Bounded rationality thus acknowledges that policymakers cannot perfectly translate their aims and values into policy due to these limitations (Cairney, 2016a; Cairney, 2016b) and contrasts with comprehensive rationality that is the ‘ideal-type’ policy process in which policymakers have clear preferences, are able to collate and comprehend all relevant information and then make their choices based on this. Policymaking in comprehensive rationality is described as a perfect cycle, as presented in Figure 1.5, of (i) agenda setting; (ii) policy formulation, (iii) legitimation, (iv) implementation, (v) evaluation, and (vi) policy maintenance, succession, or termination (Cairney, 2016b). Keating (2005) describes a similar cycle but adds ‘problem identification’ as a first step.

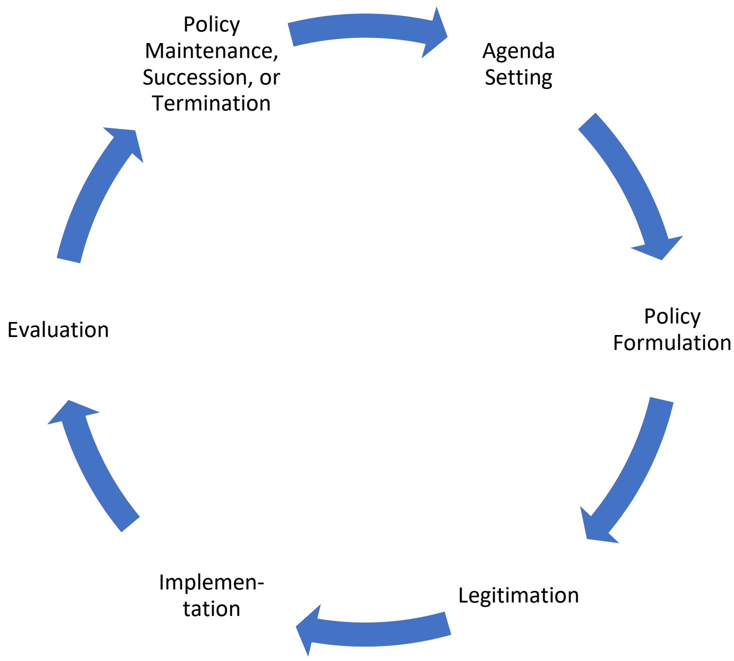


Figure 1.5: A perfect, generic policy cycle (Cairney, 2016b).

Cairney (2014) identifies the way government makes policy and how they implement it as the two dimensions that define a policy style. He comes to the conclusion that, overall, a distinctive policy style can be identified for Scotland. However, while there are notable differences in the policy implementation styles, there are similarities in the Scottish and UK consultation styles during the policy-making stage (Cairney, 2014). According to Greer and Jarman (2008) the British policy style is more top-down and indicates low trust in the implementation providers. Furthermore, it strongly relies on market mechanisms with stringent audit-based procedures as policy tool. In contrast the Scottish government tends to use a more bottom-up approach characterised by a high degree of trust in implementation providers and based on flexible initial policy (Greer and Jarman, 2008). Overall, it is thus possible to argue that there is a strong focus on consultation in both policy development and implementation in Scotland. Cairney (Cairney, 2014; Cairney, 2016a) suggests that this difference in approach may be caused by the small scale of the Scottish government system that allows the development of closer direct and personal relationships between policymakers and interest-groups, which increases the trust in these interest-groups. Moreover, the small scale of the Scottish government body comes with a diminished ability to carry out research and causes an increased need for external experts (Cairney, 2016a). The Scottish policy style, however, did not emerge in one event; in his study on ‘intelligent government’, Sanderson (2010) identifies indications that the Scottish policy style is strengthening. This fits Slater and Claydon’s (2020) observation in the context of Marine Spatial Planning that there seems to be a cooperative and constructive tradition in Marine licensing.

1.7 Research Rationale

Even though the advantages of blue carbon have been established, there is still a limited research base concerning saltmarsh blue carbon in Scotland and its potential for climate change mitigation and adaptation. While there are studies that estimate the carbon stock of Scottish saltmarshes, there is still significant uncertainty concerning the average depth of Scottish saltmarshes, and thus their total OC stock. Moreover, other aspects such as the value of this ES and how it could be incorporated into Scottish policy are under-researched. As previously discussed, ES valuation is beneficial for management decision-making; it is also beneficial to have a clear picture regarding the uncertainties that still exist about the carbon stock since a robust knowledge of the saltmarsh OC stock is helpful for decision-makers who can then prioritise areas for conservation and restoration (Wedding *et al*., 2021). This thesis is thus a valuable contribution for decision-makers who have to make decisions regarding saltmarsh management in Scotland.

The carbon storage ES which saltmarshes provide and the pressing matter of reducing carbon emissions as communicated in the IPCC's special report (2018) and set out in the Paris Agreement (UNFCCC, 2022a) give rise to the overarching assumption that saltmarsh management is best integrated into climate change policy. However, various publications (e.g., Duarte *et al*., 2013; Howard *et al*., 2017; Pendleton *et al*., 2012), including recently published literature on blue carbon policy integration (Dencer-Brown *et al*., 2022; Hilmi *et al*., 2021), highlight the importance of additional benefits provided by blue carbon ecosystems particularly for climate change adaptation. The following chapters will demonstrate that, even though climate change is a pressing issue, other saltmarsh ES should not be disregarded and that it may therefore be more beneficial to integrate saltmarshes and their carbon storage ES into the Scottish MSP framework rather than climate change mitigation policy specifically. The holistic, in-depth interdisciplinary blue carbon study first investigates the uncertainty inherent to saltmarsh data with a focus on the variability of saltmarsh depth in Scotland and what this means for the carbon storage ES, which is a significant aspect for decision-makers and saltmarsh management. This chapter is followed by an economic valuation of the carbon storage ES that investigates the WTP of the Scottish public for the carbon storage ES relative to other saltmarsh ES and their preferences for saltmarsh management, which provides valuable information on the acceptability of potential management measures or policies. Lastly, the incorporation of saltmarsh blue carbon into Scottish policy is investigated in depth. The results are then discussed, and conclusions drawn regarding the overarching initial assumption. Due to the interdisciplinary nature of this thesis, a variety of quantitative and qualitative methods were employed, which motivated the decision to include the methods in the corresponding chapters rather than a separate methods chapter. Analogous to the chapter topics, the methods that were used all link together into a holistic approach to investigate the overarching initial assumption (Figure 1.6): using the quantitative scenario approach to investigate possible carbon stock variations; the stated preference method to elicit the public's preferences and WTP; and the interviews to gain important insights into the Scottish policy process from experts enable well-founded conclusions regarding the place and integration of saltmarsh blue carbon into Scottish policy.

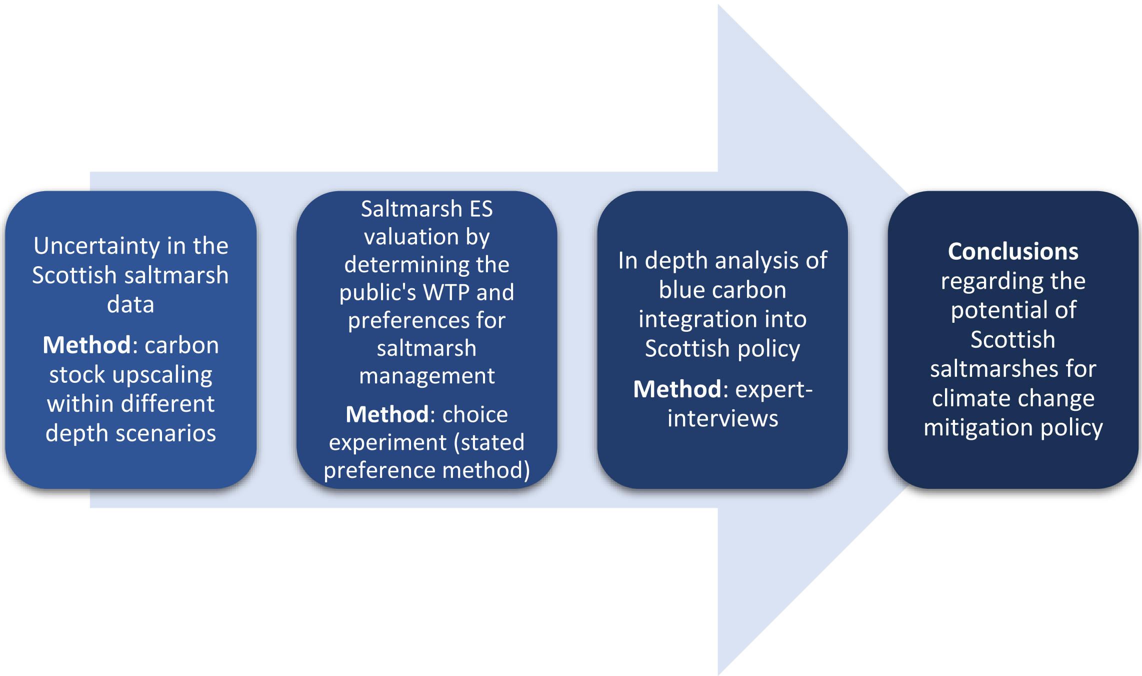


Figure 1.6: Overview of the chapters that investigate the initial overarching assumption and the thesis flow.

Overall, this thesis has the following aims: (i) to highlight the uncertainty inherent to scientific data in the context of saltmarshes and with a focus on the uncertainty of saltmarsh depth; (ii) to determine the Scottish population's preferences regarding the management of saltmarshes and their carbon stock in particular and whether there is support for management that could increase carbon storage; (iii) to determine the Scottish public's WTP for increasing the carbon storage service; and (iv) to determine how saltmarshes can be best included in Scottish policy; and the following main contributions to the climate change mitigation and adaptation literature: (i) it investigates saltmarsh blue carbon's significance for mitigation (Scottish case study); (ii) it investigates the public's preferences for using saltmarsh blue carbon as a NbS for climate change mitigation (Scottish case study); and (iii) it is an in depth analysis of blue carbon policy inclusion (Scottish case study) and what this means in terms of using blue carbon for mitigation and adaptation.

2 The Influence of Soil Depth on Carbon Storage in Scottish Saltmarshes

2.1 Introduction

A saltmarsh's soil OC stock is important for climate change mitigation since it provides information on how much carbon might be released to further amplify the GHG effect if the saltmarsh was destroyed. This can serve as a basis to model carbon sequestration and create carbon stock maps, which are useful tools for decision-makers since they provide information on carbon storage hotspots that should be prioritised for conservation (Pechanec *et al*., 2022; Wedding *et al*., 2021).

Saltmarsh OC stock is determined by three factors: the soil dry bulk density, the OC content, and the soil depth. As previous studies from the UK have demonstrated, the soil dry bulk density and OC content vary within the soil structure of different marshes but also across individual marshes (Austin *et al.*, 2021; Beaumont *et al.*, 2014; Porter *et al.*, 2020; Smeaton *et al.*, 2020). This observation of high variability between these parameters is also valid for the depth of the saltmarshes' soils (Austin *et al.*, 2021; Barlow *et al.*, 2014; Smeaton *et al.*, 2020). The overall average depths of individual saltmarshes can show a high variation and even multiple sampling points on a single marsh can have varying depths. This variability constitutes a significant source of uncertainty when the total OC stock for Scottish saltmarshes is estimated.

Saltmarshes can have varying soil profiles. Smeaton *et al.* (2020) identified four main units for the Kyle of Tongue saltmarsh: (i) a fibrous peat layer, (ii) a humified peat layer, and (iii) an organic rich silt layer, all of which sit upon (iv) a basal layer of marine mud. However, these layers are not uniform throughout the entire saltmarsh, as was demonstrated by the detailed graphic of the Kyle of Tongue saltmarsh soil profile presented by Barlow *et al.* (2014). Generally, however, Scottish saltmarsh soil layers can be described (Figure 2.1). Saltmarsh vegetation grows on the fibrous peat layer which is the uppermost layer in the soil profile and formed of peat-like organic matter interspersed with a large number of living roots. Below the peat layer is the humified peat layer which consists of humified organic matter of decomposing vegetation. A transitional layer sits between the humified peat and the basal base layer and is characterised by similarities with the basal layer but also a higher quantity of organic matter. It represents the pioneer saltmarsh formation and thus the transition from intertidal flat to saltmarsh. Below these layers sits the basal layer with low quantities of organic matter that consists of either fine silt or sand depending on whether the pre-saltmarsh intertidal environment was a mudflat or sandflat (Miller *et al.*, In Revision).

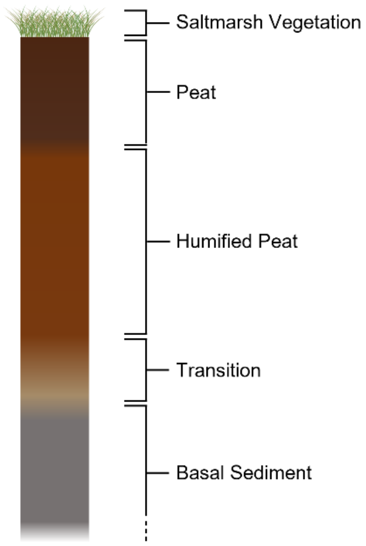


Figure 2.1: General soil profile identified in Scottish saltmarshes.

The depth of these soil layers can vary significantly and are different for every saltmarsh. It is also possible that not all layers are present in every sampling location (Barlow *et al.*, 2014). Since this thesis is focussed on saltmarsh blue carbon and its use for climate change mitigation and not on the pre-existing mud- or sandflat environments, the focus of this chapter is on the three identified saltmarsh layers (i.e., peat, humified peat, and the transitional layer) and the analysis excludes the underlying basal layer.

Depending on the location and age of the saltmarsh, their soil depth can be very significant. Scott and Greenberg (1983), for example, reported vertical deposits as deep as 8 m for a saltmarsh in the Bay of Fundy tidal system. Pendleton *et al.* (2012) reported that vegetated coastal ecosystems typically reside over organic-rich sediments that may be several meters deep and that "larger amounts of carbon are often held in as much as 6 meters of sediment and biomass beneath the emergent vegetation" (Pendleton *et al.* 2012, 4). However, the use of the term 'sediment' in Pendleton *et al.*'s (2012) statement indicates that the underlying basal layers may have been included in this depth and not just the saltmarsh soil. Haynes (2016a) reports that saltmarsh depth is related to saltmarsh type and that estuarine systems are usually associated with deeper sediments. Further, Haynes' (2016a) report points out that saltmarshes are deeper towards the landward limit of the marsh and specifies that Scottish saltmarshes are generally between 20 cm and 1-3 m deep.[8] This maximum depth is not confirmed by other studies on Scottish saltmarshes (Smeaton *et al.*, 2020; Porter *et al.*, 2020) and it may be the case that Haynes' (2016a) depth measurements also include the underlying basal sediments, which is indicated by the term 'sediment'.

[8] In this study, the term 'soil' refers to the saltmarsh soil layers, whereas the term 'sediment' refers to the basal sediments below the saltmarsh.

The deepest saltmarsh soil layers of Scottish saltmarshes were observed during fieldwork when collecting cores from the Waulkmill Bay saltmarsh; the layers extracted in one core were 69 cm deep. Furthermore, the saltmarsh layers for Loch Laxford were reported as 68.5 cm deep in one sampling location (Barlow *et al.*, 2014). The currently known maximum depth is hence far removed from the internationally reported depths above. Moreover, these thick saltmarsh layers observed for the Waulkmill Bay and Loch Laxford saltmarshes can be considered as outliers considering the currently available data for Scotland, removing the expected Scottish average saltmarsh depth even further from these deep examples.

This chapter analyses and compares two different Scottish saltmarsh datasets: data collected during fieldwork of two Orkney saltmarshes (results published in Porter *et al.*, 2020); and the dataset collected and published by Ruranska *et al.* (2020). It focuses on reducing the uncertainty associated with the depth of Scottish saltmarshes and investigates the influence of saltmarsh soil depths for the total Scottish saltmarsh OC stock. Saltmarsh depth is an important factor in calculating saltmarsh OC stock and it is thus likely to be insufficient to only consider the top 10 cm saltmarsh soil to estimate the saltmarsh OC stock for the total Scottish saltmarsh area, which is the approach taken by Austin *et al*. (2021). This analysis reduces the uncertainty inherent to saltmarsh OC stock estimates for decision-makers and while the estimates cannot be absolutely accurate, they are an improvement and beneficial for the decision-making process.

2.2 Aim of Chapter

In the overall context of this thesis, this chapter will highlight the uncertainty that is inherent to scientific data and focus on saltmarsh depth as one major factor of uncertainty regarding the OC stock in Scottish saltmarshes. It also opens the discussion on how these uncertainties can be managed in policy which will be addressed in Chapter 4.

2.3 Methodology

2.3.1 Data Collection Case Study: Two Orkney Islands Saltmarshes

Samples were collected from two Orkney Islands saltmarshes during fieldwork in February 2019. The fieldtrip was organised as part of the NERC C-SIDE project (NERC C-SIDE, 2022). The laboratory analysis of the data, the replication and correction of the calculations published in Porter *et al.* (2020), and the additional calculations in this chapter were conducted by the author.

2.3.1.1 Sampling

Only one sampling technique was used in this study to retrieve saltmarsh cores. A hand-operated 25 mm gauge corer (Figure 2.2) was used that allowed coring to a maximum depth of 1 m. The corer was pushed into the soil by hand as opposed to hammering it in, which avoids strong soil compaction as demonstrated by Smeaton *et al.* (2020). Due to the limited length of 1 m, there is a risk that a core of the full saltmarsh soil cannot be retrieved; however, this is unlikely in the Scottish context where saltmarsh soil depths exceeding 1 m, are not expected. Previous studies that employed saltmarsh coring as a method did not record saltmarsh soils with such large depths (Barlow *et al.*, 2014; Smeaton *et al.*, 2020). Gauge corers are a trusted tool to retrieve soil cores that have been used extensively in sea-level reconstructions and blue carbon research (Barlow *et al.*, 2014; van Ardenne *et al.*, 2018; Wollenberg *et al.*, 2018)

Figure 2.2: 25 mm gauge corer used for extracting cores of the Waulkmill Bay and Loch of Stenness saltmarshes.

The Loch of Stenness saltmarsh (named Bridge of Waithe and Cummi Ness in Haynes' (2016a; 2016b) dataset) is a fringing saltmarsh at the connection between Scapa Flow and the Loch of Stenness. Due to its layout, it was not possible to retrieve cores in a straight transect; instead, the natural shape of the marsh was followed. At the Waulkmill Bay saltmarsh, the first 7 cores were taken on an east-west transect from the edge of the marsh to the centre and a stream that bisects the marsh. The remaining 4 cores were retrieved further to the north and north-west of the initial transect. At each site, the GPS data was recorded using DGPS (Figures 2.3 and 2.4). A 1 m gauge corer, measuring 25 mm in diameter was used to retrieve the cores. The extracted cores were longitudinally sectioned with a sharp knife to receive a clearer picture of the soil stratigraphy, which was classified according to the Tröels-Smith (1995) classification scheme for unconsolidated sediments (Appendix A, Table A.1). Fixed volume samples of 4 cm^3 were collected at 0-2 cm, 4-5 cm, 10-11 cm and then every 10 cm depending on the depth of the core. Overall, 17 cores were retrieved, and 76 subsamples taken at the Loch of Stenness saltmarsh, and 11 cores and 72 subsamples at the Waulkmill Bay saltmarsh. The total number of samples was thus 148.

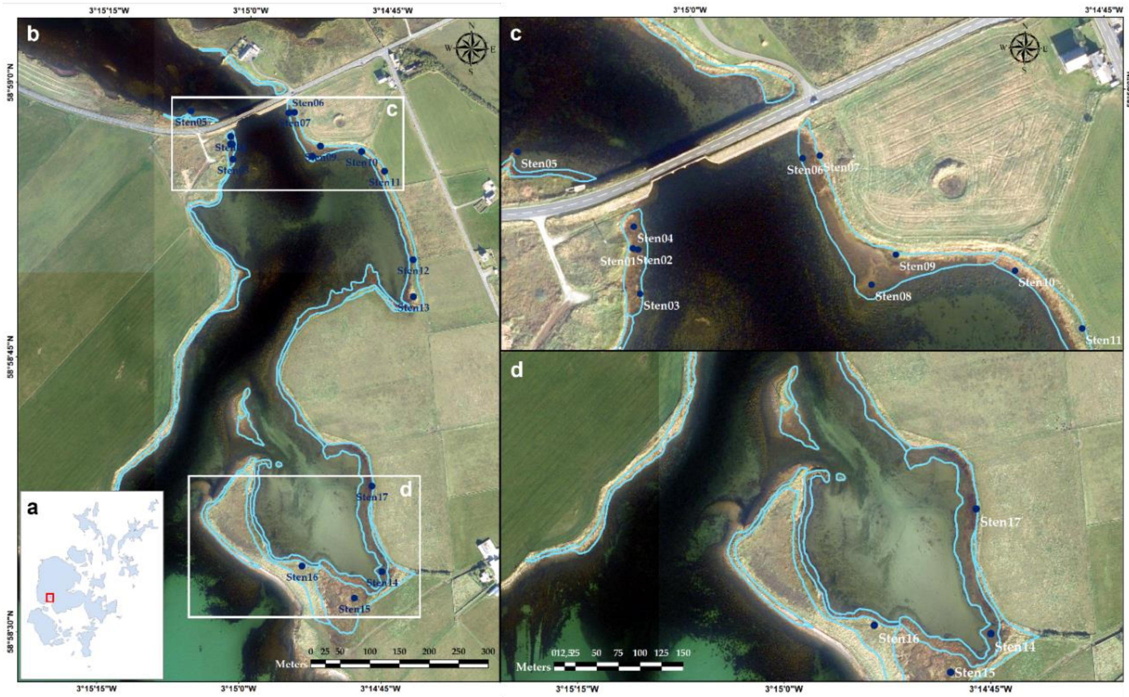


Figure 2.3: Location map of the saltmarsh on Mainland Orkney (a) and the core locations on the Loch of Stenness saltmarsh (b) with more detailed views (c) (d). Contains OS data © Crown copyright and database right (2021).

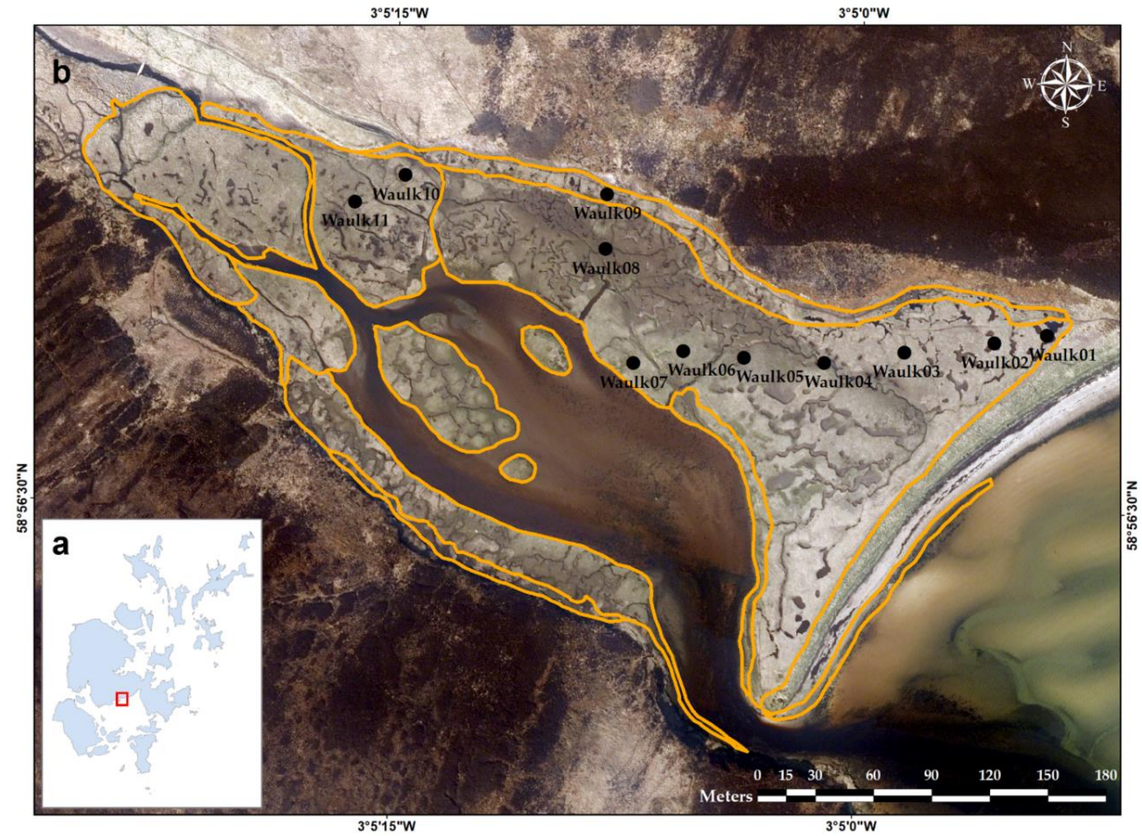


Figure 2.4: Location map detailing the location of the saltmarsh on Mainland Orkney (a) and the core locations on the Waulkmill Bay saltmarsh (b). Contains OS data © Crown copyright and database right (2021).

2.3.1.2 Bulk Density

Bulk Density is an important variable needed to calculate the carbon stock of saltmarshes. It can be reported as dry or wet bulk density. Wet bulk density is defined as the ratio of the wet sample weight to its volume (1), while the dry bulk density is defined as the ratio of the dry sample weight to the total volume of the wet sample (2) (Robinson *et al.*, 2022; Smeaton *et al.*, 2020). To calculate the bulk density, the fixed volume samples (4 cm^3) were weighed while wet and after drying at 55°C for four days. This method is in line with Dadey *et al.* (1992), except that a longer drying process was chosen with a lower temperature.

$$\text{Wet bulk density (g cm}^{-3}\text{)}: \frac{Wet\ mass\ (g)}{Wet\ sample\ volume\ (cm^3)} \quad (1)$$

$$\text{Dry bulk density (g cm}^{-3}\text{)}: \frac{Dry\ mass\ (g)}{Wet\ sample\ volume\ (cm^3)} \quad (2)$$

In this study, the dry bulk density was determined to calculate the carbon stock estimates.

2.3.1.3 Elemental Analysis

The samples were prepared for analysis by drying at a low temperature (55°C) as described above, followed by milling a small portion of each sample with a mortar and pestle to create a fine homogenised powder. To measure the OC content, IC needed to be removed from the milled subsample. The subsamples (10 ± 0.06 mg) were placed in a silver capsule to which 61µl of hydrochloric acid (HCl, 10%) was added to remove IC. To ensure full IC removal, the subsamples rested overnight before the analysis was continued (Verardo *et al.*, 1990, Nieuwenhuize *et al.*, 1994).

The OC content was determined with an Element Analyser (EA, Elementar Vario EL cube Element Analyzer). Once the samples had dried, the capsules were closed and stored in a glass desiccator until the EA could be loaded. The EA was calibrated with acetanilide (Reference material B2178) (Verardo *et al.*, 1990; Nieuwenhuize *et al.*, 1994). The prepared samples in silver capsules were loaded into the EA's automatic sampler and then combusted in an oxygen-rich environment at 1200°C. In this process, carbon was converted into CO_2, hydrogen into water (H_2O), nitrogen into nitrogen gas (N_2), and sulphur to sulphur dioxide (SO_2). An inert carrier gas, such as Helium or Argon, then carried these gases over a combustion catalyst (Cr_2O_3) and silver wool, to remove halides, before sweeping them into a reduction tube with high purity copper, heated to 800°C to remove any oxygen not consumed during combustion and a magnesium perchlorate trap to remove water. The gases then passed through absorbent traps (Instrumental Criteria Sub-committee, 2008) before being passed through the Thermal Conductivity Detector (TDC) (Figure 2.5) which senses changes in thermal conductivity of gases and compares these to the carrier reference gas. The EA contains three separation columns that absorb CO_2, H_2O, and SO_2. The nitrogen is not absorbed and passes through the TDC first. Once the nitrogen peak has been detected, the CO_2 tube was heated up and the gas released passed though the TDC. This process was then repeated with the H_2O and SO_2 columns, respectively. This separation process ensured that the gasses reach the TDC separately and could be measured. The elemental detection limit was < 40 ppm (Elementar, year unknown).

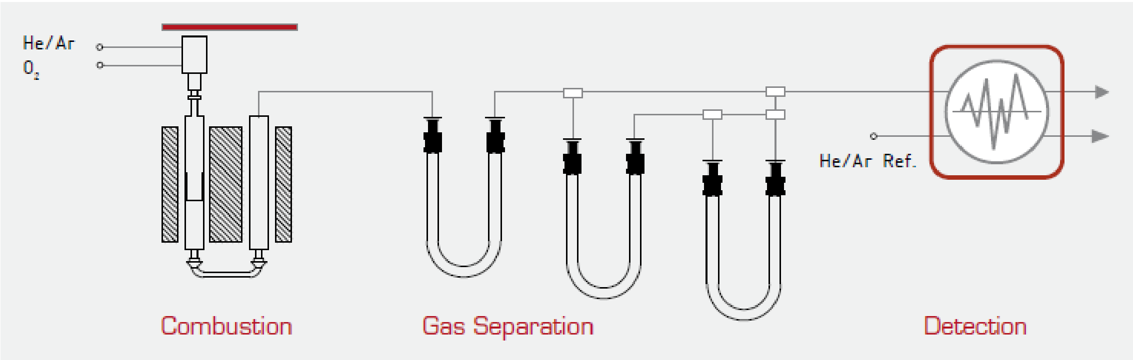


Figure 2.5: Elementar EA schematic (Elementar).

2.3.2 The Scenario Approach

The "Scenario Approach" is an Environmental Management tool (Barrow, 2005) and well-established in climate change science. The IPCC 5th Assessment Report uses the Representative Concentration Pathways (RCPs) as standard set of scenarios for future climate change. The RCPs describe four potential 21st century pathways of GHG emissions. These include a strong mitigation scenario (RCP 2.6), two intermediate scenarios (RCP 4.5 and RCP 6.0), and one scenario with little mitigation efforts and very high GHG emissions (RCP 8.5). The costs for the emission reductions that are necessary to achieve the scenarios can be calculated (IPCC, 2014b). Moreover, the scenarios can also form the foundation for relative sea-level change scenarios (IPCC, 2013).

This study takes a similar approach and develops OC stock scenarios. However, in this case, the scenarios are not time-based as the RCPs, but based on different saltmarsh depths. As established, there is still significant uncertainty regarding the depth of Scottish saltmarshes. Five data sources for Scottish saltmarsh depth were available to this study. Barlow *et al.* (2014) conducted work on the saltmarshes at Loch Laxford and the Kyle of Tongue and created soil and sediment profiles that can be used to determine the depth of the saltmarsh layers. Smeaton *et al.*'s (2020) study focuses on the Kyle of Tongue saltmarsh and provides a further data source regarding its depth. Lastly, saltmarsh samples were collected, and their depth recorded during coring fieldwork on two Orkney saltmarshes, the saltmarsh at the entry point of Loch of Stenness to Scapa Flow and the Waulkmill Bay saltmarsh. The average saltmarsh depth that was calculated from these data sources was used as the foundation for developing the three scenarios.

The average depths for the Loch Laxford and the Kyle of Tongue saltmarshes were 39 ± 22 cm and 24 ± 16 cm, respectively according to the data points from Barlow *et al.*'s (2014) study. However, Smeaton *et al.*'s (2020) datapoints suggest that the saltmarsh soil layers at the Kyle of Tongue are deeper; the average depth is 37 ± 23 cm. This result is surprising since the location of the sampling transects are very similar for both studies (Figure 2.6).

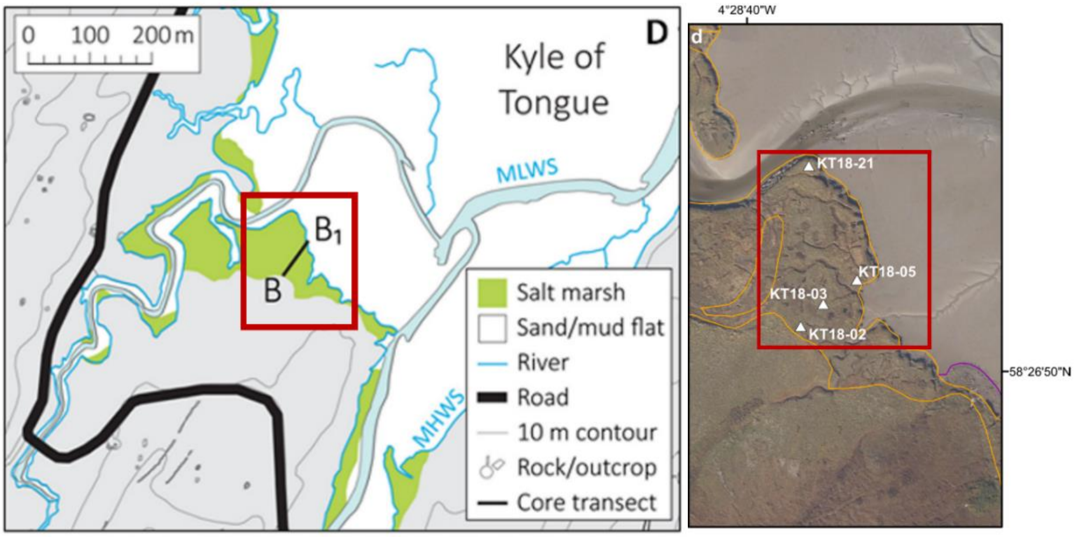


Figure 2.6: Sampling points of Barlow *et al.* (2014) (D) and Smeaton *et al.* (2020) (d).

The average depths of the Loch of Stenness and Waulkmill Bay saltmarshes were 18 ± 11 cm and 31 ± 19 cm, respectively. Hence there is a significant range of depths for Scottish saltmarshes, which is further highlighted by the wide error ranges that can be observed for all of these average saltmarsh depths. The average depth of these four saltmarshes is 30 cm with an average standard deviation of 18 cm. All average saltmarsh depths that were included in this calculation lie within the standard deviation range of the calculated average. 30 cm depth for Scottish saltmarshes was thus deemed appropriate for the middle scenario. A higher and lower scenario were then set. The lower scenario with 20 cm saltmarsh depth was selected since this depth is within the average standard deviation and Austin *et al.* (2021) interpolated the Scottish saltmarsh surface stock to 15 cm depth. The higher scenario was set at 40 cm depth and thus also within the average standard deviation.

2.3.3 Carbon Stock Upscaling to the National Level

To calculate the OC stock of a saltmarsh, three calculation steps, as laid out in Smeaton *et al.* (2020) are necessary. First, the Soil Volume needs to be calculated (1) from the Area (m^2) and saltmarsh depth (m); then the Soil Mass (kg) needs to be calculated (2) from the previously determined Volume (m^3) and Dry Bulk Density (kg m^{-3}); the final step is the estimation of the OC Stock, which is the product of the Soil Dry Mass (kg) and the OC content (%) determined in the elemental analysis.

$$Volume\ (m^3)\ =\ Area\ (m^2)\ x\ \ Soil\ Depth\ (m) \quad (1)$$

$$Dry\ Mass\ (kg)\ =\ Volume\ (m^3)\ x\ Dry\ Bulk\ Density\ (kg\ m^3) \quad (2)$$

$$OC\ Stock\ (kg) =\ Dry\ Mass\ (kg)\ x\ OC\ content\ (\%) \quad (3)$$

The total Scottish saltmarsh area was derived from the dataset published by Haynes (2016a; 2016b). The total area was adjusted to exclude perched saltmarshes, which, as already discussed, do not have a significant soil layer. When calculating the OC stock of the Loch of Stenness and Waulkmill Bay saltmarshes, the depth of the saltmarsh soil layers and the samples that where within the range of these layers were included. Data from the underlying basal layer were excluded.

2.4 Results

2.4.1 Waulkmill Bay and Loch of Stenness Carbon Stock

Since it was the aim of this chapter to determine the OC stock of the saltmarsh layers, the core subsamples of the sediment underlying the saltmarsh soil needed to be identified and excluded from this analysis. As mentioned in section 2.3.1.1, the Tröels-Smith (1955) classification was used to describe the soil and sediment units and to make this differentiation. The subsamples of the underlying basal sediment were thus excluded from the stock calculation since they can be allocated to the pre-saltmarsh environment; included were the subsamples that were classified as part of the peat layers, both fibrous and humified peat, and the transitional layer.

The two sampled saltmarshes had varying depths. However, this was to be expected since the two marshes are different types and as Haynes (2016a) points out, the depth of saltmarshes is related to the typology. Haynes (2016a) also states that estuarine saltmarshes are usually deeper, which is reflected in the collected core data. The Waulkmill Bay saltmarsh is an estuarine saltmarsh protected behind a spit of land (Porter *et al*., 2020) with an average depth of 31 ± 19 cm, while the Loch of Stenness saltmarsh is a fringing saltmarsh and only 18 ± 11 cm deep on average. Both saltmarshes had similar dry bulk density profiles (Figure 2.7), but it was noticeable that the bulk density curve for the Waulkmill Bay saltmarsh was much smoother than the curve for the Loch of Stenness saltmarsh. This may be due to the scarcity of sampling points below 20.5 cm for the Loch of Stenness saltmarsh. Nevertheless, an increase in dry bulk density can be observed with increasing depth; the saltmarsh soil layers have thus a lower bulk density than the underlying basal sediment. This observation is in line with other studies and fits the general observation that bulk density increases with depth due to compaction causing a reduction of porosity (Smeaton *et al.*, 2020; Macreadie *et al.*, 2012). The average dry bulk density of the Loch of Stenness saltmarsh was overall slightly higher than the average dry bulk density of the Waulkmill Bay saltmarsh.

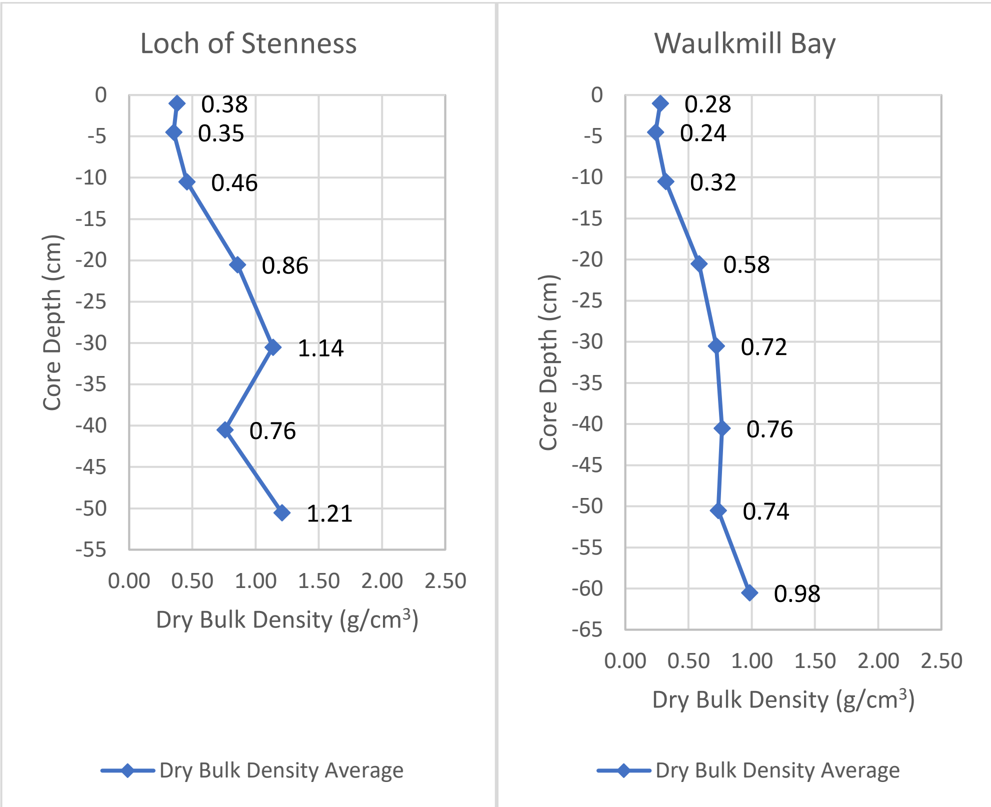


Figure 2.7: Average dry bulk densities (g/cm^3) of the Loch of Stenness and Waulkmill Bay saltmarshes.

As with the dry bulk density, similar OC content values were found for the two saltmarshes (Figure 2.8). The OC content decreases with depth; the basal layers and the transitional saltmarsh soil layer had a lower OC content than the peat layers of the saltmarsh closer to the marsh surface. As with the average dry bulk density, the average OC content of the Loch of Stenness marsh is slightly higher than the average OC content of the Waulkmill Bay marsh. Overall, the OC contents of the two saltmarshes are comparable with the observed OC contents of other Scottish saltmarsh studies (Austin *et al.*, 2021; Smeaton *et al.*, 2020).

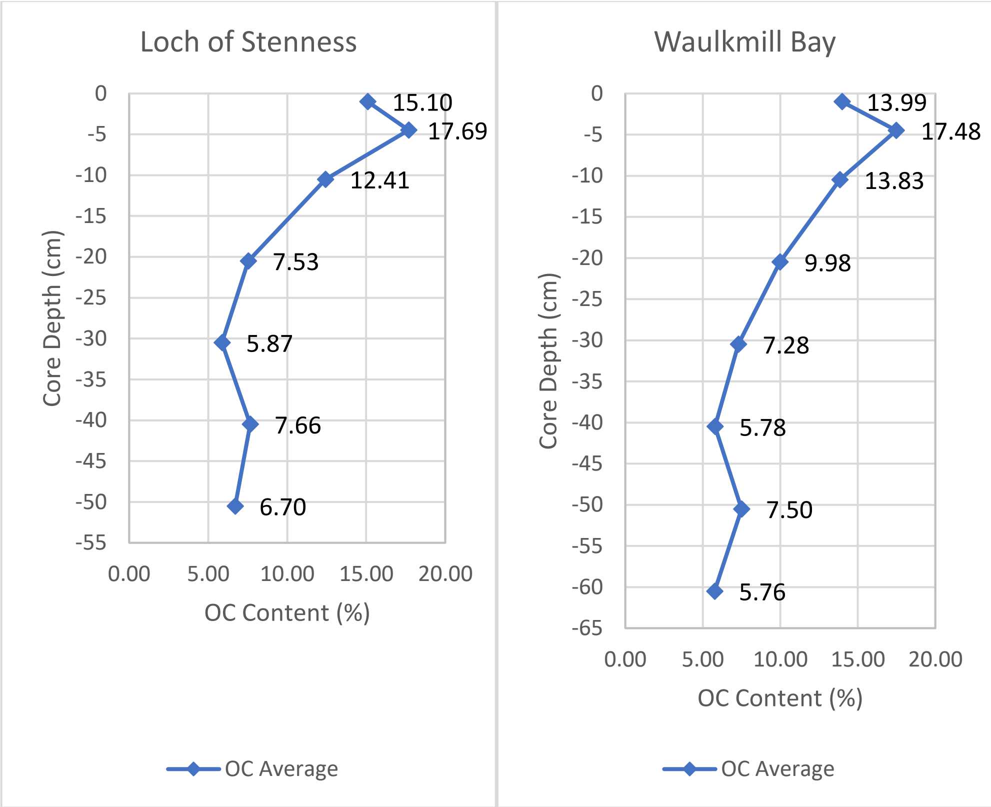


Figure 2.8: Average OC content (%) of the Loch of Stenness and Waulkmill Bay saltmarshes.

The estimated OC stock of the Waulkmill Bay saltmarsh is 816 ± 126 tonnes (Table 2.1) and exceeds the estimated OC stock of the Loch of Stenness saltmarsh (464 ± 75 tonnes), even though the Waulkmill Bay marsh data showed a lower dry bulk density as well as a lower OC content. These two factors are compensated by the larger extent of the Waulkmill Bay marsh and its deeper saltmarsh soil. The estimated OC stock has large error ranges associated with variability in depth, dry bulk density, and OC content. For comparability with the data published in Austin *et al.* (2021), the OC stock was also calculated for the top 10 and 15 cm of both saltmarshes (Table 2.1, Figure 2.9). Only the data from subsamples within these depth ranges was used for these calculations, hence the varying bulk densities and OC contents compared to the estimates for the full average saltmarsh depth. For both the top 10 and 15 cm of soil, the OC stock estimate for the Waulkmill Bay saltmarsh was lower than the estimate for the Loch of Stenness saltmarsh. The larger extent of the Waulkmill Bay marsh was thus not sufficient to compensate for the lower average dry bulk density and OC content. Only the greater depth of the Waulkmill Bay saltmarsh elevates its OC stock above that of the Loch of Stenness saltmarsh thus confirming that saltmarsh depth is a very important factor concerning the OC stock.

Table 2.1: Soil OC Stocks of the Loch of Stenness and Waulkmill Bay saltmarshes (the top 10 and 15 cm estimates have different bulk densities and OC contents since only the relevant data was used for these depths instead of the data from all available subsamples and depths).

Saltmarsh	Area (ha)	Depth (m)	Dry Bulk Density (kg m^{-3})	OC Content (%)	Soil OC Stock (tonnes)
			Full average depth		
Loch of Stenness	4.23	0.18 ± 0.11	416 ± 203	14.64 ± 7.98	464 ± 75
Waulkmill Bay	5.54	0.31 ± 0.19	345 ± 169	13.78 ± 7.1	816 ± 126
			Top 10 cm		
Loch of Stenness	4.23	0.10	362 ± 144	16.26 ± 8.17	249 ± 50
Waulkmill Bay	5.54	0.10	257 ± 108	15.74 ± 8.22	224 ± 49
			Top 15 cm		
Loch of Stenness	4.23	0.15	368 ± 146	15.63 ± 8.02	365 ± 74
Waulkmill Bay	5.54	0.15	262 ± 106	15.77 ± 7.91	343 ± 70

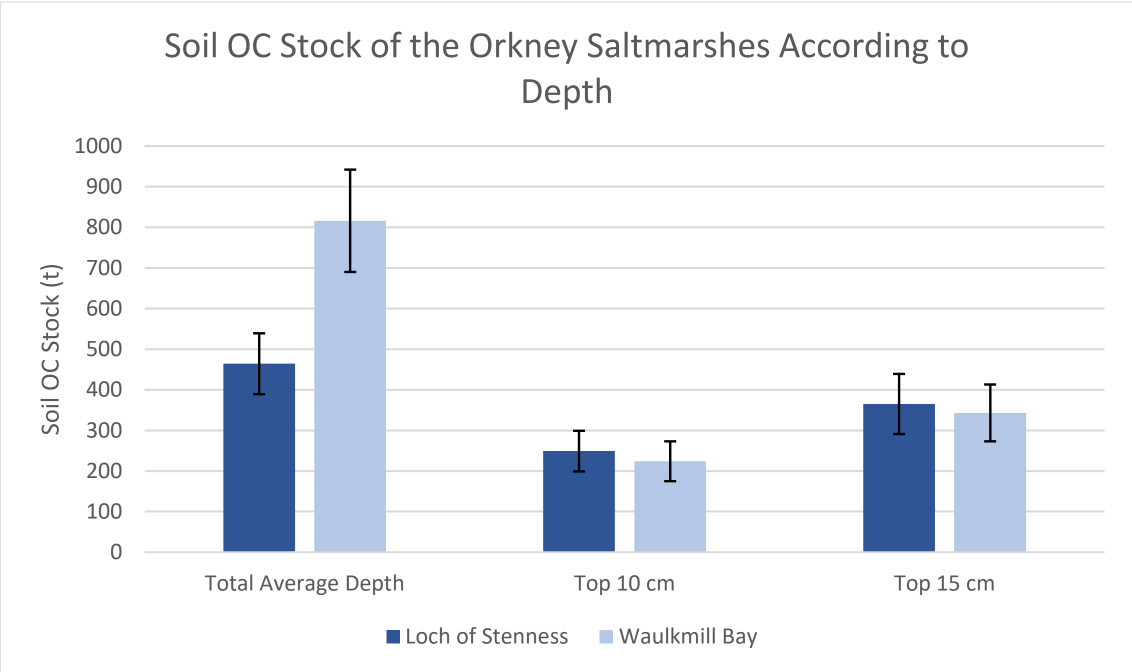


Figure 2.9: Soil OC stocks of the Loch of Stenness and Waulkmill Bay saltmarshes.

In Austin *et al.* (2021), the OC stock of the top 10 and 15 cm was estimated with average data from several hundred sites across Scotland (Ruranska *et al.*, 2020) instead of only the data from the Loch of Stenness and Waulkmill Bay saltmarshes. According to the supplemental data for the report (Smeaton *et al.*, 2021), the top 10 cm soil OC stock of the Loch of Stenness and Waulkmill Bay marshes are 240 and 362 tonnes, respectively. Compared to the estimates derived from the marsh specific data, this result is fairly close to the estimation for the Loch of Stenness saltmarsh but a significant over-estimation for the Waulkmill Bay saltmarsh; the estimate surpasses the stock estimate calculated with marsh-specific data by over 100 tonnes. This result is also reflected in the OC stock estimates for the top 15 cm of the saltmarsh soil. The estimate published in the supplemental data to the Scotland report is 359 tonnes for the Loch of Stenness marsh and 543 tonnes for the Waulkmill Bay saltmarsh. Again, the estimate is slightly below the Loch of Stenness estimate and much higher than the Waulkmill Bay estimate.

2.4.2 The Upscaling Scenarios

The different depth scenarios were developed as described in section 2.3.2 on the scenario approach. Scenario 1 assumes an average Scottish saltmarsh depth of 20 cm, Scenario 2 of 30 cm, and Scenario 3 of 40 cm. These three scenarios were calculated with both the data from the two Orkney saltmarshes and the top 10 cm surface soil data (Ruranska *et al.*, 2020) the report by Austin *et al.* (2021) is based on.

2.4.2.1 Carbon Stock Upscaling with Waulkmill Bay and Loch of Stenness Data

When calculating the three scenarios with the data from the two Orkney saltmarshes, the data of the two separate marshes needs to be combined. This can be done by combining the data from the subsamples of both saltmarshes and calculating the average dry bulk density and OC content. Following this approach, the average dry bulk density for both Orkney saltmarshes was 384 ± 191 kg m^{-3}, and the OC content was 14.24 ± 7.56 %. It is also possible to calculate first the average dry bulk density and OC content for each marsh by using the data of the respective core subsamples. In a second step, the average dry bulk density and OC content for both marshes were then calculated from the determined averages for each marsh. When this approach was followed, the average dry bulk density was 381 ± 186 and the OC content was 14.21 ± 7.54 %. The difference between these results does not seem substantial; however, when upscaled to the total Scottish saltmarsh area, its effect becomes apparent. There is already a difference of almost 5000 tonnes in OC stock estimates between the two upscaling methods in the 20 cm depth scenario based on these two different averages (Table 2.2, Figure 2.10). For the 30 cm upscaling calculations, this difference grows to almost 7500 tonnes, and for the 40 cm depth scenario, it grows to almost 10000 tonnes. However, these numbers are only a small fraction of the total OC stock and do not have a substantial impact on the overall estimate. To put it into perspective, 5000 tonnes are only 0.8%, 7500 tonnes 1.2%, and 10000 tonnes 1.6% of the total OC stock (total OC stock calculated with 384 ± 191 kg m^{-3} dry bulk density and 14.24 ± 7.56 % OC content). Yet, calculating the average of an average introduces some uncertainty into the data which could be avoided; thus, the first method where the average dry bulk density and OC content are calculated directly without the intermediate step is preferable. The OC stock estimates for Scottish saltmarshes are thus 635195 ± 168088 tonnes for the 20 cm depth scenario, 952793 ± 252132 tonnes for the 30 cm depth scenario, and 1270391 ± 336176 tonnes for the 40 cm depth scenario. With each additional 10 cm depth, the OC stock hence increases by 317598 ± 84044 tonnes. It needs to be noted that the linear increase in OC stock with increasing depth as observed here is not necessarily accurate; it is the product of using an average dry bulk density and OC content for the entire depth of the scenario. It is expected that the OC stock increases get larger with increasing depth due to the increasing bulk density unless the OC content drops very significantly to offset this effect. Using the average dry bulk density may thus cause an overestimation for the upper soil layers and an underestimation for the deeper soil layers.

Table 2.2: Upscaling scenarios with the Loch of Stenness and Waulkmill Bay saltmarsh data and different methods to calculate the average dry bulk density and OC content.

Scenario	Area (ha)	Depth (m)	Dry Bulk Density (kg m^{-3})	OC Content (%)	Soil OC Stock (tonnes)
Dry bulk density and OC content averages of all Loch of Stenness and Waulkmill Bay samples					
20 cm depth	5820.39	0.20	384 ± 191	14.24 ± 7.56	635195 ± 168088
30 cm depth	5820.39	0.30	384 ± 191	14.24 ± 7.56	952793 ± 252132
40 cm depth	5820.39	0.40	384 ± 191	14.24 ± 7.56	1270391 ± 336176
Averages of the average dry bulk densities and OC contents of the Loch of Stenness and Waulkmill Bay saltmarshes					
20 cm depth	5820.39	0.20	381 ± 186	14.21 ± 7.54	630233 ± 163255
30 cm depth	5820.39	0.30	381 ± 186	14.21 ± 7.54	945349 ± 244882
40 cm depth	5820.39	0.40	381 ± 186	14.21 ± 7.54	1260466 ± 326510

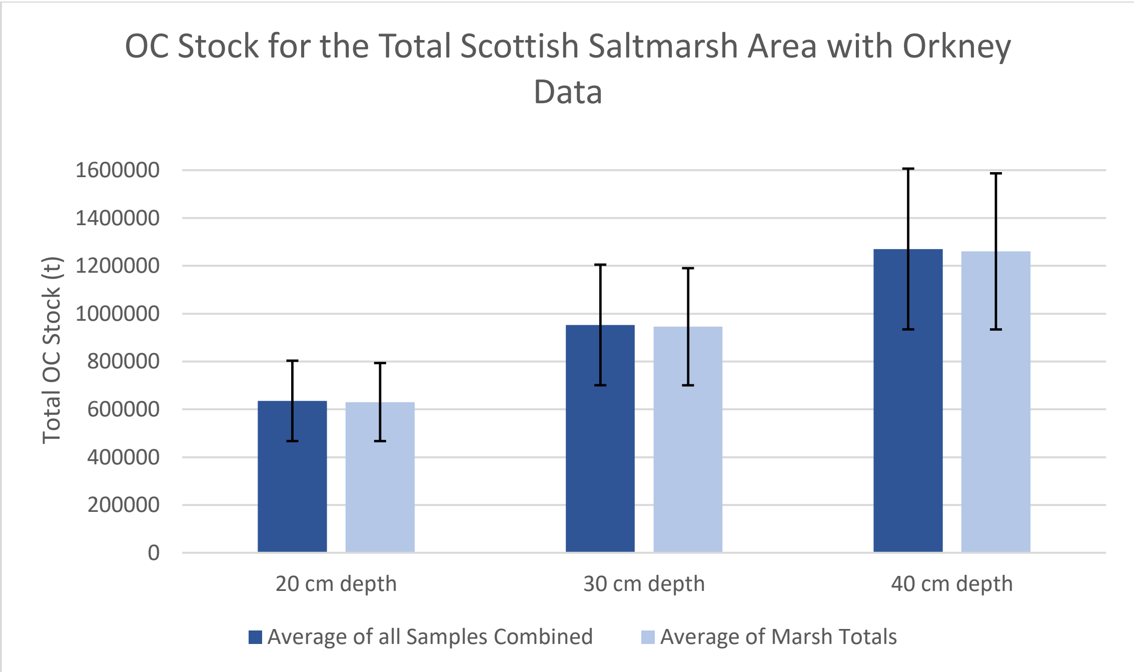


Figure 2.10: A comparison of the OC stock upscaling results for the total Scottish saltmarsh area based on two averages of the Orkney saltmarsh data that were obtained with two different averaging methods.

When only the OC stock of the top 10 and 15 cm of both the Loch of Stenness and Waulkmill Bay saltmarshes were calculated, the data of subsamples of deeper soil layers was excluded. For the scenarios, however, the average dry bulk density and OC content were calculated from all saltmarsh soil subsamples irrespective of their depth. It was reasonable previously to exclude samples of greater depth since the aim of the calculation was a comparison with the top 10 cm and 15 cm OC stock estimates of Smeaton *et al.* (2021). However, in this case, the aim is to calculate OC estimates for different average saltmarsh depths of the total Scottish saltmarsh area. Scotland has many different saltmarsh types and numerous saltmarshes with varying extent. It is thus expected that these saltmarshes will have varying dry bulk densities and OC contents. Including the data from all saltmarsh depths with varying dry bulk densities and OC contents was thus determined as the preferable approach rather than limiting the included samples due to their depth and diverging dry bulk density and OC content.

2.4.2.2 Carbon Stock Upscaling with Soil Surface Data from all over Scotland

The dataset Austin *et al.*'s (2021) report is based on (Ruranska *et al.,* 2020) has a total of 805 data points consisting of 378 dry bulk density measurements and 427 OC contents (Austin *et al.*, 2021). As previously mentioned, subsamples were only taken up to a depth of 10 cm. It can thus be expected that the average dry bulk density is an underestimation and the average OC content an overestimation since it was previously established that dry bulk density increases with depth and OC content decreases with depth. OC stock estimations based on this data are thus more likely to be underestimations rather than overestimations unless the decreasing OC content with depth fully compensates for this effect. Compared to the average dry bulk density of the Orkney saltmarshes, the average dry bulk density determined from these datapoints is over 100 kg m^{-3} higher, yet the average OC density is just over 1% lower than the average OC density of the two Orkney marshes (Table 2.3). Accordingly, the OC stock estimates for Scottish saltmarshes are thus 749284 ± 253575 tonnes for the 20 cm depth scenario, 1123927 ± 380362 tonnes for the 30 cm depth scenario, and 1498569 ± 507149 tonnes for the 40 cm depth scenario.[9]

Table 2.3: Upscaling scenarios with the top 10 cm soil Scottish saltmarsh data (Ruranska *et al.*, 2020) published in Austin *et al.* (2021).

Scenario	Area (ha)	Depth (m)	Dry Bulk Density (kg m^{-3})	OC Content (%)	Soil OC Stock (tonnes)
20 cm depth	5820.39	0.20	488 ± 287	13.19 ± 7.59	749284 ± 253575
30 cm depth	5820.39	0.30	488 ± 287	13.19 ± 7.59	1123927 ± 380362
40 cm depth	5820.39	0.40	488 ± 287	13.19 ± 7.59	1498569 ± 507149

When comparing these OC stock estimates for the three scenarios (Figure 2.11), it becomes clear that the higher dry bulk density of this dataset has a very significant effect on the OC stock estimates. For the 20 cm depth scenario, the OC stock estimation is almost 115000 tonnes higher, for the 30 cm depth scenario, the estimate is over 170000 tonnes higher, and for the 40 cm depth scenario, the OC stock estimate is almost 230000 tonnes higher.

[9] A t-test for unequal variances was carried out to assess whether these differences between the datasets are statistically significant. The result was $p < 0.001$. The differences are thus statistically significant with a high level of confidence.

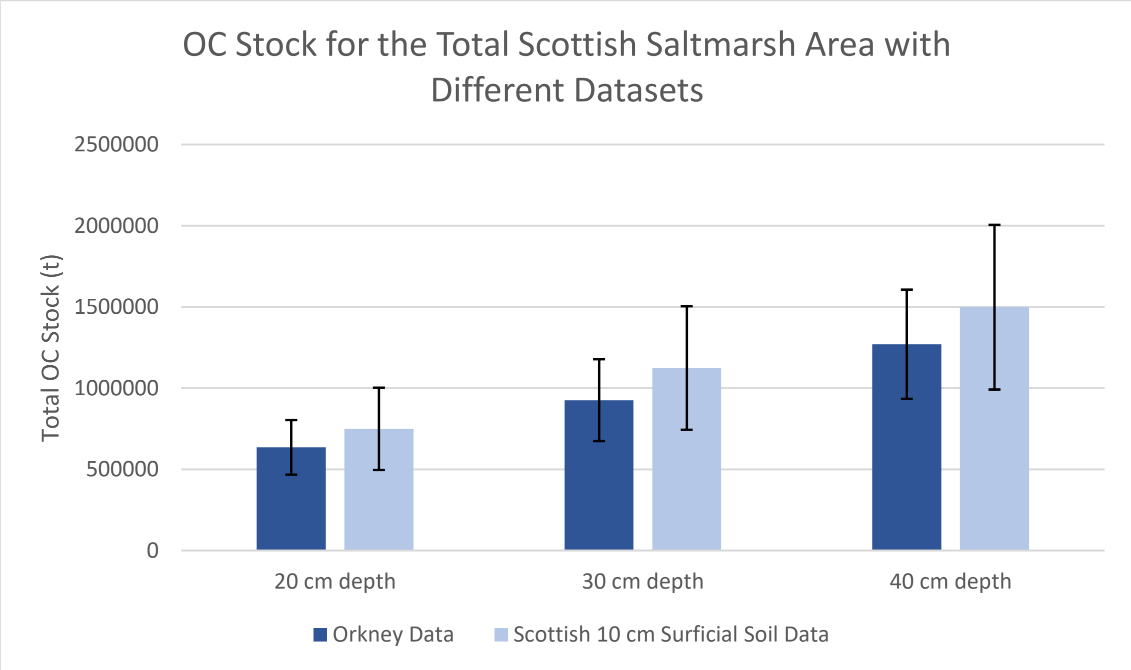


Figure 2.11: A comparison of the OC stock for the total Scottish saltmarsh obtained from the two different datasets.

2.5 Discussion: Ecosystem Management under Uncertainty

2.5.1 Extreme Error Ranges

All datasets showed large error ranges for the saltmarshes' average depths. The average depth of the Loch Laxford saltmarsh (Barlow *et al.*, 2014), for example, was 39 cm with an error range of ± 22 cm. The error range was thus more than half of the average. However, even though this error range was so large, there were still extreme outliers that were not covered by this range; the deepest Loch Laxford core was 68.5 cm deep, while the shallowest core was only 9.5 cm deep. The same observation was also made for the Waulkmill Bay and Loch of Stenness saltmarshes as well as Barlow *et al.*'s (2014) depth records of the Kyle of Tongue saltmarsh. The only exception was displayed by the data from the Kyle of Tongue saltmarsh collected by Smeaton *et al.* (2020); in this case the error range covered the largest recorded depth but not the shallowest recorded depth. These large error ranges showcase the extreme variability within individual marshes, which suggests that even if more depth records of Scottish saltmarshes are collected, uncertainty cannot be eliminated entirely.

2.5.2 Dry Bulk Density and OC Content

Comparing the different results in the previous section emphasises the impact dry bulk density and OC content have on OC stock estimates. The OC stock estimated for the top 10 and 15 cm soil of the Loch of Stenness and Waulkmill Bay saltmarshes illustrate this point further and highlight the uncertainty these elements introduce to the upscaling process. When estimated for the full average depth of each marsh, the OC stock estimate of the Waulkmill Bay saltmarsh is significantly higher than the OC stock estimate of the Loch of Stenness saltmarsh. However, if only the top 10 or 15 cm of soil are considered (Table 2.1, Figure 2.6), the reverse is true and the OC stock estimates of the Loch of Stenness saltmarsh are higher than those of the Waulkmill Bay saltmarsh. Austin *et al.*'s (2021) approach of using only surficial soil datapoints to estimate OC stocks for Scottish saltmarshes and ranking them by their estimated OC stock thus may not provide an accurate picture of their overall significance for carbon storage; it carries the risk of distorting the significance of the individual saltmarshes. However, there are also significant advantages of using only surficial soil data. The samples can be extracted with less specialised tools than a corer; the dataset used in Austin *et al.* (2021) (Ruranska *et al.*, 2020) was collected in a citizen science project in which the public collected the samples with provided modified 50 ml syringes, recorded the location with their smartphone, and mailed the samples back to the researchers. A high areal coverage and quantity of samples could thus be achieved at low cost and use of resources. Moreover, limiting the sampling to the surficial soil reduces the chances of compaction and disturbance during the collection process (Austin *et al.*, 2021; Smeaton *et al.*, 2020).

Concerning the different depth scenarios that were calculated extrapolating the top 10 cm data from across Scotland (Ruranska *et al.*, 2020), the influence of bulk density and OC content may signify that the upscaled OC stock for the scenarios with greater depth may be an underestimation due to the expected higher dry bulk density in deeper soil layers. However, this may be mitigated by the decrease in OC content with increasing depth. This can be investigated further since data points from greater depth are available for the Waulkmill Bay saltmarsh. An OC stock for 10 cm soil depth was thus calculated for the Waulkmill Bay saltmarsh only using datapoints from 10.5 – 20.5 cm depth to enable a comparison to the top 10 cm OC stock estimate. The OC stock estimate calculated from the 10.5 - 20.5 cm deep datapoints was 260 ± 46 tonnes and thus larger than the OC stock estimate of the top 10 cm of soil (224 ± 49) which suggests that the decrease in OC content is not significant enough to offset the influence of the increased bulk density that comes with increased depth. However, this supposition cannot be generalised as it is only based on data from one saltmarsh. Yet, it does allow for the conclusion that datapoints from beyond the top 10 cm of saltmarsh soil should be included to ensure good-quality OC stock upscaling for the total area of Scottish saltmarsh.

2.5.3 Applying a National Average to a Specific Saltmarsh

In Austin *et al.* (2021), the OC stock of the top 10 and 15 cm was estimated with average data from several hundred sites across Scotland (Ruranska *et al.*, 2020) instead of only the data from the Loch of Stenness and Waulkmill Bay saltmarshes. According to the supplemental data for the report (Smeaton *et al.*, 2021), the top 10 cm soil OC stock of the Loch of Stenness and Waulkmill Bay marshes are 240 and 362 tonnes, respectively. Compared to the estimates derived from the marsh specific data, this result is fairly close to the estimation for the Loch of Stenness saltmarsh but a very high over-estimation for the Waulkmill Bay saltmarsh; the estimate surpasses the stock estimate calculated with marsh-specific data by over 100 tonnes. This result is also reflected in the OC stock estimates for the top 15 cm of the saltmarsh soil. The estimate published in the supplemental data to the Scotland report is 359 tonnes for the Loch of Stenness marsh and 543 tonnes for the Waulkmill Bay saltmarsh. Again, the estimate is slightly below the Loch of Stenness estimate and much higher than the Waulkmill Bay estimate. Applying a national average to a specific saltmarsh is thus only of limited value since there can be significant differences.

2.5.4 Comparison with other Upscaling Results

Beaumont *et al.* (2014) estimated a soil OC stock for Scottish saltmarshes of 494800 tonnes, which is a lower OC stock than calculated in any of the given scenarios. However, this estimate is difficult to compare to this study since Beaumont *et al.*'s (2014) study includes the OC stock of the underlying substratum that was excluded in this analysis since the focus of this chapter is exclusively on saltmarsh blue carbon and not on the pre-existing mud- or sandflat environment. If the substratum had been included, the overall OC stock estimates would have been even further removed from Beaumont *et al*.'s (2014) results. Moreover, the estimate is based on data from nine saltmarshes on the English west coast and Wales and was then applied to Scottish saltmarshes. These saltmarshes on the English west coast and in Wales have usually a shallow organic-rich clay layer, a sandy substratum, and are frequently grazed by livestock, although ungrazed study sites were included as well. The soil of these marshes was sampled to a depth of 15 cm and the carbon content of the sandy substratum was calculated using samples from 30 cm depth. The dry bulk density was calculated as the average of 36 samples within 15 cm depth. It is noticeable that the dry bulk density was much higher at 766 kg m^{-3} than the dry bulk density observed for the Orkney marshes and the average dry bulk density presented in Austin *et al.* (2021). Yet, since the dry bulk density was only calculated from samples with a maximum depth of 15 cm it can be expected that the bulk density of deeper layers and thus the OC stock estimate is higher overall; the OC stock was calculated for a total depth of 50 cm. Furthermore, the average OC content estimated by Beaumont *et al.* (2014) is far below the measured OC content of the Orkney saltmarshes and the average OC content reported by Austin *et al.* (2021); the average OC content calculated from the samples within the top 15 cm of soil was only 4.27% and for the sandy substratum represented by the 30 cm sampling point it was only 1.27%. However, the authors do acknowledge that the soil OC stock estimate is likely to be an underestimation since organic sediments can be deeper than the included 15 cm (Beaumont *et al*., 2014).

There are two further reports that estimate an OC soil stock for the total Scottish saltmarsh area. Burrows *et al.* (2014) report an OC stock of 8600 tonnes although it is not entirely clear how this was estimated. It is a very low value that is outstripped by the estimated potential yearly carbon sequestration rate of saltmarshes which is reported as 14200 tonnes of carbon per year. The other estimate is reported in the aforementioned report of Austin *et al.* (2021). However, the estimates of Austin *et al.* (2021) only refer to the OC stock of the top 10 and 15 cm of the saltmarsh soil. They report an OC stock of 350000 tonnes for the top 10 cm and 520000 tonnes for the top 15 cm of Scottish saltmarsh soil. The data on which this report is based was also used in the analysis of this study and compared to the data from the Orkney marshes, it produces slightly higher C stock estimates for Scottish saltmarshes.

2.5.5 Factors of Uncertainty Regarding the Carbon Stock of Scottish Saltmarshes

Uncertainty in OC stock data can stem from both characteristics of the habitat and from the methods that are used to quantify OC stocks. This section discusses three major sources of uncertainty in OC stock estimates but does not claim to present a conclusive list of all sources of uncertainty.

In addition to the depth of saltmarshes, another factor of uncertainty is the exact extent of the habitat. Saltmarshes are dynamic ecosystems, and their extent can change rapidly. Ladd *et al.* (2019) found that saltmarsh extent is closely connected to sediment supply and caution that sediment flux to the coast is in decline globally, which is likely to diminish the resilience of coastal ecosystems that depend on sediment to sea level rise. However, Ladd *et al.*'s (2019) study also found that, in contrast to the global trend, in five out of the six study regions in the UK, saltmarshes are currently accreting and increasing in extent. Yet, Beaumont *et al.* (2014) assumed that there would be a loss of saltmarsh extent at a rate of 4.5% over 20 years due to sea level rise. Both of these projections, however, need to be considered with caution. Ladd *et al.*'s (2019) study does not include Scottish saltmarshes as study sites and Beaumont *et al.* (2014) projection is based on data that is more than 20 years old, which may be outdated due to the dynamic nature of saltmarshes. To address this uncertainty, Austin *et al.* (2021) applied a ± 5% error to the extent data they used in their report. While this is an issue that needs to be highlighted, the ± 5% error to the extent data was not applied in this study. This decision was made since the study focuses on the uncertainty of saltmarsh depth and introducing further errors could have masked the effect and distracted from the uncertainty associated with this factor.

Moreover, there is still uncertainty regarding the sequestration rates of Scottish saltmarshes although there has been progress. It has been pointed out that published carbon sequestration rates are frequently based on U.S. saltmarsh studies. This is important since U.S. saltmarshes are geomorphologically different from UK saltmarshes. However, UK estimates are available from

Cannell *et al.* (1999), Chmura *et al.* (2003) and Adams *et al.* (2012) (Beaumont *et al.* 2014; Lockwood and Drakeford, 2021). Burrows *et al*. (2014) used the average carbon sequestration rate estimated by Chmura *et al.* (2003) of 210 g C/m^2/year to calculate the average sequestration potential of Scottish saltmarshes. Chmura *et al.* (2003), however, based their estimate on data of a collection of globally spread saltmarshes that only included a small number of European marshes and only one within the UK. The average sequestration rate for this one UK marsh is only 121 g C/m^2/year (Chmura *et al.*, 2003, see Table 1), which is significantly lower than Chmura *et al.*'s (2003) overall average estimate that includes North American saltmarshes. Burrows' (2014) estimate is thus very likely too high. Beaumont *et al.* (2014) who used the UK sequestration rates as foundation for a valuation, opted to use the extreme values at the two ends of the range of UK saltmarsh carbon sequestration rates for their further analysis. Considering how far away from each other these values are with 64 g C/m^2/year on the lower end and 219 g C/m^2/year on the upper end (Cannell *et al.*, 1999; Chmura *et al.*, 2003; Adams *et al.*, 2012) this seems to be the prudent approach.

In addition to the characteristics of the habitat itself, the method that is used to extract cores and samples can also contribute to uncertainty in the data. Currently there is no international standard sampling approach or method for carbon stock analysis. Hence, there is diversity in methodologies for quantifying the carbon stocks of blue carbon habitats. Fest *et al.* (2022) explain that while the methods for assessing above-ground carbon stocks are robust since they are similar to established methodologies used in upland forests, this is not the case for the methods that are used to assess the below-ground carbon stocks since the established methods vary according to disciplines. Soil scientists and paleoenvironmental scientists use different methods to collect and analyse samples. The latter only extract a limited number of cores since depositional environments with long-term rates of sedimentation are usually assumed to have lower spatial variation while soil scientists use a stratified sampling design to collect many individual cores and core samples to account for the expected spatial variation in terrestrial soils. Since terrestrial soils are usually distinctly layered, subsamples can be taken from the different soil layers of the multitude of cores and provide an estimate of soil OC for each layer (Fest *et al.*, 2022). Saltmarsh soils are very similar to terrestrial soils in this regard as well as in how their OC stock accumulates; through roots below ground and litter, soil, and sediment that is trapped in the above-ground vegetation. Generally, elements of both sampling and analysis approaches are used in blue carbon research (Fest *et al.*, 2022). The method used in this study has many similarities with terrestrial soil sampling. However, these are not the only uncertainties in data that can be caused by the sampling method. Smeaton *et al.* (2020) demonstrate that the kind of corer that is used influences the OC stock estimation. Cores can be hammered into the soil to achieve a larger sampling depth. However, these hammer cores can compact the saltmarsh soil up to 28%, reduce the thickness of the soil layers, and increase the dry bulk density. This can cause an overestimation of the soil OC stock of up to 22%. While the OC content (%) remains largely unchanged (Smeaton *et al.*, 2020), it has already been demonstrated in this chapter that the dry bulk density of the soil has a large influence in calculating saltmarsh soil OC stock.

2.5.6 Management under Uncertainty

This chapter uses an approach that is closely related to the scenario approach introduced in the section on Environmental Management Tools (section 1.6.1.3). The key difference between the analysis in this chapter and the EM approach is the type of scenario that is developed. The typical EM scenario approach develops different management or policy scenarios, while in this chapter possible data scenarios were developed that can provide the foundation for different management or policy scenarios. Exploring different management scenarios is a well-established practice (Borges *et al.*, 2021; González-García *et al.*, 2014; Ogden *et al.*, 1999) and there is not always a clear-cut difference to developing data scenarios. The IPCC emission scenarios are one case where these two approaches meld together to address the issue of uncertainty in the data as well as varying possible policy responses.

Identifying areas of high carbon storage helps managers to prioritise locations for conservation and restoration and to mitigate climate threats (Wedding *et al.*, 2021). Developing OC stock scenarios for different saltmarsh depths is beneficial in a very similar way. As previously discussed, EM should be proactive rather than reactive to issues as they emerge. There is still a high uncertainty attached to the average depth of Scottish saltmarshes- being aware of how the different depths influence the OC stock provides information on the impact should management decisions be based on erring estimates. Moreover, developing different management or policy scenarios based on these OC stock scenarios enables a quick response should new data with updated average depth estimates surface. Reducing uncertainty is an important aspect for the inclusion of blue carbon into policy. For example, as demonstrated in section 2.5.2, depending on the sampled depth, individual saltmarshes can be more or less important than other saltmarshes in their contribution to the overall carbon storage potential of saltmarshes. With the different depth scenarios, a more accurate picture of their overall significance for carbon storage and how it varies with different potential depths can be created. The saltmarshes that are most important for carbon storage under the different depth scenarios could receive stricter protection allowing for an individual and local approach to saltmarsh management, while still providing confidence that the potentially most important carbon stores are protected and contribute to climate change mitigation. The local approach to saltmarsh management is important since local communities derive benefits from saltmarshes that could be curtailed if a strict protective management approach is taken for all saltmarshes equally. For example, as established in the sections 1.3.2 and

1.4.1, communities may use saltmarshes for grazing livestock or for recreation. These uses could be included in a management approach to saltmarshes that are deemed less significant for carbon storage under all depth scenarios. Or, these moderate uses of saltmarshes could be included in a step-wise management approach that allows these uses based on the assumption of a certain saltmarsh depth and significance for carbon storage. However, based on the developed saltmarsh depth scenarios, further management approaches could already be defined that may curtail these uses if the larger depth and significance for carbon storage is confirmed. Developing these different data scenarios, thus reduces the uncertainty for environmental managers how a miscalculation of this carbon storage resource could impact climate change mitigation efforts (i.e., the amount of emissions that could be released) while simultaneously providing the foundation for proactively developing management scenarios that can be fallen back on should the current best data foundation become outdated. The impact of uncertainty on blue carbon policy is further analysed and elaborated on in Chapter 4 of this thesis.

2.6 Conclusion

In this study, two datasets from Scottish saltmarshes were analysed, and OC stock estimates compared to highlight the uncertainty and influence of depth on the total OC stock of Scottish saltmarshes. Three saltmarsh depth scenarios were developed from the available data on Scottish saltmarsh depth and applied to the data of the two datasets.

The laboratory analysis of the data from the two Orkney saltmarshes collected during fieldwork show an increase of the dry bulk density and decrease of OC content with increasing depth. This trend as well as the dry bulk density and OC values are comparable to the limited data on Scottish saltmarshes that has been published. Analysing the data further revealed three major points of interest: (i) a comparison of these two marshes shows that saltmarsh soil depth is an important factor in the calculation of the OC stock since the comparatively greater depth of the Waulkmill Bay saltmarsh is the dominant factor causing its OC stock to be higher than the OC stock of the Loch of Stenness marsh; if the depth is limited to 10 or 15 cm this relationship is reversed; (ii) using marsh-specific data to calculate the OC stock produces very different results for the Waulkmill Bay saltmarsh compared to the OC stock determined from the bigger dataset that presents an average of multiple saltmarshes across Scotland; (iii) when the data of the two marshes is combined to calculate the saltmarsh depth scenarios, it becomes apparent that the averaging method that is used can influence the OC stock estimation and introduce a level of uncertainty to the data.

The application of the Ruranska *et al.* (2020) dataset with the much higher average dry bulk density and slightly lower OC content to the depth scenarios had a considerable effect on the OC estimates, which increased significantly. However, the OC stock for the total Scottish saltmarsh area is likely to be above the estimates derived from surficial soil data if only the top 10 cm of soil are sampled. The decrease of OC content with increasing depth might mitigate this, but further analysis of the Waulkmill Bay saltmarsh data indicates that this may not be the case, although no definite conclusion can be drawn from one saltmarsh. Nevertheless, there are indications that using only the top 10 cm surficial soil data does not provide an accurate picture of the significance of deeper saltmarshes for carbon storage and climate change mitigation.

It is further important to point out that the Orkney data is quite sparse for making generalisations for the total saltmarsh area of Scotland, but it is enough to highlight the issues and uncertainties presented in this study. Moreover, it needs to be highlighted that the different depth scenarios have limitations. An important one is the linear increase or decrease of the OC stock between the scenarios. This is the case due to the use of averages; these values are likely to represent an overestimate for the upper soil layers and an underestimate for the deeper soil layers.

Overall, this approach of creating data scenarios is valuable for the management and policy concerning this habitat since it is a step towards reducing uncertainty and providing information that can help with identifying and prioritising important locations or even the entire habitat for conservation and restoration. There are many uncertainties remaining though including regarding saltmarshes’ characteristics and the methods that are used for data collection and analysis and more research is thus needed to address these.

3 THE SCOTTISH PUBLIC'S PREFERENCES AND WILLINGNESS TO PAY FOR SALTMARSH MANAGEMENT

3.1 Introduction

Saltmarshes provide many ES that cover the full range of supporting-, provisioning-, regulating-, and cultural ES (Jones *et al.*, 2011). As previously established, saltmarshes are carbon sinks since they sequester and store carbon very efficiently. However, they are also potential emitters of GHG, which depends on various factors, such as how they are managed and if they have space for migrating inland with sea-level rise. If the saltmarsh is undisturbed, carbon burial can continue for millennia (Luisetti *et al.*, 2015). However, this also means that the amount of carbon that is released if a saltmarsh is disturbed or destroyed can be very significant. Unfortunately, considerable areas of saltmarsh have already been lost. Beaumont *et al.* (2014) estimated that in 2010, Scotland had lost about 13% of its saltmarsh extent since 1945 (65 years) and projected that this trend would continue and would be exacerbated in the future with a further loss of up to 12% from the 2010 baseline by 2060 (50 years). A more recent study by Ladd *et al.* (2019) observed however that saltmarshes are currently accreting in five out of the six UK study areas; it is necessary to point out though that none of these study areas was located in Scotland and that it is thus unclear whether the result can be applied to the Scottish context. Nevertheless, since saltmarshes are a potential NbS for climate change mitigation and provide a multitude of other ES (as established in Chapter 1), they are significant for EM. Moreover, threats such as land claim are anthropogenic and could still cause saltmarsh loss even if they are in a phase of accretion. A natural shift or migration of saltmarshes inland without active management is unlikely in highly developed areas due to obstacles, such as infrastructure, which emphasises the requirement for a supporting policy (Doody, 2013; Greenberg *et al.*; 2014).

This chapter presents a valuation of the Scottish saltmarsh ES with a focus on carbon storage and addresses the question whether the benefit of the carbon storage service justifies possible management interventions and if there are trade-offs or complementarity with other ES. Economic valuation studies provide important evidence for policy in terms of possible costs and monetary benefits. Choice Experiments have been widely used to determine individuals' WTP for saltmarsh management (Birol and Cox 2007; Grilli *et al.*, 2022; Interis and Petrolia, 2016; Luisetti *et al.*, 2011; Luisetti *et al.*, 2014; Petrolia *et al.* 2014). Grilli *et al.* (2022) conducted one of the most recent studies in the UK. It focussed on the Deben Estuary saltmarsh and included saltmarsh extent, biodiversity, as well as access and distance of the marsh from respondents' homes as attributes. The payment vehicle was specified as a one-off council tax increase. Compared to the other studies named here, it is noticeable that there is distinct overlap in the attributes that were included but that there is a wide range in payment vehicles. Luisetti *et al*. (2011) and Luisetti *et* *al.* (2014)[10] included attributes that strongly resemble the selection of Grilli *et al.* (2022) and was also a UK study but with a focus on the Humber and Blackwater estuaries. Birol and Cox (2007), Petrolia *et al.* (2014), and Interis and Petrolia (2016) also included an attribute referring to biodiversity. Moreover, the first two out of these three also included extent as an attribute. Furthermore, both Interis and Petrolia (2016) and Petrolia *et al.* (2014) included attributes for flood protection and commercial fishery. Concerning the payment vehicle, a broad range is covered, including a one-off increase in water rates (Birol and Cox, 2007), a general one-time cost for households (Interis and Petrolia, 2016), and a one-time tax (Petrolia *et al.*, 2014).

None of these studies included an attribute for carbon storage in their studies. This observation of the strongly overlapping ES that are included in the many studies is confirmed by Himes-Cornell *et al.* (2018) who conducted a systematic literature review of blue carbon ecosystem valuation studies of the previous 10 years and critique that valuations are rarely conducted for more than a few selected services, which is also reflected by several of the identified UK case studies (Beaumont *et al.*, 2014; Lockwood and Drakeford, 2021), and that predominantly provisioning services are valued (based on the TEEB classification). The discrete choice experiment (DCE) of this study included four benefits provided by ES spread across three ecosystem service categories: (i) regulating services in the form of climate regulation (carbon storage) and moderation of extreme events (flood protection); (ii) supporting services (biodiversity); and (iii) cultural services (opportunities for recreation and tourism). While the focus is on the carbon storage ES, there is potential to conduct further analysis for the DCE data to cover the other included ES in more detail. Furthermore, Himes-Cornell *et al.* (2018) stress the importance of additional valuation studies based on new primary data since they found that many studies are based on older possibly outdated data. Original primary data was collected for this study, which therefore also contributes to the available database of studies based on new data.

There are a few studies that investigate the value of the carbon sequestration and storage ES of saltmarshes including the previously mentioned studies by Luisetti *et al.* (2011), Luisetti *et al.* (2014), Beaumont *et al.* (2014), and Lockwood and Drakeford (2021) (Beaumont *et al.*, 2014; Lockwood and Drakeford, 2021; Luisetti *et al.*, 2011; Luisetti *et al.*, 2014; Watson *et al.*, 2020). However, the carbon valuation is not part of the choice experiment but was conducted separately with the damage cost avoided method (Luisetti *et al.*, 2011) and the DECC's prices for non-traded carbon (Luisetti *et al.*, 2014). Beaumont *et al.* (2014), Lockwood and Drakeford (2021), as well as Watson *et al.* (2020) also use the DECC's non-traded carbon values, which are set based on marginal abatement costs. Lockwood and Drakeford (2021) additionally included the SCC in the

[10] The publications by Luisetti *et al.* (2011) and Luisetti *et al.* (2014) appear to be based on the same underlying data.

valuation exercise for comparison. Himes-Cornell *et al.*'s (2018) assessment that the choice experiment approach is rarely used in the blue carbon context is confirmed by these findings. However, as other studies demonstrate, the choice experiment approach has been used to determine the value of carbon (Diederich and Goeschl, 2011; Kotchen *et al.*, 2013; Ščasný *et al.*, 2017; Shoyama *et al.*, 2013) but in the context of different habitats or directly in the context of mitigation policies, such as Ščasný *et al.*'s (2017) study on the EU's future climate mitigation policies. This chapter contributes to (i) the scarce literature on saltmarsh valuations and especially Scottish and UK case studies that value the carbon storage service; (ii) strengthening the 'rarely used' valuation methods in the blue carbon habitat valuation literature identified by Himes-Cornell *et al.* (2018), and (iii) it closes the gap of lacking UK saltmarsh carbon valuation studies conducted with the choice experiment approach, which is of particular significance due to Grilli *et al.*'s (2022) findings that choice experiment valuations are well suited for the Natural Capital Approach.

One advantage of DCEs is that they permit further factors to be investigated that are important to policy and are not captured by the carbon price approach. ES valuation through DCEs can aid the design of socially optimal policies by determining from which benefits of an ecosystem the public derive the most value (Birol and Cox, 2007). Further, as they are an established method to assess the effect of information on participants' WTP for a benefit, in this chapter the hypothesis tested is that additional information on the carbon storage service of saltmarshes has a positive effect on participants' WTP. The potential effect of information on respondent's WTP is well-established in the academic literature (Czajkowski *et al.*, 2016a; Munro and Hanley, 2001) and there are studies that demonstrate that better information can influence behaviour (Jessoe and Rapson, 2014; LaRiviere *et al.*, 2014). This effect on behaviour is not always linked with an increase in respondents' WTP but can be reflected in a reduced variance of the estimate for average WTP, which suggests that respondents are able to make more informed choices with increased information (Boyle 1989). However, there are also studies that found no significant effect of the provision of information on respondent's behaviour (Boyle *et al.*, 1990). In addition to these findings in the literature, the hypothesis is derived from the combination of two factors: (i) there is currently a favourable political climate in Scotland concerning NbS for climate change mitigation and a climate emergency has been declared, and (ii) the carbon storage ES of saltmarshes is not yet widely known in the Scottish public. Testing for this effect for the carbon storage service of saltmarshes will deliver important information for policymakers whether information campaigns could increase the public's acceptance and WTP for blue carbon climate change mitigation policies. This chapter is, therefore, a valuable and timely contribution to the current blue carbon policy development process in Scotland.

3.2 Aim of Chapter

In the overall context of this thesis, this chapter investigates the support and WTP of the Scottish public for saltmarsh management. Moreover, it investigates the relative importance of carbon storage within the range of ES that are provided by saltmarshes from the general public's perspective and what effect increased information has on participants' WTP. All of these insights provide important and significant information for Scottish environmental managers and policymakers since they (i) give an indication of the acceptability of policy related to saltmarsh management, (ii) whether there is support within the public to manage saltmarshes specifically for climate change mitigation, and (iii) whether increased information on saltmarsh carbon storage influences participants' support for this ES. Overall, this chapter thus constitutes an important element of the holistic saltmarsh blue carbon study since it contributes to the discussion of the overarching initial assumption that saltmarsh management is best integrated into climate change policy.

3.3 Methodology

3.3.1 Discrete Choice Experiment

As detailed above, Discrete Choice Experiments are a common survey-based stated preference method to assess preferences for and the value of ES; they overcome the absence of a market for indirect and non-use benefits by creating a hypothetical market to determine their value (Hanley and Barbier, 2009). The hypothetical market is created by presenting and describing the potential changes in ES provision through management policies. In the survey, respondents are asked to make a choice between alternative saltmarsh management scenarios that are described with several attributes that take different levels. These scenarios are grouped together on choice cards. Each choice card includes the 'business as usual' scenario, which describes what would happen if no saltmarsh management policy is introduced; the other scenarios included on the choice cards vary regarding the levels each attribute takes and how these levels are combined. Respondents are presented with a number of choice cards with different scenario combinations and are hence asked to make several choices.

3.3.2 The Survey Instrument, Experimental Design, and Operationalisation

Before drafting the survey instrument, three focus groups were organised to narrow down the possible attributes for the DCE scenarios, to provide information on the questions that needed to be included in the survey instrument, and to test first choice card drafts (Hensher *et al.,* 2015). Based on the results of these focus groups, five different attributes were included in the DCE: (i) Biodiversity, (ii) Flood Defence, (iii) Carbon Storage, (iv) Recreational Infrastructure, and (v) Price to determine respondent's WTP for a marginal increase in the benefits provided by the other attributes. The final versions of these five attributes are presented in Table 3.1 below.

The survey was structured in four different parts. The first section was the baseline assessment of knowledge and information text on saltmarshes and their ES. This was followed by information and instructions for the choice cards and the choice cards themselves. Debrief questions and questions concerning respondents' environmental ideation[11] were placed directly after the choice cards and the questionnaire finished with a demographics section.

In the survey, it was explained that Scottish saltmarshes are currently in decline (Beaumont *et al.*, 2014) and information was provided on each of the included attributes. Respondents were asked to make six consecutive and independent choices. They were asked to choose between two unlabelled management options and a 'Business as Usual' (BaU) option as illustrated below in Figure 3.1. The BaU option always took the value £0 and was unchanging while the management options showed hypothetical outcomes of managing Scottish saltmarshes. Respondents were asked to choose the option they preferred on each choice card.

Since it was the goal to test the hypothesis that increased information on the carbon attribute increased respondent's WTP, the study used a split sample approach. A treatment was administered in the form of a longer and more detailed information text on the carbon attribute to one half of the sample to test this hypothesis. Respondents were randomly allocated to the treatment and control groups.

A challenge for determining the value of ES with this method is hypothetical bias (Hanley and Barbier, 2009); because of the hypothetical nature of the survey, respondents may overstate the price they would be willing to pay. However, there are measures that can be taken to contain this issue. Champ *et al.* (2017) list several methods that have been used in the past to enhance validity, including cheap-talk and creating consequentiality, which are the methods that were used in this study to reduce hypothetical bias and to ensure internal validity. This included asking respondents to consider their budget when making their choices and creating consequentiality in the choice of the payment vehicle and by providing information on how the survey outcome will be shared with the government. An increase in income tax was chosen as payment vehicle as it is a tested vehicle with high consequentiality. It is important for consequentiality to use a payment vehicle that is considered realistic, available to respondents, matches the type of good, and is familiar and binding for respondents (Johnston *et al.*, 2017; Mariel *et al.*, 2021). Income tax fulfils these criteria. Since the carbon storage service of saltmarshes is a public good, income tax fits well as

[11] The idea of environmental ideation is based on Dunlap and Van Liere's (1978) New Environmental Paradigm (NEP scale), which is comprised of 12 Likert scale items that can measure pro-environmental orientation. Dunlap *et al.* (2000) updated and improved the scale. Five items were used from the updated scale and five further items added, three of those items were on actions that are broadly considered pro-environmental behaviour (e.g., recycling) and two were specific to the context of the survey.

payment vehicle as it applies to a high number of Scottish citizens. Moreover, it is a realistic payment vehicle, familiar to respondents, and binding for the members of the public that have an income (with the exception of people whose earnings are below the taxable income).

The information on how attributes would change under alternative policy interventions, and without intervention, was derived from the literature. The projected decrease in carbon storage of the BaU scenario was estimated from Beaumont *et al.*'s (2014) projection of saltmarsh area loss for Scotland and the associated carbon loss calculated from the average carbon storage value per hectare.[12] The biodiversity levels were calculated with information from Fuller (2010) regarding the number of breeding bird species on Scottish saltmarshes. From Fuller's current number of breeding bird species on Scottish saltmarshes the conservative value of a decrease or increase by two bird species breeding on Scottish saltmarshes was used and expressed in percentage values. Regarding the flood defence value, the study used Burrows *et al.*'s (2014) estimate that 3% of the Scottish coastline is currently protected by saltmarshes and transformed this into the total number of kilometres of the Scottish coastline protected by saltmarshes. In consultation with saltmarsh scientists and taking Beaumont *et al.*'s (2014) projected saltmarsh area loss into account, a conservative estimate of coastline that could lose or gain protection was determined. Concerning the initial price range, similar previous studies were consulted (Bauer *et al.*, 2004; Birol and Cox, 2007; Perni and Martinez-Paz, 2017; Petrolia *et al.*, 2014; Remoundou *et al.*, 2015) and an average range was set from £25 - £150.

Once completed, the DCE survey was tested face-to-face in a pilot with 22 participants. From the pilot it was possible to gain an understanding of: (i) the overall content validity of the survey instrument including the understandability of the terminology and instructions for the choice scenarios; the attribute and level depiction on the choice cards; the information text on the included saltmarsh ES and whether they provided enough information for the participants to make an informed decision; and the suitability of the payment range, (ii) the prior estimates for the utility parameters to be used in the final experimental design (priors), (iii) acceptability of the length of the survey. Overall, the pilot feedback was positive regarding the length, structure, and content of the survey, but a few adjustments were made to (i) improve the understandability of the choice cards and the information text, (ii) create a stronger connection to Scottish policy, (iii) increase precision in the wording of the questions and the information text. After the pilot study the price range was increased due to feedback from respondents and because the BaU option was only selected twice over 22x6 choices. The final attributes that were used are listed in Table 3.1.

[12] The carbon decrease in the BaU scenario is based on the best available information at the time the DCE was designed. As pointed out in previously, the study by Ladd *et al.* (2019) has since been published and indicates that saltmarshes are currently accreting in 5 out of six study regions in the UK.

Table 3.1: Attributes and their Levels.

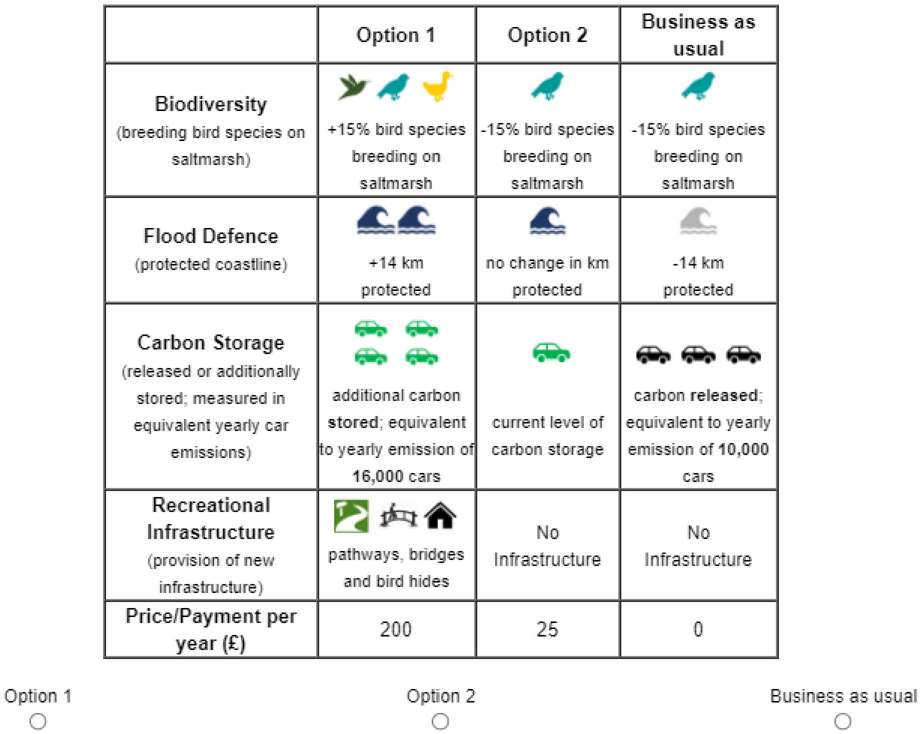
Attribute	Definition	Levels
Biodiversity	number of bird species breeding on saltmarshes	3 levels: 15% decrease (BaU), no change, 15% increase
Flood Defence	measured in the amount of coastline that would be protected by saltmarshes	3 levels: 14 km decrease (BaU), no change, 14 km increase
Carbon Storage	measured in the amount of carbon that could additionally be stored or released and was represented by the equivalent number of annual car emissions	6 levels: release of carbon equivalent to the annual emissions of 10,000 cars (BaU), no change, and additional carbon stored equivalent to the annual emissions of 4,000, 10,000, 16,000, and 20,000 cars
Recreation	measured in recreational infrastructure	3 levels: no infrastructure (BaU, since this is the case for most Scottish saltmarshes); the construction of boardwalks and bridges over creeks; the construction of boardwalks, bridges over creeks, and of bird hides;
Payment/Price	one-time increase in annual income tax for the next 10 years	6 levels: £25, £50, £100, £150, £200, £300; £0 (BaU)

The experimental design was determined with the statistical softwares SAS and NGene. The %mktruns autocall macro of the SAS software provided a list of reasonable sizes for the experimental design. The smallest design ensuring orthogonality and balance was picked, which was 36 choice situations. Of the 36 choice situations 2 were on a choice card together, which left 18 choice cards.

The choice cards were generated using a D-efficient design in NGene that minimised the D-error for the multinomial logit (MNL) model. The decision to use a D-efficient design was made since efficient designs optimise the design with the knowledge of the priors to gain the most information from each choice situation. In addition, dominant alternatives can be avoided using an efficient design (Choice Metrics, 2018). Unfortunately, it was not possible to completely avoid unrealistic attribute level combinations. For example, one factor that influences the amount of carbon stored in saltmarshes is saltmarsh extent, which is also important regarding the provision of the biodiversity and flood protection ES. A scenario where the carbon storage service increases at the highest level and the flood defence service decreases is thus not entirely realistic. However, there are also factors that influence certain ES independently from other ES. For example, the amount of carbon stored in saltmarshes can also be influenced by allochthonous carbon influx through external sediment supply. The attributes in the choice experiment are thus not completely related and opposite developments in the attribute levels are possible. Moreover, these trade-offs between the attributes are essential for the choice experiment method. Otherwise, it would not be possible to determine informative values for the ES (Mariel *et al.*, 2021). Compared to implausible alternatives, lacking trade-offs between the attributes would have been more detrimental to the experimental design and the decision was thus made to accept that there are some alternatives that are not entirely realistic. Furthermore, to ensure the most robust design possible was used, several designs were generated and checked for dominant alternatives. If a dominant alternative was present on a choice card, the design was discarded and a new design generated. This process was repeated until NGene produced a design without dominant alternatives. 18 choice sets were generated and divided into 3 blocks with 6 choice cards each to prevent respondent fatigue and the parameter estimates of the pilot survey were used as priors to generate the final design presented in Table 3.2. The D-error for the design was 0.0602. Participants were randomly assigned one of the 3 blocks by the survey software Qualtrics. An example choice card is shown in Figure 3.1.

Table 3.2: Final experimental design generated with NGene.

Choice Situation	Option 1					Option 2					Block	Choice Card
	Biodiversity	Flood Defence	Carbon Storage	Recreational Infrastructure	Payment	Biodiversity	Flood Defence	Carbon Storage	Recreational Infrastructure	Payment		
1	+15%	±0 km	+20,000	Pathways, Bridges, and Bird Hides	150	±0%	+14km	-10,000	Pathways and Bridges	50	2	1
2	-15%	-14km	+4,000	Pathways and Bridges	300	±0%	±0 km	+4,000	No Infrastructure	300	2	2
3	-15%	±0 km	-10,000	Pathways and Bridges	200	±0%	-14km	+10,000	No Infrastructure	300	3	1
4	±0%	+14km	±0	Pathways and Bridges	50	+15%	±0 km	+16,000	Pathways, Bridges, and Bird Hides	150	1	1
5	-15%	±0 km	+4,000	No Infrastructure	300	±0%	-14km	±0	Pathways, Bridges, and Bird Hides	300	1	2
6	±0%	+14km	±0	Pathways, Bridges, and Bird Hides	150	+15%	-14km	+16,000	No Infrastructure	100	1	3
7	-15%	-14km	+16,000	Pathways, Bridges, and Bird Hides	100	+15%	+14km	+4,000	No Infrastructure	150	3	2
8	±0%	+14km	+16,000	No Infrastructure	200	-15%	-14km	±0	Pathways, Bridges, and Bird Hides	25	2	3
9	+15%	+14km	+16,000	Pathways, Bridges, and Bird Hides	200	-15%	±0 km	±0	No Infrastructure	25	1	4
10	±0%	±0 km	+4,000	Pathways, Bridges, and Bird Hides	25	+15%	-14km	+16,000	Pathways and Bridges	200	3	3
11	-15%	-14km	+10,000	No Infrastructure	25	+15%	±0 km	+10,000	Pathways and Bridges	200	1	5
12	-15%	+14km	+10,000	No Infrastructure	150	±0%	-14km	+10,000	Pathways and Bridges	50	3	4
13	+15%	+14km	+10,000	Pathways and Bridges	300	±0%	±0 km	+4,000	Pathways, Bridges, and Bird Hides	25	3	5
14	±0%	-14km	+20,000	No Infrastructure	50	+15%	+14km	-10,000	Pathways, Bridges, and Bird Hides	100	2	4
15	+15%	±0 km	-10,000	Pathways and Bridges	25	-15%	+14km	+20,000	Pathways, Bridges, and Bird Hides	200	3	6
16	±0%	±0 km	+20,000	Pathways and Bridges	100	-15%	+14km	-10,000	No Infrastructure	100	2	5
17	+15%	-14km	-10,000	No Infrastructure	50	-15%	+14km	+20,000	Pathways and Bridges	150	1	6
18	+15%	-14km	±0	Pathways, Bridges, and Bird Hides	100	-15%	±0 km	+20,000	Pathways and Bridges	50	2	6



	Option 1	Option 2	Business as usual
Biodiversity (breeding bird species on saltmarsh)	+15% bird species breeding on saltmarsh	-15% bird species breeding on saltmarsh	-15% bird species breeding on saltmarsh
Flood Defence (protected coastline)	+14 km protected	no change in km protected	-14 km protected
Carbon Storage (released or additionally stored; measured in equivalent yearly car emissions)	additional carbon stored; equivalent to yearly emission of 16,000 cars	current level of carbon storage	carbon released; equivalent to yearly emission of 10,000 cars
Recreational Infrastructure (provision of new infrastructure)	pathways, bridges and bird hides	No Infrastructure	No Infrastructure
Price/Payment per year (£)	200	25	0

Option 1 ○ Option 2 ○ Business as usual ○

Figure 3.1: Example choice card.

3.3.3 The Study Sample

3.3.3.1 The Overall Sample

The survey was distributed through a market research company. The aim was a study sample of n=300 complete responses for each version of the survey as the company that distributed the survey advised that n=600 participants was realistically the maximum number of participants with sufficient response quality for the specified region and quotas. To achieve two subsamples of n=300 representative of the Scottish population in terms of sex and age, hard quotas were set. A further soft quota was set for household income. For the survey with shorter information on carbon storage, 313 completed responses with an even distribution across the quotas was achieved. For the survey with longer information, 307 completed responses were achieved but the male, age 18-24 quota had to be relaxed. The missing complete responses were distributed evenly across the other quotas. Overall, 527 participants started the survey with shorter information and 698 started the survey with longer information. 214 and 391 responses were respectively screened out due to the set quotas or response quality concerns. Three "red herring" questions were included to test whether respondents read the provided information text; respondents were screened out if they replied incorrectly to two out of the three questions. Moreover, respondents were screened out due to speed of completion concerns. The descriptive statistics of the respondents presented in Table 3.3 demonstrate the representativeness of the sample in terms of age and sex and provides an overview of respondents' other characteristics.

Table 3.3: Descriptive statistics.

	Treated Sample	Control Group		Treated Sample	Control Group
n	307	313	**n**	307	313
Age (%)			**Children (%)**		
18-24	8.14	10.54	None	36.16	42.17
25-34	17.26	16.61	1	21.50	19.17
35-44	15.64	15.34	2	25.73	23.96
45-54	18.24	18.85	3	11.40	9.90
55-64	17.59	16.29	4 or more	5.21	4.47
65 and over	23.13	22.36	Prefer not to say	-	0.32
Sex (%)			**Taxpayer (%)**		
Female	53.75	52.08	Yes	70.36	72.84
Male	46.25	47.92	No	29.64	27.16
Education (%)			**Annual household Income (%)**		
High School	28.99	30.67	£ 0-12,500	16.29	13.42
College	22.80	21.09	£ 12,501-20,000	12.70	16.61
Bachelor	25.41	26.84	£ 20,001-30,000	18.89	18.21
Master	10.75	9.90	£ 30,001-40,000	18.24	21.41
PhD or higher	3.58	3.83	£ 40,001-50,000	11.07	12.46
Technical	7.17	4.15	£ over 50,000	17.59	14.06
Prefer not to say	1.30	3.51	Prefer not to say	5.21	3.83
Marital Status (%)			**Employment (%)**		
Single	30.29	32.91	Full-time	31.60	34.82
Married or Civil Partnership	48.86	46.96	Part-time	9.45	15.34
Divorced	10.10	9.58	Self employed	7.49	7.67
Widowed	6.84	4.15	Student	5.86	6.07
Other	0.65	-	Retired	28.01	20.45
Prefer not to say	3.26	6.39	Homemaker	7.49	6.39
			Not Employed	5.21	3.51
Election Participation (%)			Other	3.26	4.47
Yes	88.27	90.42	Prefer not to say	1.63	1.28
No	10.75	8.31			
Prefer not to say	0.98	1.28			

The survey being distributed online may have led to some selection bias since it excluded potential respondents that did not have internet access. The Scottish Government reports, in its 2018 Household Survey, that 87% of the Scottish population had internet access in 2018 (Scottish Government, 2020c). They also report that while gaps narrowed in recent years, older adults and households with lower incomes were still less likely to have internet access. Accordingly, there was the potential that the sample could be biased towards younger adults and against adults from lower income households and from deprived areas. The first effect was steered against by including quotas for age to still reach a representative sample. However, it was not possible to adjust for the second effect. Moreover, respondents were selected from an opt-in panel, which may limit generalisability and thereby external validity. However, these issues were weighted against the ability to procure such a large, stratified sample. Balancing tests were conducted (Appendix B, Table B.1) to check whether the random allocation to the treated and control groups worked. The tests were conducted as Chi-square tests of the variables that may have influenced participants' decisions, between the two groups. There is no indication that the randomisation of the untreated and treated samples was not successful.

3.3.3.2 Participants who Chose Business as Usual – Protest Responses and Genuine Zeros

A follow-up question on the reason for choosing the BaU scenario was used to distinguish protest responses form genuine zero responses. Protest responses are those that systematically choose the BaU option to reject or protest against some aspect of the constructed market scenario. The follow-up question included statements that can indicate a genuine zero response and statements that indicate a protest response. According to Mariel *et al.* (2021), protest responses usually choose statements that indicate that others such as the government or other organisations should bear the cost of the management. Table 3.4 presents the available statements selected by the respondents who always chose the BaU scenario. Overall, 30 out of the 620 respondents (4.8%) always selected the BaU scenario, but since respondents could choose multiple statements explaining their choice the total number of responses in Table 3.4 is higher. The statements that indicate a genuine zero response are shaded in grey.

Table 3.4: Selected responses of the respondents who always chose the BaU scenario when asked for the reason why.

Why did you choose the 'Business as usual' option?		
1	Taxes and fees are already too high, so there should not be an additional financial burden	19
2	I cannot afford to pay any more in taxes	12
3	It is not a priority/other things are more important	10
4	Government/local authority should pay	8
5	Other	5
6	The positive change of the saltmarsh condition is not significant enough to pay for it	4
7	I am a non-taxpayer/I am not working	4
8	I am not concerned about the condition of saltmarshes	4
9	I think it is better to ask experts about how saltmarshes should be managed	3
Total		**69**

Of these thirty respondents, those that selected at least one of the statements that indicate a genuine zero response were removed from the protest response analysis since they fit the criterium for genuine zero responses. This does not mean that there was no overlap with protest responses, but these instances cannot be considered as full protest responses. From these removed responses only five consistently selected only the statements that indicate a genuine zero response. Once these responses with valid zero statements were removed, six protest respondents remained (1%); the statements these six respondents chose are presented in Table 3.5.

Table 3.5: Full protest responses.

Why did you choose the 'Business as usual' option?		
1	Taxes and fees are already too high, so there should not be an additional financial burden	3
4	Government/local authority should pay	3
5	Other	1
8	I am not concerned about the condition of saltmarshes	-
9	I think it is better to ask experts about how saltmarshes should be managed	1
Total		**8**

The respondent who selected 'Other' specified that:

> While my taxes are used to prop wars, support neonazis (Croatia, Ukraine, etc.), islamic fundamentalists (Kosovo, Bosnia, Libya, Syria, etc.) I will seek every option to avoid paying a penny. Also, as hypocritical St Andrews academics should understand, I have zero interest in the survival of our species (Anonymous respondent),

which can be counted as a protest response.

Mariel *et al.* (2021) point out that whether or not to exclude protest responses is still an open question. In this study, the protest responses were not excluded from the analysis due to their low number and since it has been argued that including protest answers provides a more conservative estimate of WTP. Thus, it increases confidence that the WTP is not an overestimation.

3.3.4 Analytical Framework and Preference Analysis

DCE's are based on Lancaster's (1966) *characteristics theory of value* and the *random utility theory*, which derives from Luce (1959) and McFadden (1973). Applying this framework to ES, means that the utility provided by a change in the management of ES to an individual is the linear sum of the utility provided by each of their characteristics or attributes. The Random Utility Model assumes that utility can be broken down into an observable part *V*, which is the sum of the utility provided by each of the *k* attributes, and a random unobservable part, or error part ε. It can be described as the following equation (1) (Bateman *et al.,* 2002):

$$U = U(X_1 \dots X_k) = V(\boldsymbol{X}) + \varepsilon \qquad (1)$$

where *X* ($X_1,\dots,X_k$) is the vector of *k* attributes describing a management scenario.

Applied to this study, the utility function is:

$$U_{nj} = (\beta_{0,n} * asc_{nj} + \beta_{1,n} * \ biodiversity_{nj} + \ \beta_{2,n} * flood_{nj} + \ \beta_{3,n} * carbon_{nj} + \beta_{4,n} * recreation_{nj} + \beta_{5,n} * payment_{nj}) + \varepsilon_{nj} \qquad (2)$$

Where ‘biodiversity’, ‘flood’, ‘carbon’, ‘recreation’ and ‘payment’ stand for the attribute levels included in the DCE presented in alternative j to respondent *n* and β_n represents the utility parameter of respondent *n* for the respective attribute.

Since this study tested the effect of information on preferences and WTP, a binary treatment variable that was either ‘short’ or ‘long’ was created and included in interaction with the carbon attribute variable. The results were then tested for the difference in the two carbon coefficients ($\beta_{3.1}$ and $\beta_{3.2}$ in equation (3) below) that were estimated in interaction with the treatment variable to assess the effect of additional information on preferences and WTP for carbon. The specification of the utility function that assesses the effect of treatment (*T*) is thus:

$$U_{nj} = (\beta_{0,n} * asc_{nj} + \beta_{1,n} * \ biodiversity_{nj} + \ \beta_{2,n} * flood_{nj} + \ \beta_{3.1,n} * carbon_{nj} * T_{short} + \beta_{3.2,n} * carbon_{nj} * T_{long} + \ \beta_{4,n} * recreation_{nj} + \beta_{5,n} * payment_{nj}) + \varepsilon_{nj} \qquad (3)$$

In the choice set, which represents all available options, individuals are asked to choose the option that maximises their utility from the available alternatives. Since it is impossible to observe all factors that influence individuals’ choices, such as intrinsic reasons, it is necessary to introduce the error and, since the error is not observable, assumptions need to be made about its distribution. The probability that any respondent *n* prefers option *j* over any other option *g* in the choice set *S* can be expressed in an equation that describes that the utility associated with option *j* is higher than the utility associated with any other option *g* (4) (Bateman *et al.*, 2002):

$$P\left[\left(V_{nj} + e_{nj}\right) > \left(V_{ng} + e_{ng}\right)\right] = P\left[\left(V_{nj} - V_{ng}\right) > \left(e_{ng} - e_{nj}\right)\right], \forall g \neq j, \ g \in S \qquad (4)$$

A typical assumption about the error terms is that they are independently and identically Gumbel distributed (IID). Under this assumption, and also the assumption that respondents will choose the option that provides them with the highest utility, the probability that alternative *j* is chosen over any other alternative *g* is expressed in the conditional logit model (5), which, together with its variants such as the mixed logit model, is the core model for choice experiment data analysis (Bateman *et al*., 2002; Johnston *et al.*, 2017). Since the conditional logit model is the simplest initial model to estimate preference coefficients for each of the attributes of the saltmarsh management scenarios, it was the first model that was fit in the data analysis. The model is described as:

$$P\left(U_{nj} > U_{ng}, \forall g \neq j, \ g \in S\right) = \frac{\exp\left(\mu V_{nj}\right)}{\sum_{g=1}^{J} \exp\left(\mu V_{ng}\right)} \qquad (5)$$

where μ is a scale parameter which cannot be separately identified in a single dataset and is hence implicit. Other authors, such as Louviere *et al.* (2000) omit μ in their equation of the conditional logit or MNL model. The probability that alternative *j* is chosen is thus expressed as in (6):

$$P_{nj} = \frac{\exp(V_{nj})}{\sum_{g=1}^{J} \exp(V_{ng})} \quad (6)$$

V_{nj} (or V_{ng}) are the linear sums of the utility provided by each of the attributes ($\boldsymbol{X}$) which determine the utility of the *j*th (or *g*th) alternative and as presented by Louviere *et al.* (2000) can be written as in (7):

$$V_{nj} = \sum_{k=1}^{K} \beta_{jk} \boldsymbol{X}_{njk} \quad (7)$$

where β_{jk} is a utility parameter associated with attribute *k* of alternative *j* that represents the weight of each attribute in utility and which is assumed to be constant across individuals. $\boldsymbol{X}_{njk}$ is the level of attribute *k* associated with alternative *j* presented to respondent *n*.

Applied to the conditional logit equation, the probability that alternative *j* is chosen by individual *n* can thus be expressed as in (8):

$$P_{nj} = \frac{\exp(\alpha_j + \boldsymbol{\beta}_j \boldsymbol{X}_{nj})}{\sum_{g=1}^{J} \exp(\alpha_g + \boldsymbol{\beta}_g \boldsymbol{X}_{ng})} \quad (8)$$

$\boldsymbol{X}_{nj}$ (or $\boldsymbol{X}_{ng}$) is a vector of *k* attributes describing alternative *j* (or *g*) presented to respondent *n* and $\boldsymbol{\beta}_j$ (or $\boldsymbol{\beta}_g$) is the vector of *k* utility parameters associated with the *k* attributes of alternative *j* (or *g*) which represent the weight of each attribute in utility and is assumed to be constant across individuals. α_j (or α_g) is an alternative-specific constant associated with alternative *j* (or *g*).

However, participants are expected to have heterogeneous preferences for at least some if not all attributes. Regarding the recreation attribute, for example, bird hides may be less attractive for the majority of the population than for birdwatchers or, for example, the flood defence attribute may be more attractive to participants who live in flooding areas than to participants who do not live at the coast. The mixed logit model may thus be the more appropriate choice as it allows for heterogeneous preferences across participants (Mariel *et al.*, 2021). It can be fit in preference space and WTP space. In preference space, the distribution of coefficients is specified in the utility function and WTP can be calculated as the ratio of the attribute preference coefficient to the monetary coefficient; in WTP space, the distribution of WTP is directly specified in the utility function and the estimated parameters thus represent the WTP distribution parameters rather than the preference coefficients (Train and Weeks, 2005). This is achieved by changing the utility function, which is specified in preference space as presented in (9):

$$U_{nj} = \boldsymbol{\beta}'_n \boldsymbol{X}_{nj} + \varepsilon_{nj} \quad (9)$$

where $\boldsymbol{X}_{nj}$ is a vector of k attributes describing alternative j presented to respondent n and $\boldsymbol{\beta}'_n$ is individual n's vector of k preference parameters. In WTP space, the utility function is adjusted so that the cost coefficient multiplies the rest of the utility function as presented in (10):

$$U_{nj} = \beta_n^{\prime m}\left(\boldsymbol{X}_{nj}^{m} + \boldsymbol{\beta}_n^{\prime -m}\boldsymbol{X}_{nj}^{-m}\right) + \varepsilon_{nj} \tag{10}$$

Where $\boldsymbol{X}_{nj}^{m}$ is the monetary and $\boldsymbol{X}_{nj}^{-m}$ a vector of all other attributes, $\beta_n^{\prime m}$ is the parameter for the monetary attribute, which was specified to be lognormally distributed, and $\boldsymbol{\beta}_n^{\prime -m}$ is the vector of parameters for all other attributes, which were normally distributed. This approach has been found to produce more realistic WTP measures (Hole and Kolstad, 2012; Train and Weeks, 2005) than calculating the WTP as the ratio of the attribute preference coefficient to the monetary coefficient. The probability function for the mixed logit model in both preference and WTP space is specified as described by Louviere *et al.* (2000) and presented in (11):

$$P(j|\mu_n) = \frac{\exp\left(\alpha_{nj} + \boldsymbol{\beta}_{nj}\boldsymbol{X}_{nj}\right)}{\sum_{g=1}^{J}\exp\left(\alpha_{ng} + \boldsymbol{\beta}_{ng}\boldsymbol{X}_{ng}\right)} \tag{11}$$

$\boldsymbol{X}_{nj}$ (or $\boldsymbol{X}_{ng}$) is a vector of k attributes describing alternative j (or g) presented to respondent n and $\boldsymbol{\beta}_{nj}$ (or $\boldsymbol{\beta}_{ng}$) is a parameter vector of k utility parameters associated with the k attributes of alternative j (or g) that is randomly distributed across individuals and represents the weight of each attribute in utility; μ_n as a component of $\boldsymbol{\beta}_{nj}$ (or $\boldsymbol{\beta}_{ng}$) is the individual-specific random disturbance of unobserved heterogeneity. α_j (or α_g) is an alternative-specific constant associated with alternative j (or g).

3.3.5 Latent Class Analysis

Latent Class Analysis (LCA) belongs to the family of latent variable techniques called finite mixture models and is a person-centred data analysis method. It analyses patterns in data to group participants together, thereby identifying latent subpopulations called latent classes. The underlying assumption of LCA is that these unobserved latent subpopulations can explain score patterns across survey questions. The latent classes are identified through participants' responses to observed categorical indicator variables. It is important to note that LCA only determines the probabilities of class membership and does not provide definite assignments to classes. LCA was conducted in this study since identifying classes can nevertheless provide valuable information for decision- and policymakers. Once classes and their characteristics are identified, this information can be used to improve communication. If a certain group is a target audience, the communication can be adapted to be more likely to reach this certain group. It can thus make communication more efficient.

The process of deciding how many classes are included in an LCA is called 'class enumeration'. During this process, the LCA model is fitted several times with a varying number of classes until the best model fit is achieved. Usually, the starting point is the one-class model and then one additional class is added at a time until the model quality starts to deteriorate. The one-class model hereby serves as a comparative baseline. (Nylund-Gibson and Choi, 2018; Weller *et al.* 2020). However, in Stata, the statistical analysis software, only models with two classes or more can be fitted with the LCA command since the conditional logit model represents the model with only one class. Fit indices help decide which model has the best fit for the data. Oberski (2016) states that the AIC (Akaike information criterion) and BIC (Bayesian information criterion) are the most commonly used, to which Weller *et al.* (2020, 6) add that, while several fit statistics should be used and reported, the BIC "may be the most reliable fit statistic and should routinely be reported". Other common fit statistics are the SABIC (Sample-size adjusted Bayesian information criterion); CAIC (Consistent Akaike information criterion); AWE (Approximate weight of evidence criterion); VLMR-LRT (Vuong-Lo-Mendell-Rubin adjusted likelihood ratio test); and BLRT (bootstrapped likelihood ratio test). The BIC, AIC, SABIC, CAIC, and AWE are approximate fit indices and lower values indicate better fit. In contrast, the VLMR-LRT and BLRT are likelihood-based tests, which provide p-values that indicate whether the model fit improvement is statistically significant when a further class is added. A non-significant p-value ($p > 0.05$) therefore indicates that of the compared models, the one with one class less should be used. The different fit indices can support more than one model. In this case, the researcher needs to check which model is a better fit for the data according to model stability (e.g., relative sizes of the classes; small sample sizes can make it difficult to recover small classes) (Nylund-Gibson and Choi, 2018).

LCA can be conducted for choice modelling as a latent class conditional logit (LCL) model and is implemented in Stata with the lclogit2 and lclogitml2 commands introduced by Yoo (2020). The model is based on the conditional logit model which assumes that the error terms representing individual preference are IID. However, instead of assuming IID, the LCL incorporates a "discrete representation of unobserved preference heterogeneity across [respondents]" (Yoo 2020, 407), which allows the respondents to be allocated to C distinct classes where each class c "makes choices consistent with its own clogit [conditional logit] model with utility coefficient vector $\boldsymbol{\beta}_c$" (Yoo 2020, 407). The probability that respondent n belongs to class c is represented by

$$\pi_{nc}(\boldsymbol{\Theta}) = \frac{\exp(\boldsymbol{z}_n \boldsymbol{\theta}_c)}{1+\sum_{l=1}^{C-1} \exp(\boldsymbol{z}_n \boldsymbol{\theta}_l)} \tag{12}$$

where z_n is a vector of individual n's characteristics; θ_c is a vector of membership coefficients for class c, with θ_C set to 0 for the reference class; and $\Theta = (\theta_1, \theta_2, \ldots, \theta_{C-1})$ represents the $C - 1$ membership coefficient vectors. (Yoo 2020, 407-408).

Respondent *n*'s choices' joint likelihood in the LCL model is represented by

$$L_n(\boldsymbol{B}, \boldsymbol{\Theta}) = \sum_{c=1}^{C} \pi_{nc}(\boldsymbol{\Theta}) P_n(\beta_c) \tag{13}$$

"where $\boldsymbol{B} = (\boldsymbol{\beta}_1, \boldsymbol{\beta}_2, \ldots, \boldsymbol{\beta}_C)$ denotes a collection of the C utility coefficient vectors and each $P_n(\boldsymbol{\beta}_c)$ is obtained by evaluating (8) at $\boldsymbol{\beta} = \boldsymbol{\beta}_c$" (Yoo 2020, 408).

The parameters of interest, $\boldsymbol{B}$ and $\boldsymbol{\Theta}$, are estimated in preference space using maximum likelihood estimation.

3.3.6 Cluster Analysis

The DCE dataset contained a wide range of independent variables that could explain class membership. However, using them all in the LCA caused convergence issues and their number thus had to be reduced. Cluster analysis is a method that can be used to organise or provide a summarisation of a large dataset in a small number of groups. The group descriptors, if well-picked, can provide an illustration of the patterns, similarities, or dissimilarities in the data (Everitt *et al.*, 2011; Hennig and Meila, 2016). Cluster Analysis was hence employed in this study to reduce the independent variables of the dataset for the LCA.

3.3.6.1 The Independent Variables for the Latent Class Analysis

For this purpose, the participant-constant independent variables were divided into three groups: (i) demographic variables; (ii) variables that provide information regarding participants' familiarity and existing knowledge about saltmarshes; and (iii) variables that provide information regarding participants' personal experiences and attitudes. An overview of the variables is provided in Table 3.6. The aim was to conduct separate cluster analysis on these three variable-subsets.

Table 3.6: The three independent variable groups.

Demographic variables	
Pay income tax	Number of children
Sex	Income
Age	Employment
Education	Participated in the last election
Marital Status	
Variables that provide information regarding participants' familiarity and existing knowledge about saltmarshes	
Have heard about saltmarshes	Know about saltmarsh carbon storage
Have been to a saltmarsh	Know about saltmarsh flood protection
Know about saltmarsh biodiversity	Know about saltmarsh recreational value
Variables that provide information regarding participants' personal experiences and attitudes	
NEP scale	Buy organic products
Support a political focus on the environment	Risk scale
Donate to conservation associations	Discount scale
Practice recycling	Have been affected by flooding
Should consider moving infrastructure so the coastline can naturally adapt to sea-level rise	

3.3.6.2 Hierarchical Cluster Analysis

In a first step, hierarchical cluster analysis that partitions data into clusters in a series of steps was used with the Caliński-Harabasz (CH) and Duda-Hart stopping rules to determine a range of possible cluster numbers (Everitt *et al.*, 2011). The results of possible cluster numbers were then used to specify the cluster numbers for k-means clusterings. The complete linkage, average linkage, and Ward's linkage methods, which are all standard agglomerative hierarchical clustering methods, were determined to be the most suitable. Everitt *et al.* (2011) report on several studies which found that Ward's linkage performs well with clusters of similar size, while complete linkage is preferable if the opposite is the case. Another favoured method is average linkage. These methods were used since the hierarchical cluster analysis was a first exploration of the data and possible cluster numbers.

There is a wide range of proximity measures that can be used in cluster analysis to define the distance between two objects or individuals; the most suitable measure should be selected according to the nature and scale of the data. Stata offers a wide range of proximity measures for both continuous and binary data such as Gower's dissimilarity coefficient for mixed data (Stata 2019), which was used in this study for the third groups of variables (i.e. variables that provide information regarding participants' personal experiences and attitudes). Categorical data can be transformed into binary data; each level of the categorical variable is then regarded as a binary variable. Since most of the variables included in the variable groups were categorical variables, they needed to be transformed into binary variables. However, Everitt *et al.* (2011) caution that researchers need to be aware of a shortcoming of this method because it causes a large number of negative matches. For example, if a categorical variable with five levels is transformed into five binary variables, four of these binary variables will be negative as they represent the four levels of the categorical variable that were not chosen. In the cluster analysis these negative matches can then outnumber the positive number of matches, which is inconvenient since clusters are supposed to group participants according to similarities. The Dice similarity measure that strongly weights positive matches was therefore selected to mitigate this effect for the first variable group (i.e. the demographic variables); it gives a higher weight to agreements, which reduces the influence of the negative matches (Everitt *et al.*, 2011; Stata, 2019).

However, for all three variable groups (i.e., (i) demographic variables, (ii) variables regarding participants' familiarity and existing knowledge about saltmarshes, and (iii) variables that provide information regarding participants' personal experiences and attitudes), the results of the hierarchical cluster analysis were quite inconclusive as the different methods and the two stopping rules available in Stata determined varying 'best' cluster numbers. To illustrate, for the first group, which included the demographics variables, the complete linkage method recommended nine clusters with both stopping rules; the average linkage method recommended two clusters with both stopping rules; and the ward's linkage recommended two clusters with the CH stopping rule and no clear cluster number was apparent with the Duda-Hart stopping rule.

3.3.6.3 K-means Clustering

For each of the three variable groups several k-means cluster analyses with different cluster numbers were thus covered to investigate all cluster numbers that were deemed appropriate for reducing the variables for a subsequent LCA and to double check the possibilities that were suggested by the hierarchical cluster analysis outcomes. K-means clustering is the most popular non-hierarchical clustering method, which use algorithms to maximise homogeneity within clusters. The objective of this method is to divide the observations into *k* clusters "such that those within each cluster are the closest to each other if compared to any other that belongs to a different cluster" (Fávero and Belfiore 2019, 339). The algorithm seeks to optimise these criteria; k-means clustering can therefore be allotted to the optimisation methods (Everitt *et al.* 2011). As a starting point, k cluster centres have to be defined to which the observations are then allocated to form initial clusters according to their distance to these centres. It is hence required that the researcher specifies the number of clusters.

The maximum cluster number to consider as input for the LCA was four clusters. The k-means clusterings with the different cluster numbers were then compared with the CH index, which was also used as a stopping rule in the hierarchical cluster analysis. A larger index value indicated a more distinct clustering (Stata, 2019). For the comparison, the clusterings were run several times with different cluster starting options (krandom, firstk, lastk, random, prandom, everykth, segments) (Stata, 2019). For all variable groups, the cluster solution with two clusters scored the highest on the CH index overall, so the cluster analysis was continued with two clusters for each variable group. Since clustering results can vary according to the starting option that is chosen, the cluster analysis was conducted twenty-five times with each of the seven starting options named above; the cluster sizes as well as the CH index value were recorded for comparison since the CH index can also be used to compare clusterings with the same number of classes k. Again, a higher CH index indicated a better clustering (Halkidi *et al*., 2016).

For the demographics variables, the cluster solution with the highest CH index value was also the solution that was produced most frequently and consistently with three different starting options: (i) firstk, (ii) everykth, and (iii) segments. These results support the assumption that this clustering solution is the 'best' solution for these variables. Furthermore, the consistency with three starting options also makes the clustering solution reliably reproducible as foundation for further data analysis, such as the LCA.

It was not possible to stabilise the results for the second variable group that contained the variables regarding participants' familiarity and existing knowledge about saltmarshes. The series of analyses with different starting options produced more varying results, which were neither stable with a specific starting option nor when a seed was specified for the krandom starting option. However, as Everitt *et al.* (2011) note, "for k-means and other hill-climbing techniques, different seeds for the initial clusters should not affect the cluster solutions." Since even specifying seeds could not stabilise the results, the confidence is low that these are acceptable clustering results as the robustness of the results is severely lacking. It also raises the issue of reproducibility; unstable results cannot be used as a foundation for further analysis since different versions of the cluster outcome variable may feed into the subsequent LCA if the results in the preceding cluster analysis change. The clustering of these variables was hence abandoned. However, this should not be considered a problem. In some cases, data is unsuitable for grouping and therefore no grouping is justified (Everitt *et al.* 2011; Hennig and Meila 2016).

For the personal experience and attitude variables, the clustering results were not stable, but they improved for a subset when the risk and discount scales as well as the variable about previous flooding experience were excluded. The process of running several k-means cluster analyses with different cluster numbers to determine the best number of clusters was repeated for this subset and the two-cluster solution still scored the highest on the CH index. Again, the cluster analysis was conducted twenty-five times with each of the seven starting options. Three starting options (prandom, everykth, and segments) produced consistently the same result. This cluster outcome did not have the highest score on the CH index but was just slightly below it. Due to this closeness to the highest score and its stability and consequent reproducibility, the result was deemed appropriate to use for the following LCA. These variables that were excluded from this clustering were included directly into the LCA.

3.4 Results

3.4.1 The Conditional Logit Model

The conditional logit model was fit to the data to determine participants' preferences regarding the management of the included ES. As expected, and presented in Table 3.7, it shows a negative coefficient for the payment attribute; all other attribute coefficients are positive. Moreover, all coefficients are significant at $p < 0.01$, which indicates that all attributes influenced respondents' choices.

Table 3.7: Conditional logit model results.

Number of observations		10,536
Number of parameters		9
Log likelihood		-3304.7856
Choice	**Mean**	**Standard Error**
Maintaining current biodiversity level	0.433***	0.054
Increasing biodiversity level	0.605***	0.063
Maintaining current flood defence level	0.240***	0.053
Increasing flood defence level	0.616***	0.064
Marginal increase in carbon storage	0.031***	0.002
Providing bridges and boardwalks	0.341***	0.054
Providing bridges, boardwalks, and bird hides	0.180***	0.056
asc (alternative specific constant)	0.258***	0.089
Increase in income tax for 10 years	-0.003***	0.000

***, ** and * indicate 1,5 and 10% significance levels respectively

For both the biodiversity and flood defence attributes, preferences are higher for an increase in the attributes than they are for only maintaining them. The recreation attribute, however, shows a higher preference for the simpler improvements in infrastructure than the more extensive improvements that include bird hides in addition to the bridges and boardwalks to make the saltmarsh more accessible. The carbon attribute shows a preference for a marginal increase in carbon storage and the positive asc indicates a preference for change over the status quo. The asc (alternative specific constant) shows, independent of the results for the other attributes, the utility individuals get simply from either leaving or staying in the status quo (Hanley and Barbier, 2009).

Hausman tests were conducted to test whether the Independence of Irrelevant Alternatives (IIA) property[13] was violated. The null hypothesis that the IIA property is not violated was rejected in two out of the three cases, hence the model results are not robust in two out of three cases when the irrelevant alternatives are dropped. The null hypothesis was not rejected when alternative three, the BaU option, was dropped and the results thus stayed robust. However, this robustness may be attributed to the low number of times the *status quo* option was selected; only 461-times compared to the 1,403- and 1,648-times management options 1 and 2 were chosen, respectively. The perceived robustness may thus stem from the fact that the sample stayed largely the same as so few irrelevant alternatives were dropped. This could indicate that participants demonstrated a preference for change over the BaU option, which will be investigated in more depth in the following sections in which the results of the mixed logit model are analysed, since the Hausman tests indicated that the mixed logit model would be a better fit for the data.

3.4.2 The Mixed Logit Model

The mixed logit models were analysed both in Stata and R. The analysis was started in Stata but continued in R since it is only possible to use Halton draws for the mixed logit model in Stata. However, it has been established that Halton draws do not perform well in models with high dimensions (i.e., a high number of random coefficients) (Lancsar *et al.*, 2017; Mariel *et al.*, 2021) and are not recommended for models with more than 5 random coefficients (Hess and Palma, 2022). The analysis was thus switched to R (version 4.1.2) using the apollo package (version 0.2.7) published by Hess and Palma (2019). 3000 Sobol draws were used for the analysis in R since they outperformed various other draws, including Halton draws in a study by Czajkowski and Budziński (2019). The results presented in this chapter are hence those of the models run in R.

3.4.2.1 General Results of the Mixed Logit Model

The data was analysed with the mixed logit model in preference and WTP space with treatment interacting with the carbon attribute. Overall, the results show positive and significant coefficients for all ES attributes which suggests that all attributes influence participants' preferences and WTP. The coefficients for the payment attribute are significant and as expected negative, which indicates that the payment factor worked as a deterrent for respondents when they made their choices. The results thus confirm those of the conditional logit model. Moreover, the standard deviation coefficients, which show levels of heterogeneous preferences, are significant for the carbon attribute, an increase of the biodiversity attribute, the asc, and to a lower level of

[13] IIA assumption: "the ratio of the probabilities of choosing one alternative over another (given that both alternatives have a non-zero probability of choice) is unaffected by the presence or absence of any additional alternatives in the choice set" (Louviere *et al.*, 2000).

confidence for the payment attribute. Respondents hence showed heterogeneous preferences for these attributes. The results are presented in Table 3.8.[14]

Table 3.8: Mixed logit model results with interaction treatment - carbon in preference and WTP space.

	Preference Space		WTP space	
Number of observations		10,536		10,536
Number of parameters		20		20
Log likelihood		-2733.08		-2876.58
AIC		5506.16		5793.17
BIC		5629.44		5916.44
Choice	**Mean (St. error)**	**Standard deviation (St. error)**	**Mean (St. error)**	**Standard deviation (St. error)**
Maintaining current biodiversity level	0.649*** (0.070)	0.141 (0.304)	107.541*** (14.582)	38.322** (15.174)
Increasing biodiversity level	1.003*** (0.087)	0.074 (0.424)	202.743*** (16.348)	0.894 (15.203)
Maintaining current flood defence level	0.462*** (0.070)	0.011 (0.228)	77.821*** (12.428)	84.744*** (13.405)
Increasing flood defence level	1.054*** (0.091)	0.649*** (0.144)	222.433*** (16.315)	82.998*** (15.176)
Marginal increase in carbon storage (short information)	0.054*** (0.005)	0.046*** (0.007)	10.133*** (0.994)	6.535*** (0.978)
Marginal increase in carbon storage (long information)	0.048*** (0.005)	0.030*** (0.008)	8.594*** (0.745)	2.520*** (0.886)
Providing bridges and boardwalks	0.493*** (0.071)	0.095 (0.498)	72.070*** (11.756)	23.442* (12.615)
Providing bridges, boardwalks, and bird hides	0.342*** (0.074)	0.346 (0.200)	58.408*** (10.656)	70.483*** (12.577)
asc (alternative specific constant)	3.243*** (0.328)	2.732*** (0.331)	808.605*** (87.755)	858.574*** (97.178)
Increase in income tax for 10 years	-0.041*** (0.010)	0.819* (0.507)	-0.020*** (0.007)	0.058* (0.041)

***, ** and * indicate 1,5 and 10% significance levels respectively

3.4.2.2 The Effect of Information

Since the effect of information on the preferences and WTP for a marginal increase in saltmarsh carbon storage was of interest, the carbon attribute was interacted with treatment (i.e., the carbon attribute was split into the two subsamples by treatment). A Wald test was conducted to determine whether there was a significant difference in the carbon attribute coefficient means according to treatment in preference space (null hypothesis: Mean(untreated sample) = Mean(treated sample)); the null hypothesis could not be rejected ($p > 0.05$). Hence, there was no statistically significant

[14] The WTP space payment coefficient presented in Table 3.8 is the underlying payment coefficient used to calculate the WTP coefficients of the attributes.

difference between the carbon coefficient means of the two subsamples. This result was also confirmed in WTP space.

However, the Wald test was also conducted for the standard deviation of the carbon attribute preference coefficients to determine whether there was a significant difference between the subsamples. Since the null hypothesis could be rejected ($p < 0.05$), the results thus also indicate that participants allocated to the treated subsample (i.e., more information) make less random choices than the respondents that received no treatment (i.e., less information). This allows for the interpretation that the increased information decreased the randomness of the participants' choices as they were better able to develop their preferences and make decisions. Again, this result was confirmed in WTP space.

Further, to determine the model that best fits the data, the mixed logit model was also fit in preference space without any interactions with treatment (Model 1, see Appendix B, Table B.2). Since the models are nested, the Likelihood Ratio test was then used to determine which specification of the mixed logit model fits the data better. The test indicated that Model 1 fits the data better.[15] In WTP space however, the Likelihood Ratio test indicated that the model allowing for interaction between the treatment and the carbon attribute is the better fit for the data than Model 1 even though previous Wald test results were the same both in preference and WTP space. This may indicate that the difference in standard deviation between the two subsamples (i.e., the difference in the heterogeneity in respondents' choices) was not significant enough in preference space for the model to fit better but that it is significant enough in WTP space to cause a better fit for the model where the carbon attribute interacts with treatment. The analysis of participants' average WTP was thus continued with the model where the carbon attribute interacts with treatment.

3.4.2.3 Robustness Tests

There is an ongoing discussion in the choice modelling literature regarding the adjustment for scale heterogeneity. Scale heterogeneity is the "variance of a variance term or the standard deviation of utility over different choice situations" (Greene and Hensher 2010, 413) and it is necessary to adjust for it when observations from two datasets are combined (Swait and Louviere, 1993). Accounting for scale heterogeneity has the advantage that it captures 'extreme' respondents whose preferences were either almost lexicographic or very random (Faccioli *et al.*, 2019; Fiebig *et al.*, 2010). Moreover, Czajkowski *et al.* (2014, 328) criticise that "the MNL model implausibly assumes that the random term is iid for all choices, that is, the scale coefficient [σ] is

[15] LR test comparing mixed logit in preference space with carbon interacting with treatment (Table 3.8) vs Model 1 (Appendix B, Table 2): $p > 0.05$.

the same for every respondent, choice task, and alternative. This results in the assumption that each respondent makes his choices with the same degree of randomness." According to Fiebig *et al.* (2010), the generalised multinomial logit model (GMNL model) is well-suited to account for scale heterogeneity. However, Hess and Train (2017) argue that models which assume that they are only accounting for scale heterogeneity actually capture all sources of correlation and that using a mixed logit model with full covariance also takes account of scale heterogeneity. The mixed logit model with full covariance was thus used as a robustness test for the results in preference space before fitting the model in WTP space. The results of the models with full covariance are presented in Table 3.9.

The result that there is no significant difference between the means of the carbon attribute when it interacts with treatment but that there is, however, a significant difference between the standard deviations holds true when full covariance is specified. Moreover, the models with correlation specified provide evidence that respondents' choices are correlated and not independent, which demonstrates that all attributes are important in shaping their preferences. The model with all correlation between attributes and without carbon interacting with treatment shows a strong correlation of the biodiversity attribute with all other attributes. The increase level of biodiversity in particular, correlates with the increase level of the flood protection attribute, the carbon attribute, and both recreation attribute levels. The model with all correlation between attributes and carbon interacting with treatment confirms this observation (significant ($p < 0.1$) correlations results table see Appendix B, Table B.3).

Table 3.9: Mixed logit model results in preference space, all correlation between attributes allowed.

	No interaction with treatment		Interaction of treatment with the carbon attribute	
Number of observations		10,536		10,536
Number of parameters		54		65
Log likelihood		-2689.36		-2686
AIC		5486.72		5501.99
BIC		5819.57		5902.65
Choice	**Mean (St. error)**	**Standard deviation (St. error)**	**Mean (St. error)**	**Standard deviation (St. error)**
Maintaining current biodiversity level	0.939*** (0.12061)	0.873*** (0.207)	0.983*** (0.127)	0.947*** (0.203)
Increasing biodiversity level	1.438*** (0.15281)	0.777*** (0.184)	1.459*** (0.160)	0.856*** (0.186)
Maintaining current flood defence level	0.671*** (0.10780)	0.707*** (0.203)	0.644*** (0.107)	0.712*** (0.215)
Increasing flood defence level	1.534*** (0.16)	0.762*** (0.235)	1.450*** (0.161)	0.784*** (0.246)
Marginal increase in carbon storage	0.066*** (0.006)	0.051*** (0.012)	x	x
Marginal increase in carbon storage (short information)	x	x	0.070*** (0.009)	0.060*** (0.015)
Marginal increase in carbon storage (long information)	x	x	0.062*** (0.008)	-0.026 (0.031)
Providing bridges and boardwalks	0.631*** (0.108)	0.209 (0.402)	0.689*** (0.120)	-0.011 (0.297)
Providing bridges, boardwalks, and bird hides	0.378*** (0.110)	0.358 (0.396)	0.397*** (0.128)	0.256 (0.336)
asc (alternative specific constant)	3.083*** (0.377)	1.836*** (0.438)	2.986*** (0.357)	2.308*** (0.410)
Increase in income tax for 10 years	-0.010*** (0.004)	0.053* (0.036)	-0.008*** (0.001)	0.037*** (0.013)

The sign of the estimated standard deviations is irrelevant: interpret them as being positive
***, ** and * indicate 1,5 and 10% significance levels respectively

3.4.2.4 Further Considerations

Overall, six different versions of the mixed logit model in preference space and two in WTP space were run in both Stata and R to check for robustness and best model fit. Table 3.10 provides an overview of these models and whether they provided results. The results in Stata and R were largely the same, except that for the models in preference space with treatment interacting with carbon and with all correlation allowed, the standard deviation of the two carbon coefficients was not significantly different ($p = 0.4176$) in the Stata results while it was significantly different ($p = 0.0116$) in the R results. The R results, however, match the results of the corresponding models where correlation between the attributes was not specified (in Stata and R). A table that presents these results and differences in detail is included in the Appendix (Appendix B, Table B.4).

It needs to be pointed out that it was also attempted to run the mixed logit model in preference space with treatment interacting with all attributes, but this model did not produce reliable results in R. This is also the case for the mixed logit model in preference space with treatment interacting with all attributes and allowing for all correlation between the attributes. In Stata, the model did not converge after over a week of run-time. Furthermore, the WTP space models in Stata did not converge with the number of Halton draws required for reliable results.

Table 3.10: Mixed logit models in Stata and R and whether they produced results.

Model	Description	Stata	R
MixlPrefC	Mixed logit model in preference space, no interaction with treatment	✓	✓
MixlPrefT-C	Mixed logit model in preference space, interaction treatment - carbon	✓	✓
MixlPrefT-A	Mixed logit model in preference space, interaction treatment - all	✓	x
MixlPrefCcorr	Mixed logit model in preference space, no interaction with treatment, allowing for all correlation between attributes	✓	✓
MixlPrefT-Ccorr	Mixed logit model in preference space, interaction treatment - carbon, allowing for all correlation between attributes	✓	✓
MixlPrefT-Acorr	Mixed logit model in preference space, interaction treatment - all, allowing for all correlation between attributes	x	x
MixlWTPC	Mixed logit model in WTP space, no interaction with treatment	x	✓
MixlWTPT-C	Mixed logit model in WTP space, interaction treatment - carbon	x	✓

3.4.2.5 Participants' WTP

The results (Table 3.8) suggest that respondents, on average, preferred change over the status quo and support the management of saltmarshes for their ES. This is indicated by the positive mean preference and WTP coefficients of the alternative specific constant[16]. Due to the coding in the data analysis, the carbon attribute WTP coefficients represent the participants' marginal WTP per year over a time span of 10 years for a marginal increase of the carbon storage service equivalent to the emissions of 1000 cars/year. The participants' marginal WTP to increase the carbon storage service equivalent to the emissions of 1,000 cars was thus £10.13/year (untreated subsample) and £8.59/year (treated subsample) for a time span of 10 years (see Table 3.8). The carbon attribute had 6 levels, one of which was the status quo (i.e., release of stored carbon equivalent to the release of the annual emissions of 10,000 cars) at no cost. The WTP for the other levels can be calculated relative to the status quo level. Maintaining the current carbon storage level of saltmarshes thus requires preventing the release of carbon equivalent to the annual emissions of

[16] The alternative specific constant was coded to take the value of 1 in both management options and 0 in the BaU option.

10,000 cars and is valued at £101.30/year (untreated subsample) and £85.90/year (treated subsample); the average WTP to achieve an increase of carbon storage equivalent to the annual emission of 4,000, 10,000, 16,000, and 20,000 cars are presented in Table 3.11.

Table 3.11: The average WTP for the different levels of the carbon attribute.

Carbon Attribute Level with corresponding % change in total carbon stored in Scottish saltmarshes	**Average WTP (payment/year for 10 years)**	
	Untreated Subsample	**Treated Subsample**
BaU: *release* of carbon currently stored equivalent to the annual emissions of 10,000 cars (-2.5%)	-	-
Maintaining current levels of carbon storage (±0%)	£101.30	£85.90
Increase of carbon stored equivalent to the annual emissions of 4,000 cars (+1%)	£141.82	£120.26
Increase of carbon stored equivalent to the annual emissions of 10,000 cars (+2.5%)	£202.60	£171.80
Increase of carbon stored equivalent to the annual emissions of 16,000 cars (+4%)	£263.38	£223.34
Increase of carbon stored equivalent to the annual emissions of 20,000 cars (+5%)	£303.90	£257.70

Furthermore, it is worth noting that for both the flood protection and the biodiversity attributes, a higher mean WTP was shown for an increase of the service rather than for just maintaining it (i.e., preventing a decline, which was the BaU option). This is not the case for the recreation attribute; respondents have a positive mean WTP for increasing access to marsh with bridges and boardwalks but show a lower mean WTP for adding the same infrastructure with additional bird hides. This suggests that the majority of respondents have a low interest in bird hides but would like improved access to saltmarshes. Since there is no significant statistical difference between the means of the carbon attribute coefficients of the two subsamples in both preference and WTP space and for simplification, the overall present value results of the average WTP for the carbon ES and the following LCA are presented without splitting by subsamples.

3.4.2.6 Present Value

Since the payment vehicle has a timespan of 10 years, the Present Value (PV) was calculated for this timespan. A discount rate of 3.5% was used in line with current policy practices in the UK (HM Treasury, 2020) and 2.3%, which was used by Lockwood and Drakeford (2021) based on the analysis of Drupp *et al.* (2018) regarding suitable discount rates. The WTP of the carbon attribute when the sample was not split by treatment was £8.94. Applying the discount rate of 3.5% to this value returned the PVs presented in Table 3.12.

Table 3.12: Total PV of the WTP for the carbon attribute with a discount rate of 3.5% and 2.3% applied to the 10-year timeframe specified in the DCE.

	Total PV of WTP for the different levels of the carbon attribute in £					
	Equivalent to emissions of 1,000 cars	**Maintaining current C storage (±0%)**	**Incr. carbon storage equiv. to emissions of 4,000 cars (+1%)**	**Incr. carbon storage equiv. to emissions of 10,000 cars (+2.5%)**	**Incr. carbon storage equiv. to emissions of 16,000 cars (+4%)**	**Incr. carbon storage equiv. to emissions of 20,000 cars (+5%)**
3.5% disc. rate	£74.35	£743.50	£1040.91	£1487.01	£1933.11	£2230.51
2.3% disc. rate	£79.06	£790.58	£1106.82	£1581.17	£2055.52	£2371.75

The total PV of the WTP for maintaining the current carbon storage level of Scottish saltmarshes would thus be £743.50. For comparison, with the lower discount rate of 2.3%, the total PV for maintaining the current carbon storage level would be £790.58.

3.4.3 Latent Class Analysis

To conduct the LCA, the explanatory variables had to be initially summarised through a k-means cluster analysis. Without this summarisation, the LCA model struggled to converge due to the high number of explanatory variables. As mentioned in section 3.3.6.1, three groups were identified within the explanatory variables on which the cluster analysis was performed: (i) demographic variables, (ii) variables regarding participants' familiarity and existing knowledge about saltmarshes, and (iii) variables that provide information regarding participants' personal experiences and attitudes.

3.4.3.1 The Number of Classes and General LCA Results

It was determined through class enumeration that the model with three classes is the best fit for the DCE data. The AIC, CAIC, and BIC fit statistics were used (Table 3.13) and the posterior class probability (i.e., how likely is it that a participant ends up in a particular class when their sequence of choices is taken into account) was checked. For the three-class model, this probability was 95.52% for class 1, for class 2 it was 90.67%, and for class 3 97.07%. Then the results of the most promising class model were checked to see whether they seemed reasonable.

Table 3.13: LCA fit statistics.

Classes	**AIC**	**CAIC**	**BIC**
2	5763.856	5932.178	5901.178
3	5484.46	**5772.235**	**5719.235**
4	5459.483	5866.712	5791.712
5	**5443.893**	5970.576	5873.576

The mean WTP of the members of the different classes was determined with a postestimation command available in Stata.[17] The results are presented in Table 3.14 below.

Table 3.14: Latent class model with 3 latent classes.

Latent Class model	**Class 1 (64.2%)**			**Class 2 (26.9%)**			**Class 3 (8.9%)**		
3 Classes	**Estimate**	**Std. Err.**	**p-val(0)**	**Estimate**	**Std. Err.**	**p-val(0)**	**Estimate**	**Std. Err.**	**p-val(0)**
Preferences									
Maintaining current biodiversity level	0.759***	0.092	0.000	0.121	0.145	0.402	0.552	0.372	0.138
Increasing biodiversity level	1.263***	0.132	0.000	0.060	0.175	0.734	-0.410	0.474	0.387
Maintaining current flood defence level	0.508***	0.078	0.000	-0.035	0.147	0.810	-0.857*	0.457	0.061
Increasing flood defence level	1.270***	0.137	0.000	0.147	0.183	0.422	0.082	0.388	0.832
Marginal increase in carbon storage	0.053***	0.005	0.000	0.015***	0.005	0.004	-0.016	0.017	0.330
Providing bridges and boardwalks	0.451***	0.079	0.000	0.242	0.150	0.107	-0.076	0.440	0.863
Providing bridges, boardwalks, and bird hides	0.277***	0.083	0.001	0.091	0.155	0.556	0.357	0.400	0.372
asc (alternative specific constant)	0.944***	0.224	0.000	2.411***	0.231	0.000	-1.912***	0.531	0.000
Increase in income tax for 10 years	-0.001	0.001	0.103	-0.013***	0.001	0.000	-0.002	0.002	0.234
WTP									
Maintaining current biodiversity level	531.0	328.4	0.106	9.7	11.7	0.406	249.1	265.3	0.348
Increasing biodiversity level	883.4*	495.1	0.074	4.8	14.0	0.735	-185.3	271.8	0.495
Maintaining current flood defence level	355.4	228.5	0.120	-2.8	11.8	0.810	-386.9	370.0	0.296
Increasing flood defence level	888.6*	490.3	0.070	11.8	14.6	0.420	37.1	174.6	0.832
Marginal increase in carbon storage	37.3*	20.9	0.073	1.2***	0.4	0.003	-7.4	10.8	0.492
Providing bridges and boardwalks	315.5	202.5	0.119	19.3	12.2	0.114	-34.3	200.2	0.864
Providing bridges, boardwalks, and bird hides	193.7	123.9	0.118	7.3	12.3	0.556	161.3	223.6	0.471
asc (alternative specific constant)	660.2*	380.0	0.082	192.6***	15.6	0.000	-863.3	829.2	0.298
Class membership							**Reference Class**		
Demographics (clustered)	0.367	0.344	0.286	0.141	0.373	0.704	-	-	-
Have heard about saltmarshes	0.159	0.371	0.669	-0.008	0.398	0.983	-	-	-
Have been to a saltmarsh	-0.293	0.427	0.493	-0.744*	0.446	0.095	-	-	-

[17] While the survey was only sent out to Scottish residents, some participants were located outside of Scotland in England and Northern Ireland when the survey was undertaken. A robustness test was conducted to check whether the results change when participants who were located outside of Scotland were excluded. The test showed that the results are robust; excluding the participants located outside of Scotland did not significantly change the class allocation.

Knowledge Question: Biodiversity	0.197	0.232	0.397	0.093	0.254	0.714	-	-	-
Knowledge Question: Flood Protection	0.359	0.253	0.156	0.755***	0.275	0.006	-	-	-
Knowledge Question: Carbon Storage	0.303	0.267	0.256	-0.102	0.281	0.717	-	-	-
Knowledge Question: Recreation	0.408*	0.231	0.077	0.102	0.246	0.677	-	-	-
Environmental Attitude (clustered)	-1.490***	0.372	0.000	-0.586	0.406	0.149	-	-	-
Have been affected by flooding	-1.493	1.198	0.213	-2.161*	1.188	0.069	-	-	-
Risk Scale	-0.074	0.076	0.331	-0.082	0.081	0.308	-	-	-
Discount Scale	0.337***	0.090	0.000	0.184*	0.095	0.052	-	-	-
Distance to Coast	-0.005	0.017	0.785	-0.022	0.020	0.272	-	-	-

***, ** and * indicate 1,5 and 10% significance levels respectively

3.4.3.2 Class 1 – Improvement of all Attributes: The Ideologists

Class 1 is the largest class (64.2%), and respondents have statistically significant positive preference coefficients for all attributes including the ASC, except for the payment attribute. The payment attribute is negative as expected but is not statistically significant. This indicates a strong preference of the management scenarios over the status quo. Although, no WTP coefficients were statistically significant at $p < 0.05$, 95% confidence level for this class, there were several that were above this threshold by only a fine margin and significant at $p < 0.1$. Consistent with the strong preference for change, these coefficients were those for an increase in biodiversity ($p = 0.074$) and flood protection ($p = 0.070$) as opposed to the coefficients that represented only maintaining those services at the current level. The carbon attribute WTP coefficient and the ASC WTP coefficient were also almost significant at the 95% confidence level ($p = 0.073$ and $p = 0.082$ respectively). The observed preferences and lacking significance of the payment attribute suggest members of this class followed an ideological inclination without paying attention to the payment attribute. This interpretation is supported by the characteristics that are significant for participants that were sorted into this class. Respondents were more likely to be allocated to this class if they scored high on the attitude variable, which was summarised from questions regarding respondents' environmental attitude through a cluster analysis. Respondents of this class were thus more likely to have a favourable attitude towards the environment. Moreover, respondents were more likely to be allocated to class 1 if they scored high on the discounting scale and were thus more likely to give something up that was beneficial to them in the present in order to benefit more from it in the future. Although statistically not as robust ($p = 0.077$), respondents were also more likely to have a previous knowledge of saltmarshes' value for recreation.

3.4.3.3 Class 2 – Improvement of Carbon Storage: The Rationalists/Prioritisers

Class 2 is the second largest class (26.9%). Respondents have statistically significant positive preference coefficients only for the ASC, the carbon attribute, and the payment attribute, which is negative as expected. Respondents allocated to this class thus prefer the management options over the status quo, but the payment acts as a deterrent. The preference for change regarding the carbon attribute is also reflected in the WTP coefficients of this class. Respondents are willing to pay for change and a marginal increase in carbon storage, but not for a marginal increase in any of the other attributes. However, their average WTP for both of these factors is lower than the average WTP of members of class 1 and the payment attribute acted as a deterrent. This indicates that members allocated to this class behaved rationally regarding the payment attribute, exhibiting a more cautious attitude with preferences for change decreasing when prices increase. An interpretation for this could be that since members of this class only showed an interest in the marginal increase of the carbon attribute, the payment associated with the management options were frequently considered too high; or that the carbon attribute was prioritised over other attributes due to limited means to pay for change as opposed to members of class 1 who preferred change at all costs. Yet, the demographics variable, which includes income, was not significant for determining class allocation, which supports the notion that members of this class exhibited more rational behaviour rather than being limited by a higher budget constraint. Respondents were more likely to be allocated to class 2 if they scored high on the discounting scale ($p = 0.052$); however, they were less likely to do so relative to class 1. They were also more likely to be allocated to this class if they had a previous knowledge of saltmarshes' value for flood protection ($p = 0.006$). This may imply that these participants were more familiar with saltmarshes due to proximity; however, distance to the coast was included as a variable in the LCA and was not a significant factor in class allocation. Nevertheless, this knowledge implies a greater familiarity with saltmarshes.

3.4.3.4 Class 3 – Business as Usual: The Immovables

Class 3 is the smallest class (8.9%). The ASC coefficient was significant; however, it was negative which indicates that respondents prefer the status quo over the management scenarios. Furthermore, the coefficient for maintaining current flood protection levels is negative and significant but at a low level of confidence. The payment coefficient is not significant since the BaU option comes at no cost. Correspondingly, no WTP coefficient was significant for respondents of this class, and none were close to the threshold of $p = 0.05$. Relative to members of classes 1 and 2, respondents were more likely to be allocated to this class if they scored lower for the environmental attitude variable, which means that they were, for example, less likely to support a political focus on the environment, recycle, or donate to conservation associations (see Table 3.6). Moreover, they were more likely to be allocated to this class if they scored lower on the discounting scale and if they scored lower on the previous knowledge questions for the recreation and flood protection services of saltmarshes. They were also less likely to have previously heard of saltmarshes before taking the survey. This implies that a lower familiarity with the ecosystem may lead to a reduced preference for change and WTP for initiatives that improve the state of the ecosystem or at least of some of its services.

3.5 Discussion

3.5.1 The Effect of Information

Overall, participants preferred change over the status quo, but the payment acted as a deterrent. The treatment, in the form of additional information, did not have an effect on WTP but participants who received the additional information made less random choices. Boyle (1989) reports comparable results regarding the effect of information on preferences for a contingent valuation study on the trout fishery in Wisconsin; he finds that "gross changes in a minimal commodity description can significantly alter value statements and small refinements in a specific commodity description do not alter estimated means" (Boyle 1989, 61). He cautions against simply applying the findings to other contexts since these results were derived from a distinct application (i.e., trout fishery in Wisconsin). He also stresses that researchers need to be careful to provide complete information on the commodity that is to be valued to the respondents allocated to the control group. Consequently, the additional information provided to the treated group would be a refinement of the information the control group receives. This matches the approach taken in this study of providing additional information as a treatment to one subsample. Further, Boyle's (1989) general results can be confirmed. Boyle's (1989) appeal to be cautious about transferring results to other contexts is confirmed by Shapansky *et al.* (2008) who found that different levels of information and involvement in the valuation assessment did not reliably decrease the error variance.

The results of the LCA also revealed another interesting point regarding information. The members of class 3 who preferred the status quo over change, were less likely to have previously heard of saltmarshes before taking the survey. A greater familiarity with saltmarshes thus appears to be connected to a WTP for change. This observation is in line with previous observations made in the literature. Accordingly, people tend to be more knowledgeable about things they care about and familiarity with the subject of the study can influence participants' WTP (Needham *et al.*, 2018).

3.5.2 The Environmental Management Budget

The DCE analysis consistently demonstrated that the preference and WTP for saltmarsh management goes beyond managing the carbon ES. Instead, there is a preference and WTP for a holistic approach that increases all saltmarsh ES. Both the conditional logit and mixed logit models' preference coefficients were significant for all attributes. Furthermore, the LCA class with respondents that preferred an increase in all ES was by far the largest class (64.2%). The class whose members prioritised the carbon storage ES was significantly smaller with less than half (26.9%) the number of members. And although there are indications that the largest class that preferred an increase in all ES did not choose rationally (i.e., the payment attribute had no effect on the choices), these results still provide a clear indication of the Scottish public's preferences. Moreover, providing more information on the saltmarsh carbon storage ES did not significantly increase respondents' preferences or WTP for this attribute. Based on these results, prioritising the carbon storage ES can thus not be justified with public preference. This, however, has an influence on the management budget for saltmarshes. Since there was a preference and WTP for all ES, the overall management budget is based on the WTP of participants for all ES and is therefore larger than it would have been if there was only a preference for the carbon storage ES.

Since the DCE was completed by a representative sample, of the Scottish population, it is possible to extrapolate and calculate an Environmental Management budget from the WTP results of the DCE. The resulting values as presented in Table 3.15 are quite high. This may be due to the fact that more than the majority of the respondents (i.e., members of allocated to LCA class 1) did not make rational choices that were influenced by the payment attribute.

Table 3.15: Total PV in £ of the average WTP for maintaining and increasing all ES applied to the number of Scottish income taxpayers.

	Maintaining current ES (in million £)	**Increasing current ES (in million £) (highest level for carbon and recreation)**
3.5% disc. rate	£21,081	£30,932
2.3% disc. rate	£22,416	£32,891

To illustrate how high these values are, the overall value of the carbon attribute as presented in Table 3.16 can be compared to the Department for Business, Energy & Industrial Strategy's (BEIS) 2020 central series carbon price (£241/tCO_2; BEIS, 2021a). When the 2020 central series carbon price (£241/tCO_2; BEIS, 2021a) is applied to the estimated total carbon storage of Scottish saltmarshes (494800 tonnes of C) on which the levels of the carbon attribute are based (Beaumont *et al.*, 2014), the management budget is £437,635,756; if the 2020 high series carbon price (£361/tCO_2; BEIS, 2021a) is applied, the budget is £655,545,676.[18] Both of these values are less than half of the value estimated from the DCE results for maintaining the current carbon storage of saltmarshes.

[18] Since the carbon price is presented per tonne of CO_2, it needs to be converted to per tonne of C before it can be applied to the total carbon storage estimate for Scottish saltmarshes since this is presented in tonnes C.

Table 3.16: Total PV of the WTP for the carbon attribute with a discount rate of 3.5% and 2.3% applied to the 10-year timeframe specified in the DCE and the total number of Scottish income taxpayers.

	Total PV of WTP for the different levels of the carbon attribute (in million £)					
	Equivalent to emissions of 1,000 cars	**Maintaining current C storage (±0%)**	**Incr. carbon storage equiv. to emissions of 4,000 cars (+1%)**	**Incr. carbon storage equiv. to emissions of 10,000 cars (+2.5%)**	**Incr. carbon storage equiv. to emissions of 16,000 cars (+4%)**	**Incr. carbon storage equiv. to emissions of 20,000 cars (+5%)**
3.5% disc. rate	£188	£1,878	£2,629	£3,756	£4,883	£5,634
2.3% disc. rate	£200	£1,997	£2,796	£3,994	£5,192	£5,991

A further reason why these values are so high could be the increased awareness of the public regarding climate change and its effects and the possibility of capitalising on a NbS to mitigate some of its effects. Studies have found that there is a connection between respondents' WTP or willingness to donate and how charismatic a species is (Shreedhar and Mourato, 2019) and there are also indications that this assumption holds true in the context of carbon credits generated from Scottish peatland restoration (Segal, 2022). This may hence also be the case for Scottish saltmarshes.

3.5.3 Comparison to Other Literature/Study Results

It is difficult to compare the WTP results for the carbon storage service with the valuation results of other studies. All of the studies that value the saltmarsh carbon storage ES in the UK use different valuation methods. Even Luisetti *et al.* (2011) and Luisetti *et al.* (2014) who include a DCE approach, value the carbon storage service separately with a differing method. The WTP results of this study are presented per respondent and had to be transformed into a medium that is comprehensible to respondents (i.e., car emissions) while other studies determine value by ha or by tC. To illustrate the difficulty of comparing the estimates, the average WTP per respondent was calculated for an increase of 1 tC storage in saltmarshes, which is £ 0.007; converted to CO_2e it is even less with 0.002 £/tCO_2e. However, the carbon price, which is based on marginal abatement costs (BEIS, 2021a) would not be paid by a single person. The 2020 carbon price, which was the year the DCE was carried out, was set as 241 £/tCO_2e with a 50% sensitivity range (BEIS, 2021a). A further factor that makes a comparison difficult is the dynamic nature of the carbon price. Luisetti *et al.* (2011) use a carbon price based on the SCC, which has since been declared as not fit for use for determining the carbon price (BEIS, 2021a). Luisetti *et al.* (2014) and Beaumont *et al.* (2014) used the DECC's (2011) non-traded carbon price that replaced the SCC. Yet, this carbon price has since also been updated and increased (BEIS, 2021a). The dynamic nature of the carbon price is caused by the "target consistent approach" (BEIS, 2021a) that BEIS has been using since 2009 to estimate the carbon values. The values are estimated as the marginal abatement cost of meeting targets and are thus based on several variable factors, such as changes in international and domestic targets as well as new understanding of technology costs (i.e., decarbonisation technologies) and availability (BEIS, 2021a).

Additionally, independent from the method that was used in valuation studies, the determined values are also not fixed. Quite the contrary, it can be expected that natural capital and ES values will increase in the future since the habitat decline causes them to become more stressed and scarcer in the future; while the supply diminishes, the demand will remain or even increase leading to rising prices. This creates a strong argument for conservation which can supplement the ethical rationales for conservation (Costanza *et al.*, 1997; Grunewald and Bastian, 2015).

3.5.4 Limitations

This study was focused specifically on saltmarshes, but there is a wide range of different coastal, terrestrial, and marine habitats that can provide similar ES (e.g., seagrasses, mudflats, sand dunes, etc.) and as Himes-Cornell *et al.* (2018) caution, there are differences how communities value services provided by their local ecosystems; further studies across different ecosystems are thus necessary to determine whether the findings of this study regarding the effect of information can be generalised for EM in Scotland.

Moreover, it could be the case that the additional information provided to the treated group had no effect on average WTP since it was too in depth and that the information provided to the control group was perceived as sufficient to make a well-informed choice; hence, there could be a saturation threshold for information where additional new information no longer influences average WTP estimates (Needham *et al.*, 2018). This effect has been previously reported by Munro and Hanley (2001) and Bergstrom *et al.* (1989). Bergstrom *et al.* (1989) explain that their analysis indicated that additional information presented to respondents may not have been necessary for respondents to develop their preferences and WTP. Munro and Hanley (2001) similarly argue that if the good has a use value and respondents have already decided that they wish to use the good in question, values may not be very sensitive to new information.

A further limitation of the study design is that no information on attribute non-attendance was collected. Attribute non-attendance means that respondents ignore one or more attributes. These ignored attributes are thus not relevant to the respective respondent's utility and the need to make trade-offs is circumvented (Lagarde, 2013). Great care was taken to make this study as robust as possible. However, some trade-offs needed to be made to avoid including too many questions in the questionnaire and risking respondent fatigue. The inclusion of questions on respondents' background and existing knowledge on saltmarshes as well as the provision of information on the experiment and attributes was prioritised to enable the collection of robust data to address the research questions.

In addition, a limitation of the stated preference method is summarised by Costanza *et al.* (1997):

> In many cases the values are based on the current willingness-to-pay of individuals for ecosystem services, even though these individuals may be ill-informed and their preferences may not adequately incorporate social fairness, ecological sustainability and other important goals. In other words, if we actually lived in a world that was ecologically sustainable, socially fair and where everyone had perfect knowledge of their connection to ecosystem services, both market prices and surveys of willingness-to-pay would yield very different results than they currently do, and the value of ecosystem services would probably increase (Costanza *et al.* 1997, 258).

The validity of Costanza *et al.*'s (1997) concerns regarding ill-informed respondents is confirmed by Adam (1990) for saltmarshes. He stated that for a long time, the public attitude towards saltmarshes could be best expressed as "wetlands are wastelands" (Adam 1990, 381); so it cannot be expected that the public is aware of the benefits saltmarshes provide. This effect was mitigated by providing information on the habitat and the different ES. The other factors Costanza *et al.* (1997) name, such as the social fairness of respondents' preferences are inherent to the method and difficult to mitigate. Nevertheless, as the authors also state, these factors are more likely to cause an undervaluation and thus conservative valuation estimates rather than an overvaluation of ES. To address the issue of diminishing supply and increasing demand, Costanza *et al.* (1997) highlight the importance of increasing the influence of the natural capital stock in the decision-making process and that ecosystem service loss must be weighed against the benefits of a proposed project in the project appraisal stage. Moreover, Costanza *et al.* (2017) emphasise that public engagement and discourse are key elements in achieving sustainable resource use.

3.5.5 The Importance of Valuations for Policy

Costanza *et al.* (2017) included a table with different uses for ES valuation ranging from raising awareness to specific policy analyses. This highlights the value and importance of valuation studies for policy. The characterisation places Costanza *et al.*'s seminal (1997) article in the 'Raising Awareness and Interest' category, for example, and explains that the article was aimed at no specific policy or decision but that it was nevertheless successful due to the interest it received. Luisetti *et al.*'s (2013) article also falls into this category; the authors explain that their study aims to raise awareness about the blue carbon issue in the political domain. Raising awareness is thus not limited to the method of ES valuation but is also used for its application to different ecosystems. This study can be of use for policy in several ways; (i) it can be allocated to two of the uses the authors included in the table (i.e., (a) the 'Raising Awareness and Interest' category and (b) the Payment for Ecosystem Services' category) and (ii) it contributes to the understanding of the effect information can have on public preferences. Especially the latter contribution as well as the determination of participant's WTP for saltmarsh ES are valuable to policy. Provided with this information about WTP for ES, policymakers can propose management schemes in the hope that they will be supported.

3.6 Conclusion

The aim of this study was to conduct an ES valuation of Scottish saltmarshes with a focus on the carbon storage service and a test of the effect of information on respondents' average WTP. It was expected that better informed respondents would have a higher average WTP. A DCE was used to conduct the valuation and a split sample approach to test the hypothesis regarding the effect of information. One subsample received more information on the carbon storage ES (treatment) of saltmarshes than the other subsample (control group); respondents were randomly allocated to these subsamples. Otherwise, the surveys were identical for both subsamples; respondents received one out of three blocks of choice cards, again randomly allocated. This approach allowed for the testing of the effect of information on respondents' average WTP and to collect valuable information on the Scottish public's attitude towards management interventions for saltmarshes, which would not have been possible with a market price valuation. Moreover, the DCE allowed us to value several ES instead of focusing on the carbon storage service alone. This provides us with information on its value relative to other services and particularly, how the Scottish public values this service relative to other saltmarsh ES.

The study found that the management scenarios were preferred over the BaU option and that respondents showed a positive average WTP for all included ES, but that the payment acted as a deterrent. There was no indication that providing more information to the treated subsample increased their WTP for the carbon storage service; there was no statistically significant difference in the average WTP for a marginal increase of carbon storage between the treated group and the control group. However, the study found that providing more information decreases the randomness of respondents' choices. The results were robust across the preference and WTP space models. Moreover, they matched with the LCA results, with the biggest class showing a preference for a marginal increase of all included attributes.

The results show that there is support within the Scottish public to manage saltmarshes for their carbon storage benefit and to realise their potential as a NbS for climate change mitigation; information campaigns have the potential to support this process since they can help the public make more informed decisions. Additionally, although the focus was on the carbon ES, the results of the study also revealed that there is considerable support and WTP for the management of the other saltmarsh ES included in the experiment (i.e., biodiversity, flood defence, recreation). The flood protection ES, in particular, can provide additional benefits for climate change adaptation. It can thus be concluded that management of saltmarshes should go beyond the carbon storage service and the potential for climate change mitigation and take all ES into account in a whole-ecosystem approach to realise a wide range of benefits including both benefits for climate change mitigation and climate change adaptation.

4 BLUE CARBON POLICY INTEGRATION

4.1 Introduction

In 2018, the IPCC published the *IPCC Special Report on Global Warming of 1.5°C*, which clearly warned that while limiting global warming to 1.5°C was still possible, it would be out of our reach unless we strongly increased our mitigation ambitions to significantly reduce GHG emissions by 2030 (IPCC, 2018). The IPCC's results were highly publicised and on 28th April 2019, a climate emergency was declared in Scotland by the First Minister, Nicola Sturgeon, during a speech at her party's conference. The announcement was followed by the passing of the Climate Change (Emissions Reduction Targets) (Scotland) Act in September 2019, which establishes the Scottish Government's commitment to reach net-zero emissions by 2045 as opposed to the Climate Change (Scotland) Act 2009's 80% emissions reduction target by 2050 (Climate Change (Emissions Reduction Targets) (Scotland) Act 2019; Climate Change (Scotland) Act 2009). This requires reductions across all sectors and the exploration of nature-based emission reduction solutions such as blue carbon habitats.

The UK's GHG Inventory presents the UK's official reported GHG emission estimates and is the key tool for understanding UK emissions, including their origin and magnitude. It is used to measure progress towards domestic and international emission targets and is published by the Department for Business, Energy & Industrial Strategy (BEIS). The reporting follows the IPCC reporting requirements and standards and reports emissions dating as far back as 1990 to provide a baseline and comparable timeseries. The methodology for estimating emissions is improved every year. (BEIS, 2021b; BEIS, 2022). The GHG Inventory is relevant for blue carbon policy, especially for UK level target setting. However, this research focusses on strategic management of the resource in the Scottish policy framework, within which saltmarsh blue carbon habitats fall under marine spatial management and are covered by the Marine (Scotland) Act 2010 (Marine (Scotland) Act 2010) as introduced in section 1.6.2.

4.2 Aim of Chapter

The carbon storage potential of saltmarshes has recently gained interest in the policy arena as a NbS to climate change mitigation. However, the integration of this ES into Scottish policy is still in its infancy. This chapter is thus an important contribution to the blue carbon policy-making process since it addresses the overarching research question '*How can the saltmarsh carbon storage ecosystem service best be integrated into Scottish policy?*'. It also aims to complete the connection between science and policy that this thesis sets out to make. After investigating the uncertainty that is inherent to carbon storage in Scottish saltmarshes and the public preferences regarding its management and the public's WTP, this chapter dives into the question regarding which factors are important for a successful integration of this blue carbon resource into policy, which is an aspect that has not been researched in detail. Its objective is to analyse which factors shape blue carbon policy in the Scottish context, how they interact, and what their relative importance is.

As mentioned, it may be possible to include the carbon storage benefit of some blue carbon habitats, including saltmarsh, into the UK GHG Inventory. The UK Climate Change Committee recommended the inclusion of saltmarshes into the national GHG Inventory and BEIS as the responsible government department is progressing it, however, this process has not yet been completed (Austin *et al.*, 2022). Consequently, the management of all Scottish blue carbon habitats currently falls under the Marine (Scotland) Act (2010) within the remit of Scottish policy. Moreover, even if saltmarshes are included in the inventory, as the previous chapter demonstrated, saltmarshes are also valued for various other ES that should be taken into account and may require wider management considerations. The research question of this chapter is thus of considerable importance because it represents a timely reflection on the effectiveness of policy regarding Scottish blue carbon habitats.

4.3 Methodology

4.3.1 The Interview Method

There are different types of interviews that can be used for academic research ranging from highly structured interviews such as survey interviews, to unstructured interviews such as the in-depth interview. In structured interviews, the researcher uses pre-determined questions in a pre-set order and there is no deviation from the interview script. According to Robson and McCartan (2016), the only differentiating factor to survey questionnaires is the more frequent use of open-response questions. Unstructured interviews are more comparable to conversations and can be completely informal. The researcher has a general area of interest and within this area lets the conversation develop without asking pre-determined questions. Marshall and Rossmann (2006, 101) point out that in-depth interviews are "based on an assumption fundamental to qualitative research: The participant's perspective on the phenomenon of interest should unfold as the participant views it," which is referred to as the emic perspective. The opposite is the view of the interviewer or the etic perspective. Semi-structured interviews present the middle between these two opposing methods. The interviewer has pre-determined questions and a guide for the interview, but the wording and order of the questions are flexible and can be adapted according to how the interview develops (Robson and McCartan, 2016). Instead of being considered separate categories, interview types are part of a continuum with unstructured and structured interviews representing the two ends (Dunn, 2021). The interviews of this study can be located on the continuum between semi-structured and structured.

In contrast to other research methods, interviews do not just investigate facts and behaviour (e.g., what people want and what people do), but also participants' beliefs and attitudes (e.g., what they think and what they feel); beliefs and attitudes, however, are more difficult to ascertain than facts and behaviours. Robson and McCartan (2016) indicate that facts should be relatively easy to obtain and that specific questions on current or recent issues are most likely to provide high quality responses. The topic of climate change mitigation and adaptation is a timely and prevalent topic and can be quite emotionally charged (e.g., the Fridays for Future movement; climate change scepticism), which could influence interview results. However, the interest of this study lies in facts and behaviours; the interviews were conducted with experts in the environmental policy field and the questions were focussed on their work and professional experience rather than personal beliefs and feelings.

Interviews have many advantages such as the ability to provide deep insights into the interview's topic area. They can also be highly flexible and be adjusted to ask follow-up questions to investigate the underlying motives of participants' actions. However, interviews can suffer from a lack of standardisation and biases are difficult to rule out. Fundamentally, interviews are based on human interactions, which requires cooperation and honesty on the participant's side and good listening and interpersonal skills on the researcher's side (Marshall and Rossmann, 2006). Interviews are also time consuming and can cause respondent fatigue if they are too long. On the other hand, they only provide limited insights if they are too short. According to Robson and McCartan (2016), interviews shorter than 30 minutes hardly provide valuable material, whereas interviews longer than one hour place an unreasonable demand on participants' time and strongly increase the probability of respondent fatigue. Concluding from these thoughts, interviews of 45 minutes to one hour are desirable.

The interviews were conceptualised as expert-interviews. This stems partly from the researcher's positionality, which is defined as referring to "a researcher's social, locational, and ideological placement relative to the research project or to other participants in it" by Watson (2021, 127), since the researcher was a novice in the field of blue carbon policy compared to the target group of the interviews. And it stems from the fact that the interviews targeted participants' professional knowledge and expertise instead of their personal experiences or preferences. They were thus interviewed in their role as professionals and experts in the field.

4.3.2 Interviewee Selection

The snowballing method was used to identify and contact potential participants. For the snowballing technique it is important to find a person as entry point in the relevant organisation who is able to provide names of other potential participants. For this method it is important to keep track of and be transparent of how the selection of interviewees evolved. Interview studies usually include 10 – 30 participants, but this number can be higher, especially if several researchers are involved. The number of interviews that are conducted also depends on whether the material is central to a study or supplementary in a bigger project (Secor, 2010). Stratford and Bradshaw (2021) emphasise that there are only few rules regarding the sample size in qualitative inquiries and that it can also depend on other factors, such as the purpose of the research and what is needed in terms of knowledge, logistics, and resources. They further add that the richness of information and its validity is more dependent on the skills of the researcher than on sample size.

This study used two initial contacts, one within Marine Scotland and another within NatureScot, which are two principal organisations involved in policy for marine, nature conservation and climate. Marine Scotland is a department of Scottish Government and NatureScot is Scotland's nature agency that works to improve the Scottish environment and provides advice to Scottish government. It was established as an executive non-departmental public body under the Natural Heritage (Scotland) Act 1991 and its board is appointed by Ministers (NatureScot, 2022; Scottish Government, 2018b). These contacts then provided further names of potential interviewees. The snowballing method drew on interviewees' knowledge of their professional field and their networks to identify further interviewees. The snowballing method requires a loose sampling framework; however, to ensure a best effort is made for representative sampling, it was attempted to get a good horizontal and vertical distribution of interviewees within the two organisations. This was more successful within NatureScot than within Marine Scotland. NatureScot is organised under four strategic outcomes: (i) 'More People Enjoying and Benefitting from Nature', (ii) 'The Health and Resilience of Scotland's Nature is Improved', (iii) 'More Investment in the Management of Scotland's Natural Capital', and (iv) 'We have Transformed how we Work'. The interviewees of this organisation covered the first three of the four outcomes and had different positions in the NatureScot hierarchy. Whereas there were six interviewees from NatureScot (P3, P4, P5, P6, P7, and P8), it was only possible to get two participants from Marine Scotland (P1 and P2), with one having a policy facing role as the policy lead of the Marine Conservation Team and the other a science and evidence facing role as the Climate Change Lead of the Science Division. Thus, the total sample size is eight interviews.

4.3.2.1 Limitations of the Interviewee Selection

There were a few limitations to the interviewee selection. First, it would have been beneficial to have members from the Marine Analytical Unit, which is responsible for economic matters within Marine Scotland, as participants for the interview but unfortunately it was not possible for any of the members to participate. Second, it became clear quite early on that the sample size for the expert-interviews would be relatively small since the same names would come up at the end of the interviews when the question was asked about other potential participants. However, this fits with the context of blue carbon policy in Scotland. As blue carbon is still an emerging topic in policy, Scotland has a correspondingly small network of relevant policymakers and other experts. Furthermore, not every potential participant that was contacted responded. This was a difficulty particularly within Marine Scotland. Third, the interview sample is tightly focussed on members of two institutions. On one occasion, academic professionals were suggested as potential contacts. However, they were either outside of the Scottish context or too science-focussed. Two potential interviewees were located at the intersection of NatureScot and academia and of Marine Scotland and academia; however, after initial contact, it was impossible to set an interview date due to a lack of response. Moreover, when asking participants for further potential contacts, the interviewer also mentioned these contacts could be from other organisations than NatureScot and Marine Scotland. Yet, one participant responded that the key players were within these institutions.

4.3.3 Conceptualisation of Policy Phenomena

Prior to conceptualising the interview questions, several themes that are significant for Scottish environmental policy making were identified in literatures on the Scottish government, Scottish policymaking, and evidence-based policymaking. The emerging themes are illustrated in Figure 4.1 below.

Uncertainty was one of the themes that emerged. Cairney (2016b) describes the policy process in terms of its ideal but unrealistic form, called 'comprehensive rationality' and its realistic form, called 'bounded rationality'. While comprehensive rationality is characterised by aims ranked according to priority and a comprehensive search for information to support the policy process, bounded rationality describes a more realistic situation with competing aims, the inability to gather comprehensive information, and decisions based on **uncertainty**. **Uncertainty** has also been identified as a significant factor in blue carbon science (Duarte *et al.*, 2013; Spivak *et al.*, 2019) and was thus included as a candidate theme of interest that is likely to shape blue carbon policy. The inability to gather comprehensive **information** is closely related to this (Cairney, 2016b) and the amount and kind of information that is required for policy was thus also included as a theme. A further theme that appeared significant was the **policy process** due to the identification of a particular Scottish 'policy style' that focuses on consultation and public participation in policy development and implementation (Cairney, 2014). This Scottish 'policy style' together with Scotland's commitment to sustainable development (Scottish Government, 2020d), that includes the health and happiness of people as a goal, also suggests that **public acceptability** is an important factor. Yet sustainable development also sets out economic viability as a key goal (Barrow, 2005), highlighting that it remains an important factor under the sustainability agenda; **economic considerations** were thus likewise identified to be a likely significant factor. Lastly, Cairney (2016b) points out that mitigation and adaptation operate on different scales, with mitigation operating on a global and national scale, while adaptation is focussed on the regional and local scale. Blue carbon is significant for both mitigation and adaptation efforts; consequently, **scale** was identified as a potentially highly relevant factor in shaping blue carbon policy.

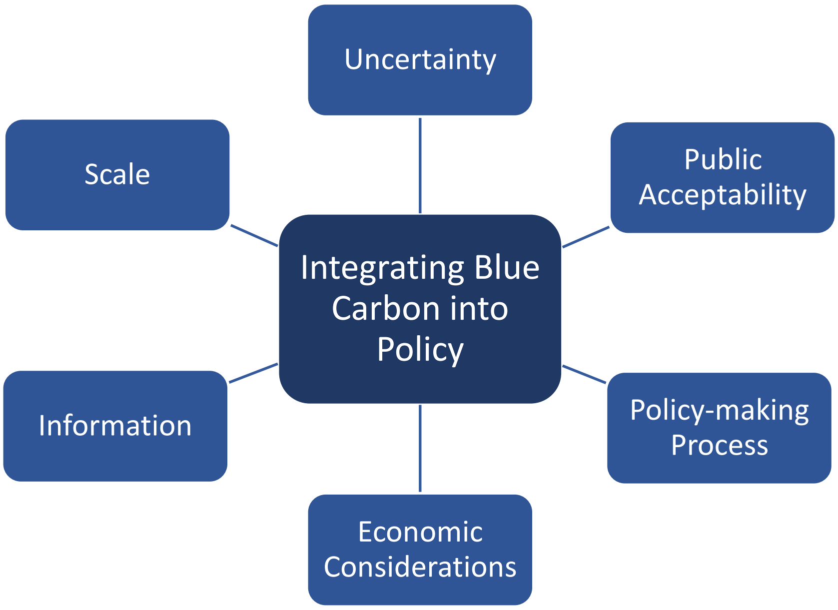


Figure 4.1: Themes identified as likely to shape blue carbon policy.

Once these key themes were identified, interview questions were developed that aim to address the research questions that correspond to each theme (See Appendix C, Table C.1).

For this study, the semi-structured interview method was the most prudent for several reasons: (i) the number of questions and the freedom provided by a semi-structured interview format for the respondents to explore themes; (ii) the focus on participants' expertise and requirement for a looser structure to raise new points that were not previously considered; (iii) the adaptability of questions targeted at different representatives in the policy process, and; (iv) in-depth exploration of research questions.

In line with Robson and McCartan's (2016) advice on semi-structured interviews, an interview script was prepared with questions and a number of possible subsequent questions (Appendix C, Figure C.1). The questions were grouped by themes, which could then be checked off by the interviewer once the questions of the corresponding theme were sufficiently covered. Prompts were integrated in the script to provide context for the respondents and as a starting point for the questions. The prompts were identical and consistent across all interviews. Yet, dependent on the participant's background, the wording of the questions was changed slightly to accommodate the perspectives of policymakers and participants who have a more advisory role. The open-ended questions provided many advantages such as more flexibility and in-depth answers, but also disadvantages such as the possibility of loss of control of the interviewer and increased difficulty regarding their analysis. Nevertheless, the advantages of open-ended questions outweighed the disadvantages in the context of this study.

Overall, the interview was structured with an introductory section, the main body that included the questions on the identified themes, and a wrap-up section. The introductory section followed the recommendation by Newing (2011) and started the conversation by inviting the interviewees to talk about themselves. Questions were asked about their position within their organisation and where they located themselves on a scale between science and policy. The wrap-up section included the question whether there were any aspects important to drafting and implementing blue carbon policy that were not touched upon in the interview to ensure that all important factors or themes regarding blue carbon policy were identified, and a snowballing recruitment question on whether the interviewees could suggest other important respondents that should be contacted for an interview.

Interview questions can be divided into primary and secondary questions. The former are original questions to initiate discussion on a new topic, while the latter are follow-up questions to encourage the interviewee to expand on their answer and dive deeper into the discussion (Dunn, 2021). The main interview questions on the different themes could be largely classified as sixteen primary questions and ten secondary questions according to this definition, although there seemed to be more of a smooth transition between the two question types rather than a clear delineation as is suggested by Dunn (2021), so these numbers should not be viewed as definite. As recommended, the questions followed a funnel structure from a broader introductory question to more detailed and focussed questions. The primary questions took the form of storytelling questions that encourage sustained input from the interviewee and identify an ordering of events or causative links, and opinion questions, which can elicit assertions and guesses. The opinion questions were questions that asked for the importance of different aspects about the themes on a scale from ‘not important’, ‘somewhat important’, ‘important’, and ‘very important’; all other questions were storytelling questions. A variety of prompts was used during the interview, including the formal secondary questions, clarifications, summaries, and receptive cues. These prompts, such as the secondary questions, can be part of the interview schedule but are often used without prior planning when appropriate. It is important to point out though that these are prompts according to the definition of Dunn (2021). Robson and McCartan (2016) would classify these stylistic devices as probes. Prompts according to them suggest a range of possible answers the interviewer expects from the participant, are a part of the interview record, and must be consistently used across all participants. These kinds of prompts were also included in the interview schedule in the form of small introductions to the theme sections to set the context for the interviewees and keep the interview within the ranges of climate change and blue carbon policy. Lastly, there is conflicting guidance on the type of question words that should be used. Newing (2011) recommends avoiding questions that start with 'what', 'how', and 'why' if in-depth answers are wanted, while Secor (2010) only discourages 'why' questions since they can be seen as challenging (i.e., testing knowledge or participants may feel like they are being asked to justify themselves) and recommends that interview questions should be phrased as 'what' and 'how' questions since they are more productive. In this study, the advice by Secor (2010) was followed; 'what' and 'how' questions were included in this study, while 'why' questions were avoided. Newing's (2011) observation that these types of questions may lead to quick answers with lacking depth was not confirmed.

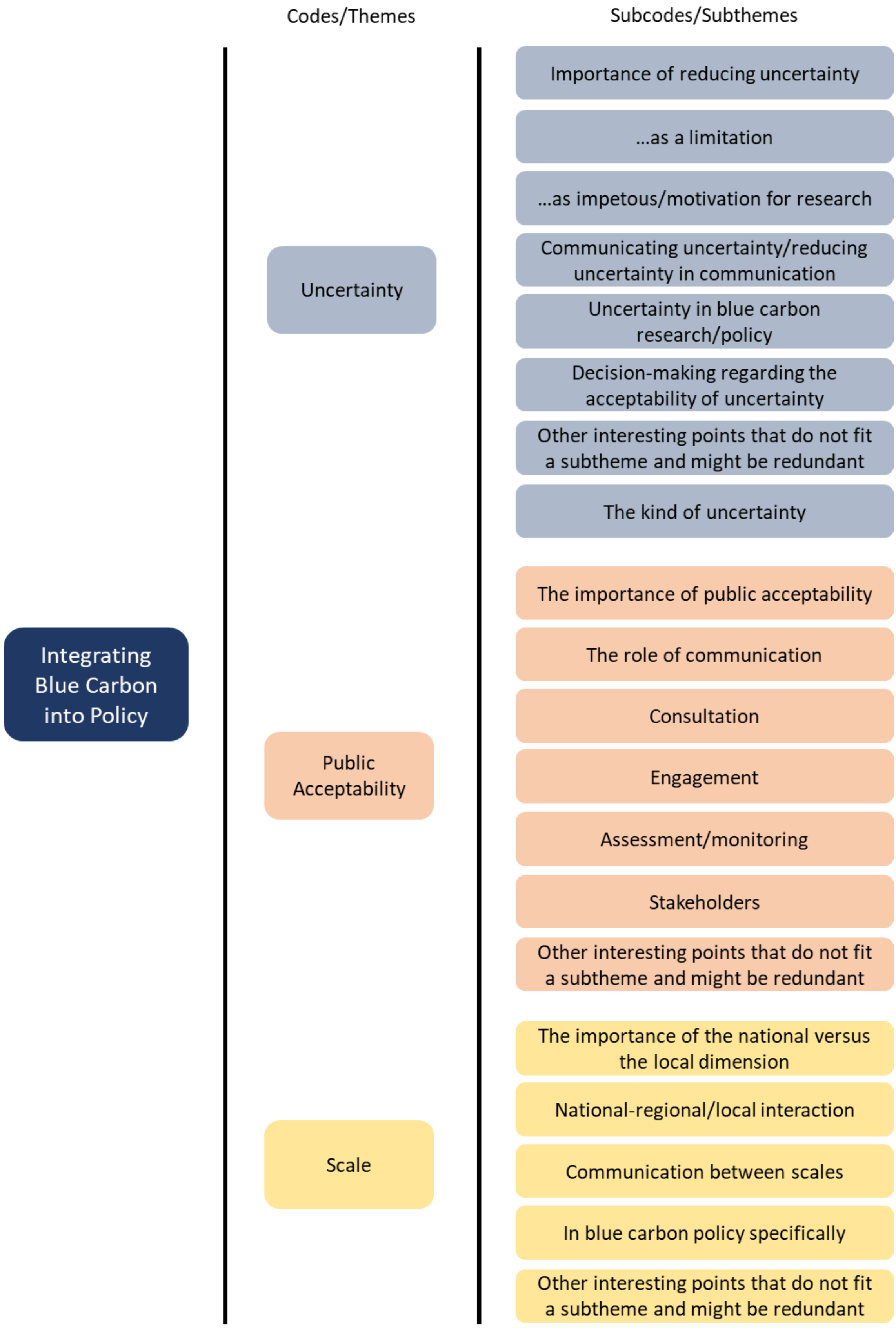
4.3.4 Interview Conduct

The interviews took place between May and September 2021 and were conducted via video calls. A main reason for this was the ongoing Covid-19 pandemic that made it impossible to conduct interviews in person. Using video calls can produce unique challenges but also advantages (Dunn, 2021). Technological failures can disrupt the interview, which was the case during one interview, but the issue could be resolved by switching spontaneously to a phone interview. Participants were made aware that they were being recorded, and this was set out as an option in consent forms that were sent and signed by the participants before the interview was conducted. Moreover, if participants had consented to being recorded, the researcher notified them once the recording had started and once it had been turned off again. Only one participant opted out of being recorded but alternatively consented to the transcription option that had just been introduced by Microsoft Teams and produces a fairly accurate transcription of the video call. The advantages of video calls include the ability to reach participants that may otherwise be out of reach due to mobility issues and the saving of resources in terms of environmental and financial resources as well as the researcher's time. Overall, Dunn (2021) reported that a majority of video call reviewers concluded that this medium performs as well as face to face interviews. The researcher can confirm that the video calls worked well in the context of this study, which may also be due to the fact that by May 2021 video calls had become mainstream in professional and personal communication due to the worldwide pandemic. The interviews were between 45 and 90 minutes long; mostly they lasted around 60 minutes with the 90-minute interview being an exception. All participants were asked the questions from the interview script with some questions or prompts added in when clarification of main points was necessary or to elicit more detailed responses. Although most of the interviews were recorded, handwritten notes were taken in case of technological failures.

4.3.5 Interview Analysis

The first step in the interview analysis was to create transcriptions from the interview recording. During this process, filler words such as ‘mhm’ and ‘ahm’ as well as double words were removed if they signified the thinking process of formulating the sentence rather than emphasising a point. The decision to ‘clean up’ the transcripts in this way was made since the interviews were focussed on the professional expertise of participants; the ethnographic moment could thus be neglected (Dunn, 2021). Once this step was completed, the software NVivo was used to code the data.

The purposes of coding are data reduction by distilling key themes, organising the data, and data exploration (Cope, 2021). Coding should never be the end itself but rather the means to reach an end; thus, coding should always have a purpose (Richards, 2015). In this study the purpose was to detect patterns in the data across the different participants and to enable finding connections between the identified themes. A mixed approach was taken during the coding and data analysis. Initial codes were deduced from the different themes within the interviews; within these themes, an inductive or data-driven approach was taken and subcodes applied according to the themes that emerged. Coding across the themes did occur on a few occasions though to accommodate if relevant information for one theme came up during the questions for another theme. The resultant coding tree is presented in Figure 4.2.


Codes/Themes
Subcodes/Subthemes
Integrating Blue Carbon into Policy
Uncertainty
Importance of reducing uncertainty
...as a limitation
...as impetous/motivation for research
Communicating uncertainty/reducing uncertainty in communication
Uncertainty in blue carbon research/policy
Decision-making regarding the acceptability of uncertainty
Other interesting points that do not fit a subtheme and might be redundant
The kind of uncertainty
Public Acceptability
The importance of public acceptability
The role of communication
Consultation
Engagement
Assessment/monitoring
Stakeholders
Other interesting points that do not fit a subtheme and might be redundant
Scale
The importance of the national versus the local dimension
National-regional/local interaction
Communication between scales
In blue carbon policy specifically
Other interesting points that do not fit a subtheme and might be redundant

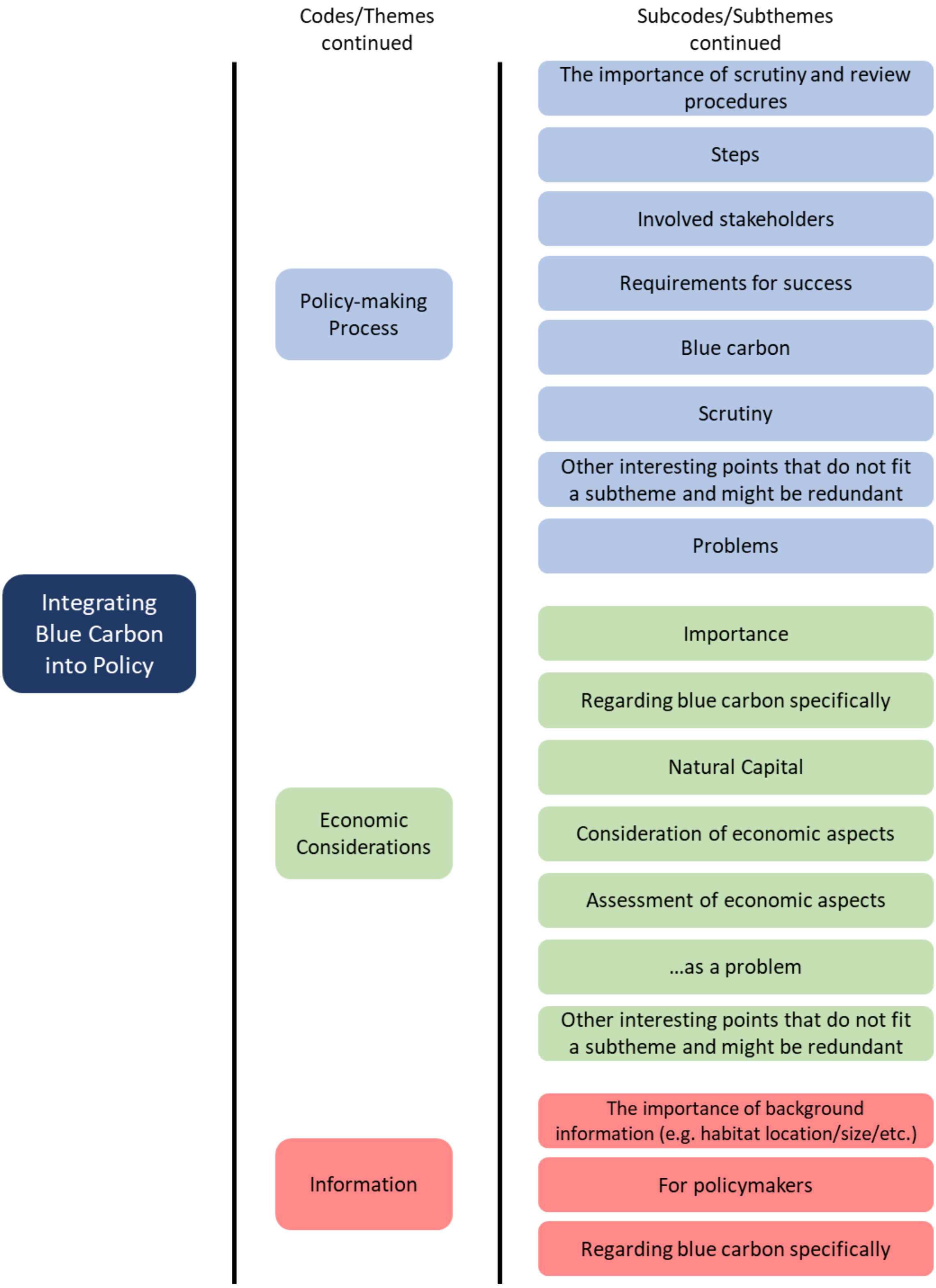


Figure 4.2: NVivo coding tree.

Overall, the approach that was taken has many similarities to what Robson and McCartan (2016) term the thematic coding approach. In this approach parts or all of the data are coded and labelled and grouped together in themes according to the labels. These codes and themes can be determined deductively from previous research or inductively from the data itself. The themes then form the foundation for further analysis and interpretation. This can take a purely descriptive or exploratory form or be used within a theoretical framework. The approach takes several phases: (i) familiarising oneself with the data, which can take the shape of transcribing the data and noting down initial ideas; (ii) generating initial codes to organise the data and enable identifying themes; (iii) identifying themes where the initial codes are sorted into themes or become themes in their own right; (iv) constructing thematic networks in which a thematic map of the analysis is constructed; and (v) integration and interpretation, which explores within and across the themes (Robson and McCartan, 2016). This study followed phase one and two, and phases three to five to a lesser extent. During phase three, most subcodes became subthemes in their own right instead of grouping them together. One exception was the subtheme ‘stakeholders’, which appeared several times under different themes and was aggregated under one theme due to extensive similarities. Due to the mixed deductive and inductive approach, the networks of phase four were derived from the themes and identified subthemes instead of taking a fully inductive approach of building them up from groups of themes. The coding can thus be described as taking a funnel shape, starting broad and narrowing down. This coding informed the structure of the results section of this chapter by helping to determine the implied importance of the subthemes through the frequency of their appearance; in the more cases they appeared, the higher their implied importance as shown in the list order of the coding tree (Figure 4.2). The exploration and interpretation within the themes and subthemes (i.e., phase five) is conducted in the results section of this chapter, and the exploration and interpretation across the themes is conducted in the discussion and conclusion section.

4.4 Results

Overall, the interview data collection produced eight interviews between 45 minutes and 90 minutes long. As described in section 4.3.2, a question on importance with a scale was included for each of the six overarching interview question themes. However, it needs to be clarified that the score on this scale can only give an indication of the importance of a theme and is not a definitive ranking. Nevertheless, the results from the scale make it possible to conclude whether a theme was important or not important. Overall, it is possible to conclude with high confidence that all of the above identified overarching themes are important for shaping blue carbon policy since ‘not important’ was never selected. Moreover, two additional themes that may influence blue carbon policy emerged through the coding approach. However, more research into these themes is necessary to confirm their importance. The question at the end of the interview regarding any further points participants wanted to raise did not reveal additional major themes that had been missed; rather, additional points that were made were more supplementary, which provides confidence that no major themes that shape blue carbon policy were missed. The reminder of this results section is divided into subchapters to present the data participants provided, reflecting the overarching themes and sub-themes that were identified using a coding approach in NVivo, as described in section 4.3.5.

4.4.1 Uncertainty in Scientific Evidence

There was no absolute consensus about the importance of addressing and reducing uncertainty in evidence that supports a policy. As highlighted in section 2.5.5, one of the major uncertainties in Scottish policy is the stock of organic carbon of Scottish saltmarshes. Other issues include the behaviour and fluxes of dissolved organic carbon under different climate scenarios (Codden *et al.*, 2022). However, there was a broad consensus that uncertainty does not have to impede or even stop policy development. Some participants highlighted that it is somewhat important but that there are other factors that influence policy making, such as stakeholder pressure, that can be more determinative and that the importance of reducing uncertainty depends on the unique situation and the consequences of a specific policy; for example, for non-contentious policies or those that affect only relatively few people, reducing uncertainty was not as important as for those policies that impact livelihoods or require large investments. It was also pointed out that the importance of uncertainty depends on the political climate and the ministerial willingness to take risks. Furthermore, P4 stressed that uncertainty should not prevent taking action since "it's more important to take action in relation to the worst-case scenario" to avoid being unprepared. Several respondents concurred that uncertainty is inherent in policy making and that it would not be productive to spend too much effort on trying to reduce it. P8 summarised that "Obviously, we have to recognize the value of the evidence base, but if we wait indefinitely for perfect science to achieve absolute values, we'll never get the results we need" and that waiting for a maximum amount of evidence would just stall policy development. However, one participant cautioned that uncertainty can also be instrumentalised by interest-groups to keep the status quo and reported that they observed this particularly with climate change policies, so in that sense reducing uncertainty can become very important for achieving a policy change. This is a very interesting observation since climate change policies frequently fall into the category of policies that require a large investment, which were highlighted as those policies where reducing uncertainty in scientific evidence has a higher importance, but at the same time they also fall into the category of policies that require action towards the worst-case scenario to avoid being unprepared, effectively creating a conundrum.

4.4.1.1 Uncertainty as a Limitation for Policy

As already indicated in the previous section, uncertainty can become a limitation for policy. P3 further elaborated on his point that uncertainty can be instrumentalised to keep the status quo. Uncertainty has become a problematic facet in policy making, highlighted by the dominant role of the sciences and of scientific evidence as support for policy. Yet it was recognised that all knowledge about an aspect should flow into policy, so instead of the current bias towards science and quantification, knowledge generated by the humanities and social science, for example, needs to get more attention. Other participants agreed that there is a risk of uncertainty being inhibiting due to reluctance to allocate monetary resources, change policy, or put restrictions in place when evidence cannot prove with full clarity that there is a problem that needs to be addressed or if there is no full evidence for a "critical pinch point in a policy development" (P8). Yet, P8 qualified that this does not apply to the bigger picture of the climate change scenarios and another participant surmised that uncertainty is not that inhibiting anymore in the context of climate change since there is now largely agreement regarding its causes and that action needs to be taken. This seems to contradict the finding of the previous section; however, climate change is a dynamic field with a high potential of overlap between developments. P5's statement that "we can work with it [uncertainty] by using other functions of the environment to guide our thinking," indicated agreement with P8's assessment presented above that uncertainty is not limiting in all aspects of climate change policy. Further, it was highlighted that there can be difficulties in explaining uncertainty in scientific evidence at the political level, but that it is possible to convey to people the significance of results, which is made easier now that there are good examples, such as sea level rise and its impact on communities, where no effective measures were put in place in the past and that are now causing significant cost and damage in the long term.

There was no consensus in terms of at which stage of the policy development process uncertainty is most detrimental. It was pointed out that uncertainty can be detrimental at the start of the development process when stakeholders discuss uncertainties and their different standpoints on them and the policy, while other participants specified that it is most detrimental at the end of the process if those uncertainties from the start become embedded in the policy and cause contention later on or when the final decision on the policy is made and that it is less likely to go through if confidence in it is lacking due to uncertainty in the evidence base. P2 argued that if high uncertainty around the evidence is a problem for the policy, then it would be an issue throughout the whole policy development process. It would be the most detrimental, however, during stakeholder engagement since it can cause a loss of trust in government, which could then spread to other developments in the same policy area or even wider areas the same stakeholders have interests in. Parliamentary scrutiny was named as an example for this. Accordingly, if policymakers are trying to push through a policy with high uncertainty in its evidence base, it might make parliament more cautious of other decisions in this area and cause a long-term loss of trust in government. This could then lead to parliament scrutinising and being doubtful of other decisions made by government that are based on good evidence.

4.4.1.2 The Role of Communication

Based on the statements of all participants, it is evident that communicating uncertainty is important. It is important to be transparent regarding uncertainty towards politicians and the public and communicate confidence levels. It should be presented as part of the evidence to the regulator who then has to make the assessment whether the uncertainty is acceptable. P8 cautioned though that there is also a risk of 'over-caveating' regarding the uncertainty and that to avoid this, clear messaging is important. It should be made clear that there is uncertainty but that the evidence base is still of sufficient quality to support the worst-case and best-case scenarios presented in the advice that is provided. In addition, it is important to communicate clearly to what degree there is uncertainty and why it is there. However, scientists are not always good at communicating risk and uncertainty, so standardising communication regarding uncertainty to ensure that people have a common understanding is important as well. The IPCC developed a system of standard terminology to describe uncertainty (i.e., a likelihood scale based on probability of a certain outcome and statements for confidence in evidence) to ensure consistency in communication, which Marine Scotland started to incorporate (IPCC, 2010). During engagement with stakeholders, communication is a main point of importance since uncertainties can cause difficulties in contentious areas where there could be disagreement among stakeholders, which makes it important to include it in the discussion until a consensus is reached.

4.4.1.3 Decision-making on the Acceptability of Uncertainty

There are several layers of acceptability; the acceptability of the uncertainty itself and the acceptability of the risk to make a decision based on evidence that has uncertainty to it. Neither of these decisions are the responsibility of the policymaker though. The assessment of acceptability of uncertainty in evidence is the responsibility of experts within the authorities to whom advice is provided, while the acceptability of risk stemming from uncertainty is generally with the politicians, such as Ministers. It became quite clear during the interviews that regulators such as Marine Scotland, SEPA, or the local authority, decide on whether uncertainty is acceptable and that a variety of factors can play a role in this decision. Respondents from NatureScot provide the evidence or advice to the regulators to equip them with the knowledge, to be used in assessments outlined in legislation which must be complied with. In the case of advice to Scottish government, this could be a ministerial decision on whether to implement management of a habitat. Thus, this decision also depends on how much risk a Minister is willing to take, which can also be influenced by political climate, views of an impacted community, or the sector that is going to be impacted.

4.4.1.4 As an Impetus for Research

Uncertainty can be an impetuous for research as well. Outputs of the SBCF, for example, are research that is presented to policymakers and can already include recommendations for further research; for example, the driving factors in saltmarsh loss. Policymakers will review the outputs and then make a decision whether further research is needed or if the answers gained from the piece of evidence are sufficient. Details on what information is needed on blue carbon are presented in section 4.4.5.2. Another way that uncertainty can lead to further research is the implementation of pilot studies. P2 described this as a tool that is suitable when there is a moderate amount of uncertainty to help get a better idea of the effects a policy would have. Again, the importance of the context was highlighted and that the size and scale of the policy was important. Particularly for a large-scale policy, it might be preferable to gauge the effect at a smaller scale first. From the NatureScot perspective, it was mentioned that refining the science is a part of their work and that research needs to be targeted where uncertainty is greatest.

4.4.1.5 Uncertainties in Blue Carbon Scientific Evidence

It was highlighted in the interviews that there is still a significant amount of uncertainty regarding the blue carbon resource in Scotland itself and the pressures on this resource. Initial assessments were very high level and for some blue carbon habitats, such as mearl beds and oyster beds, only general point locations of the resource may be available; data on extent and condition of these habitats is incomplete. Moreover, carbon sequestration and storage rates are often still based on findings from species that are not native to Scotland, and it is important to close this gap since these rates may not be applicable to the Scottish context. Regarding saltmarshes and mud flats specifically, it was reported that there is still uncertainty about the carbon storage mechanism, how and where carbon is stored. Furthermore, it was highlighted that saltmarsh cannot be considered a uniform resource since they have varying soil compositions according to their location and the surrounding landscape, which adds to the uncertainty regarding the carbon stored in this habitat in Scotland (c/f Chapter 1 and 2). These uncertainties regarding the Scottish blue carbon resource have impacts on policy. P7 described that a policy for safeguarding buffer zones for blue carbon stores was included in the pre-consultation draft of the Clyde RMP but was removed due to the uncertainty regarding the importance of these buffer zones which made it difficult to enforce the policy or make a judgement against it.

Regarding the pressures on blue carbon habitats, there are still uncertainties how they interact with habitats' ability to sequester and store carbon. In the context of saltmarshes, it is not entirely understood yet what effect grazing on saltmarshes has (P4; also see subchapter 1.3.2), and what impact climate change will have on the ability to sequester and store carbon. Climate change may also affect biogenic habitats through warming seas and acidification and there is uncertainty around the question whether such habitats will continue to function as a blue carbon habitat. A very present question in Scottish policy making is how much of a pressure different types of fishing gear and trawling are on the carbon stored in marine sediments (P2).

4.4.2 Public Acceptability

Responses to the importance of public acceptability for policy making were quite nuanced ranging from it being very important to just somewhat important. Finding socially acceptable policies often entails dealing with the variety of benefits which saltmarsh habitats provide for humanity and the trade-offs or winners and losers between them, including: providing fish nursery habitats, provision of habitat for natural biodiversity, supporting health, water quality regulation, recreation, coastal protection and agricultural food production such as livestock grazing (McKinley *et al.,* 2020). P2 pointed out that "as a government, we have to accept that some policies that need to be put forward for the benefit of different groups might not be publicly palatable. And that that's just something that, as a government we have to manage" and that public perception of a policy is important but cannot be the end goal. P3 commented that public acceptability is vital for governments to be re-elected and that the importance of it may be connected to the position in the electoral cycle with less importance and space to make bold policies at the beginning and more importance and caution regarding public acceptability at the end of such a cycle. Further insights that were provided regarding climate change policy concur with P2's point in that some measures that are disadvantageous for a group of people may have to be put in place to reach set targets, but also raise the point that climate change is an area where public acceptability has transformed, mainly in the last decade. For example, "it's not a coincidence that pretty much all the party manifestos in the Scottish election talked about marine conservation in one way or another and climate change" (P4). Policy development in a certain area is not usually driven by the public input, but rather by the identification of a need or challenge through assessments and regional planning, making the consideration of public acceptability secondary to this need. Considering these different insights, it is thus reasonable to argue that climate change policy is a special case where the identification of a need or challenge overlaps considerably with public input. Nevertheless, public buy-in to a policy can be very important regarding the public's motivation to act according to a policy. Acceptance influences action. Therefore, it is important to demonstrate why policies are in place. None of the arguments brought forward in this section seem conflicting, yet they cover a wide range that highlight that there are many facets to public acceptability.

4.4.2.1 The Role of Communication

The participants concurred that communication takes an important and central role for public acceptability with one participant expressing that it is of increasing significance. The notion that this communication should be carried out not just by the government but by all involved actors, such as academics and the advising agencies (i.e., NatureScot), and through a variety of channels such as websites, social media, traditional press, and project officers working on the ground was another prevalent theme across participants. Two participants further elaborated that scientific papers are not a fit medium to communicate with the public due to the possible delay caused by the publishing process and the need to translate key findings into media that are easy to understand for non-academics. The involvement of academics and agencies that are seen as independent from government and not as connected to politicians as government, was raised as a factor that could increase the public's trust in policies. Communication could be used to explain more complex issues that are not usually included in mainstream media, such as regulating and supporting ES. The participant who raised this, elaborated that this would really help "to illustrate the value of certain policies and the impact that they can have if they're protecting those services or looking to […] safeguard" them (P7). This assessment is supported by a second participant that highlighted that it is of critical importance for communication to "translate the policy in a way that makes it relevant and understandable to people" (P8). Marketing style assessments to find out how people respond to certain messaging are one strategy to achieve this. The example the participant used concerned the messaging around the central goal of reaching net-zero emissions. The catchphrase that was eventually decided on is "Let's do Net Zero" (Scottish Government, 2022). A further aspect that was touched upon by one participant was that good communication also offers benefits for the initiator of the communication; in this case the participant received valuable feedback from a community regarding their stance on adaptation which is an important pillar of Scotland's response to climate change.

4.4.2.2 Engagement and Consultation

During the interview, it became clear that there are various ways in which engagement with the public takes place during the policy development process. In addition to public consultation on policy drafts, engagement with stakeholders can already start before a first draft is developed. Policymakers may receive interesting evidence and start engaging with stakeholders as they work on a new draft. Engaging with stakeholders early in the process has the advantage that stakeholder views can be taken into account early and help with the development process. Even if it is not possible to adjust according to stakeholder feedback during early engagement, it is still valuable to know their concerns and be able to acknowledge them and communicate sympathetically and effectively, which can have an effect on the acceptability of the policy or management measure for these stakeholders. Early engagement with communities by researchers to communicate scientific findings that affect these communities before publication in a peer-reviewed journal can also be mutually beneficial. It can produce valuable feedback and, in the example P5 shared, also provided "a feel from the community about how far they would prepare to go in adapting".

Community knowledge about their environment can be more valuable and accurate than the scientific literature in certain cases.

As previously referred to, in addition to these forms of informal engagement, the government is also committed to formal public consultation. The NMP, MPA designations, or the future fishery strategy are examples where this approach was carried out. The results of these consultations are also taken into account by advising agencies such as NatureScot who consider them and may adjust their advice accordingly. Public consultations are a wide opportunity for everyone to contribute feedback to a development, although this opportunity to comment on a strategy or proposed legislation is usually taken up by larger organisations, such as NGOs rather than individuals. Public consultations are also a further opportunity for policymakers and the government to get an idea of public perception. Regarding blue carbon strategy and implementation, consultations could provide information on communities' support for specific management measures. As far as the interview respondents were aware there were no formal monitoring mechanisms in place to assess public perceptions on blue carbon, except keeping an eye on media.

4.4.3 Procedural Validity – The Policy-making Process

In this interview section, the participants were asked about the steps of the policy making process and the importance of review procedures during that process. It is important to point out that this question about the importance of review procedures was understood differently by different respondents. It was intended as asking about the importance of scrutiny but was understood by some respondents as reviewing the policy, as is done in a three-year cycle for the NMP for example. However, the differing understanding of the question also opened up an additional valuable point that may have been missed otherwise. When considering results of the importance rating scale, according to these two ways of understanding the question, it stands out that the two of them are the highest ratings on the scale of all sections. (Although this rating has a lower confidence than the ratings for the other sections since a lower number of participants engaged in the rating for each understanding of the question, it is still a good indication that both scrutiny and regular review of the policy are two highly important processes).

Scrutiny was highlighted as "essential" (P2) and the review as very important to ensure that a policy keeps reflecting and addressing the character of the problem it was designed for. P3 also caveated though that review processes are probably given less attention than should be their due since there is a tendency to aspire to reach certain targets that often only address symptoms of a problem instead of its causes. An example for this in nature conservation would be targets of having a certain number of protected areas and losing track of the original conservation challenge that was supposed to be addressed. Moreover, even if there is a formal review cycle in place for a policy, the review process "is tempered significantly by the degree of public interest for or against a policy and that will largely determine how any review is progressed" (P3) The review can take a light touch approach that 'ticks the box', or it can be much more thorough and interrogate issues anew. A regular review provides the opportunity to address any aspects that might have been missed previously. Additionally, reviewing and engaging in adaptive management are also important due to the uncertainty inherent in some evidence. Once improved evidence is available, it is important to be able to amend policy and potentially tighten it. For example, the UK climate projections (UKCP) were updated in 2018 to provide improved tools to help decision-makers assess their risk exposure to climate. However, these updates are only helpful if there is the possibility of amending or tightening the policy that addresses climate risk or impacts. A review that recently took place was the second review of the NMP; it was not revised after the first review but after the second review, the advice is to produce a revised plan, due to changed circumstances that make the policies it contains no longer fit for practice. Climate change causes major challenges for the environment and the expectation is that features will move or disappear entirely, which requires flexibility in management and protection of these features, and regular review of the evidence base.

4.4.3.1 Steps of the Scottish Process

The different steps of the policy development process vary slightly depending on the role of the participants and their contribution. This signifies that rather than there being one uniform policy process, there are several processes that interlink as part of a wider policy network. The more science-focussed participant of Marine Scotland described sharing and providing evidence as a first step, which can take a formal or informal shape; it ranges from sending an informal email to officially notifying Ministers and special advisors when it is in the form of a scientific publication. Scotland's marine assessment, for example, was a formal piece of evidence since it was an underlying requirement of the NMP review; it included blue carbon and recognised it as an opportunity for climate change mitigation (Marine Scotland, 2020). A step following the communication of evidence can then be that special advisors or policy colleagues return with clarificatory questions.

From the perspective of a policymaker, a first step would be to draft a plan of what should be done based on the evidence that was received. As mentioned in section 4.4.2.2 on engagement and consultation, informal engagement with trusted stakeholders can already take place at this stage, particularly if it is not clear how the public and stakeholders might react to a certain idea. Whether a proposal then goes through the entire hierarchy within government and ultimately to Ministers for their approval depends on the proposal that needs to be decided on. If it is just a small proposal with little cost involved, it would not have to be run past Ministers but could be decided lower on the hierarchy chain. However, if there is a chance that the proposal is very sensitive for some stakeholders or that there may be major (potentially negative) media uptake, the idea should have senior or even ministerial approval. Before a policy document is published, it needs to be presented to Parliament for scrutiny, which can include Parliament (or certain parliamentary committees) calling for evidence or a general parliamentary debate. At the end, Parliament would then notify the Ministers with feedback and potentially recommendations for action. After this process, a final document would be drawn up and published. There is thus an interplay of scrutiny internal to Scottish government and external during the policy development process. The records of the external scrutiny by Parliament are published on their website[19] and are available to the public, which adds a layer of accountability.

The development of the Clyde RMP provides a good example of the policymaking process. The starting process for the Clyde plan was an assessment of the area. In this assessment the importance of saltmarshes for carbon storage and for coastal protection was acknowledged (Mills *et al.*, 2017). From this assessment followed the identification of workstreams and the decision in which areas policy needed to be developed and through which mechanisms. In the next stage, drafts were developed and circulated for feedback and the policies had to be put together into the plan. This version of the plan then went out for pre-consultation. The pre-consultation plan explicitly acknowledges saltmarshes and other blue carbon habitats as carbon stores and that they need to be protected. Moreover, it also acknowledges the benefits saltmarshes provide for coastal protection. Objective CC2 specifies that natural carbon sinks and their associated benefits need to be maintained and where possible enhanced; additionally, Policy CC2 stipulates that developments or activities will be supported if they can demonstrate that they will avoid damage or where possible enhance carbon sinks. Furthermore, it stipulates the safeguarding of natural assets and the necessity to ensure that they are able to adapt to climate change, which includes providing space for these habitats further inland. Natural assets are to be given precedent over hard-engineered coastal flood and storm damage alleviation structures, unless it is not possible to use these natural assets (Clyde Marine Planning Partnership, 2018). This was followed by a review of the plan taking into account the collected feedback of the pre-consultation stage, creating an iterative process. At the time of the interview, the plan was being prepared to go through statutory consultation after which a final amendment will be made before it is submitted for approval from Ministers.

From the perspective of scientific advisors of NatureScot, the process would be to look at the evidence and then agree amongst the specialists within the teams what advice should be provided

[19] https://digitalpublications.parliament.scot/

to regulators or other appropriate parties. The advice with the evidence base would then also be published. Scientific advisors can also be involved in this process both on the government side, through working groups that lead to the drafting of policy and consultations on policy drafts, and on the Parliamentary side when called on for evidence. Their involvement can also take a more informal form through regular contact and exchange of information with local authorities and regulating organisations such as SEPA by causing a "sort of background level of trickling of science into policy and feedback from policy to science" (P5). In Scotland, this working relationship between scientific advisors and government can be quite close, which may be partly due to the small size of the academic community that promotes more direct input. P4 reported that from their experience, the scientific input their organisation provides is taken seriously and into consideration and that while other factors such as impact on communities and economic sectors also have an influence, the final policy is oftentimes pretty close to the scientific advice that was provided.

As already mentioned, the policy process can vary depending on the nature of the policy. Yet, the process is frequently described in technical literature as a cycle that starts with a problem, then evidence is developed, a policy response formulated and implemented, the impact of this response is reviewed, and if necessary, the policy is altered and so on. At the centre of this cycle is the science. P3 maintained that in reality, this is not the case but that the process is rather a messy interaction of all these steps in the cycle that is much more dynamic than a linear or cyclical progression. They also criticised that it is sometimes dominated by science to the detriment of other evidence that can be brought forward as already mentioned in section 4.4.1.1. Policy development is not always a straightforward journey due to the variety of factors that need to be taken into account. In addition to the factors already mentioned above, legal aspects also need to be considered. The progression previously described from development by government to scrutiny by Parliament can be reversed if Parliamentary committees decide to call an inquiry into an area that they think requires policy development. In this instance the committee in question calls witnesses, produces a report, and pushes government to develop a new policy. The steps can also be sped up significantly if an emergency arises that causes public pressure and outcry that requires a quick response. In this case features could be protected within a matter of months instead of requiring a two-year process. This does not mean that no care is taken in the response but rather that "all bend over backwards and get things done" (P4).

4.4.3.2 The Network of Stakeholders in Policymaking

There is a tightly interwoven network of stakeholders that is involved in the policy process. Within government there is a flow of information and questions both vertically up and down the hierarchy but also horizontally between colleagues in different government sections, for example the Marine Science Policy Unit, and Marine Scotland Science. The general public and their perception are important, but other players such as NGO's and lobbyists are also involved; particularly the former can also influence public perception. In some instances, the entities putting pressure on the government are not the same as those that would be affected and suffer a loss of income should the government give in to the pressure. For example, the RSPB (Royal Society for the Protection of Birds) is promoting a managed realignment scheme at the Inch of Ferryton, Inner Firth of Forth, with the argument that the value of this scheme to the people in the area is larger than the value of continued agricultural production. However, the displacement of agricultural production would affect local farmers who would lose productive land and hence potential income (Ceci, 2017; MacDonald *et al.*, 2017). Thus, the government is required to manage these conflicting stakeholder inputs and balance risks and benefits. Stakeholders also have varying access to members of government such as Ministers; big corporates and large social groups such as NGOs usually have an advantage in this regard. Depending on the policy area, there are different changing groupings of stakeholders that can be called on to provide advice or respond to calls for advice. These can be academics, academic forums, members from industry, or organisations such as NatureScot or MASTS (Marine Alliance for Science and Technology for Scotland). During public consultations, communities can get involved alongside individuals, big organisations or their representatives, agencies, and academia. On the national level more stakeholders would be expected to be involved than on the local level that has by default a smaller stakeholder group. The network of stakeholders is very dynamic and stakeholders that engaged initially in various sectors may have moved on or their representatives may have changed. This flux of representatives can also be challenging for other stakeholders, who can get frustrated with staff turnover of the involved organisations.

4.4.3.3 Requirements for Policy Success

The participants identified various elements that are important for the policy development process to be successful. A common theme was good engagement. This included enough time to facilitate appropriate engagement with key stakeholders to ensure they feel listened to and to then take the feedback into consideration and be transparent about it. Related to this, good communication, which was already highlighted as an important factor in dealing with uncertainty and public acceptability, and facilitation of discussions with stakeholders were raised as important for a successful process. Depending on the situation, this can mean having an external person facilitate the discussion to reach compromise in difficult situations. It was highlighted that the most successful policies or most effective management measures are those where relevant communities or sectors are involved in the development. Additionally, it is important to have the evidence base to be able to demonstrate why a policy or management measure is necessary. Enough time is also required to ensure that all the relevant parties can prepare and collate the best evidence to feed into the different stages of the policy development process; to provide the opportunity for people and organisations to provide evidence at the right points; and to be able to highlight if there are any issues or unforeseen consequences. Good engagement should also be regular and provide a routine exchange of information between relevant organisations and policymakers. Good engagement also goes hand in hand with good communication. This is also the case within government. It is important to believe in the proposal and communicate clearly with senior officials and Ministers. Getting everyone on board and to believe in the proposal is important for its success. An additional benefit would be to clearly demonstrate good benefits and value for of the resources that have to be spent on the policy; the benefits should outweigh any negative impacts.

4.4.3.4 Problems within the Current Policy Development Process

Problems within the policy development process can be categorised in a number of ways- structural and substantive aspects were highlighted in interview data. Structural difficulties include points of memory loss in the system; policymakers move between departments, so evidence and information that was sent may be lost during a handover of a position. Problems can also be in the nature of shortcomings with the substantive evidence base that is taken into consideration. The issue of socio-cultural context and that it should be taken into consideration more often was raised. Instead, economics (i.e., costs and outcomes) frequently guides decisions, losing track of the bigger picture. Similarly, P3 made a compelling argument regarding the evidence that is used to address an issue. They stated that

> good policy should reflect the character of the problem that […] it's seeking to solve, and most of the time the character of that problem is going to be, well, particularly for the environment or climate will be an interaction between various natural science factors that are operating at different scales and temporal cycles […], and a whole range of social factors, which are, interacting with those natural factors and good policy should recognize all of that and find the right place to intervene for kind of maximum effect. But the fact that most of the evidence is narrowly defined as science means that most of the time you've only got half the story in your evidence base (P3).

4.4.3.5 Blue Carbon and the Broader Policy Development Process

As illustrated in Figure 4.3, there are multiple ways in which blue carbon habitats can be incorporated into policy. Saltmarshes and seagrasses could be included in the UK GHG Inventory. However, this decision lies with BEIS, which is a UK government department; the Scottish government is engaging with BEIS to lobby for this inclusion. In the case that these habitats are included in the inventory, saltmarshes and seagrass would then have a place in the Climate Change Plan since they would count towards reaching GHG emission reduction targets. P2 thought that wider blue carbon habitats currently have "no chance" of being included in the GHG Inventory and thus would also not be included in climate change mitigation policy. P2 did not specify a reason for this assessment but as highlighted in the thesis introduction, the Climate Change Committee adopts a strict blue carbon definition that only includes saltmarsh and seagrass since they are habitats that can be managed for the carbon storage benefit (Kershaw *et al.*, 2022); this may be the underlying reason for P2's statement. However, this does not signify that other blue carbon habitats cannot be included in policy. There is space for these habitats in marine spatial management, which is under the control of Marine Scotland. Potential frameworks they could be included in are the NMP when it is revised, RMPs, the Blue Economy Action Plan, or MPAs. The possible blue carbon policy inclusion pathways are illustrated in Figure 4.3.

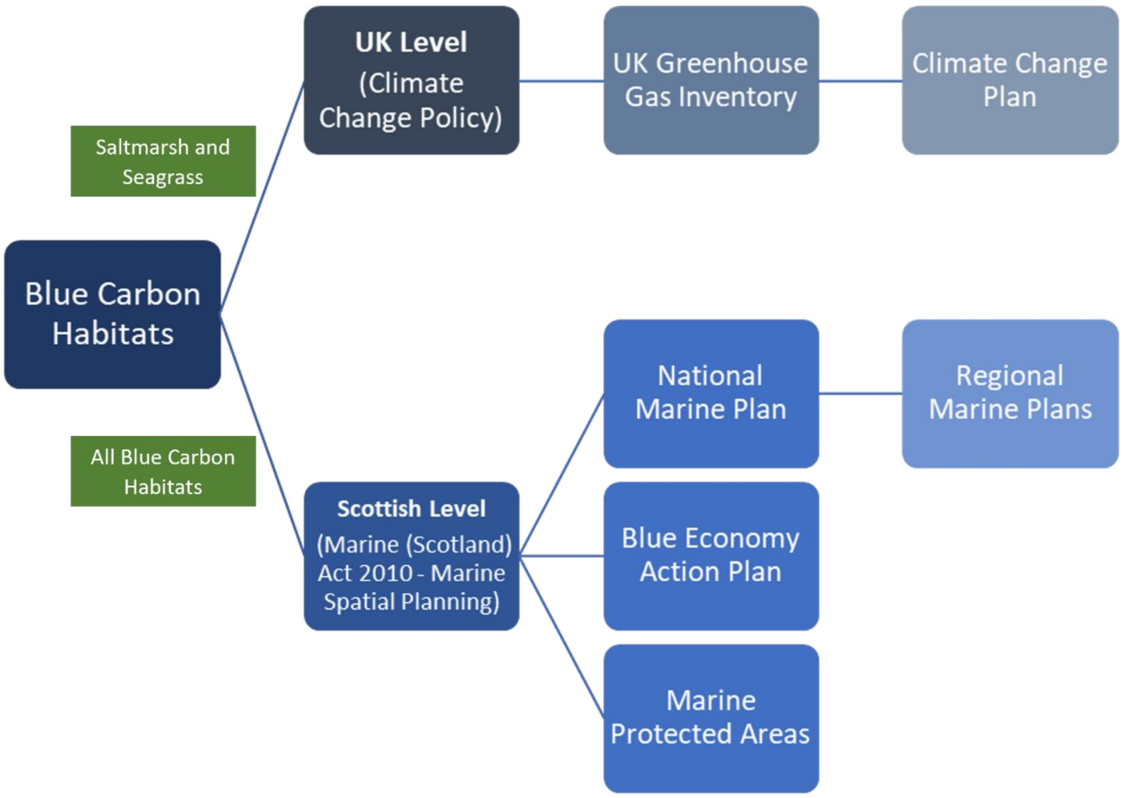


Figure 4.3: Blue carbon habitat inclusion pathways.

Attention was called to the fact that carbon sequestration and storage rates are easier to assess in terrestrial systems and that there are more established opportunities to include the carbon benefit in policy through agri-environment schemes that do not exist for the seas. The difference in how ownership works as well as issues with accessibility, evidence and, monitoring for marine systems compared to terrestrial systems may thus have an impact on the development of policy for blue carbon. However, one participant raised the interesting point that NbS should be about achieving multiple benefits. Since blue carbon habitats fall among this category, the focus should not just be on their contribution to climate change mitigation through carbon storage but also on societal benefits and their contribution to biodiversity. Including blue carbon habitats in marine spatial management frameworks may hence deliver more benefits than just including them in the Climate Change Plan.

Participants agreed that most policymaking is incremental, building or integrating it into existing frameworks; this can be achieved through the review processes. However, it was clarified that ideally, blue carbon is picked up as an important factor in the early stage of developing a plan, such as the assessment phase. This may be the case because of the importance of a holistic approach that is more likely to achieve sustainability; disregarding one benefit during the initial policy development could lead to policy that ends up negatively affecting this overlooked ES, which could be avoided by the holistic approach. This is still possible for the RMPs that are under development. The NMP, however, was published first in 2015 and does not presently include blue carbon, but might do so through the review process, drawing on an improved evidence base.

Overall, there should be no insurmountable barriers to incorporating blue carbon habitats into policy. While one participant cautioned that the inclusion of blue carbon into policy should not be rushed since the first objective should be to have a good understanding of blue carbon, how it was accumulated in blue carbon habitats, how it stays there and how it may escape from these habitats, a different participant pointed out that there is an agreement that more action needs to be taken to safeguard blue carbon habitats through planning policies and to identify the most damaging interactions and manage them, which is in line with the assessment made in the uncertainty section that incomplete information should not necessarily be a barrier to policy. It was also highlighted that in Scotland there is a close working relationship between the government and advising organisations as well as academia as exemplified in the context of blue carbon by the SBCF, and that science has a high influence in policymaking. New findings regarding blue carbon habitats can thus be easily communicated, which could further reduce barriers to policy inclusion.

4.4.4 Economic Feasibility

The importance of economic considerations was ranked quite high by the participants, between important and very important. Unfortunately, as previously mentioned in section 4.3.3, it was not possible to talk to economists from Scottish government. The interviewees still had interesting insights though. The point was made that given the current situation with the COVID-19 pandemic and the strain it put on the economy, economic considerations are probably quite high up on the agenda. P3 commented based on previous experience with working on the Sustainable

Development Commission, that the UK Treasury was an important body in policy development. For the Treasury two factors were important; how much a policy intervention would cost and what the public support was for it. This direct link may not be as important in the Scottish context, but it indicates the importance of economic considerations that come with policy development. In addition to the implementation costs of a policy, the effect on the economy is commonly required to be taken into account by government as well. A further point that was raised as to why economic considerations are important was that it can be helpful to be able to provide evaluations of measures to demonstrate their benefits; but at the same time, it is not possible to do that for everything. Not being able to assess something in a monetary way should thus also not be a barrier to interventions. Several participants agreed that considering the environment in economic terms is becoming increasingly important. P8 made the compelling argument that it is necessary to recognise that resources need to be used for the benefit of the whole of society, so economics cannot be detached from policy. It was clarified though that in the advisory role, economic considerations are not as important as they are for government who is the decision-maker. Unfortunately, it is not possible to provide more information on the economic considerations the government needs to take into account beyond implementation costs and the effect a policy may have on the environment. It is possible though to delve a bit deeper into the considerations of NatureScot as an advising agency. At the moment these considerations only extend to the 'balancing duty', which requires socioeconomic factors to be taken into account. And, while NatureScot is primarily focused on the natural heritage, it is important to consider impacts on stakeholders. However, it is still up to the regulator, which could be Scottish government, to undertake these social impact assessments. The current consideration of economic factors is thus quite indirect but is predicted to become more direct with the increasing relevance of natural capital and valuing the environment. The development of a Scottish Blue Economy Action Plan and related natural capital asset and risk registers are likely to reinforce the importance of this dimension (SEFARI, 2023).

4.4.4.1 Natural Capital Assessments in Policymaking

Natural Capital is still an emerging topic within policymaking and government and as such not all necessary tools are available yet to conduct holistic natural capital accounting. Thus, it is currently focussed on accounting for the carbon benefits of a habitat. In general, the natural capital approach is a promising tool to improve the valuation of ES that are provided by habitats and species and properly developed could be included in the next NMP. Previously marginalised ecosystems or specific ES can be taken into account in a more fundamental way if they are valued as part of the natural capital approach, allowing for an estimation of their worth to the economy. For example, it would be possible to assess "the cost that's saved if you allow a healthy saltmarsh to be maintained" (P4). Natural capital also provides a new dimension to assess trade-offs of different policies because it makes it possible to dig deeper into long-term trends and impacts on environmental assets. Scottish government are intent on expanding this work going forward and a wide application of the natural capital approach is expected for the future. It is predicted to become embedded within all aspects of the work regarding nature. This is very significant for blue carbon since it highlights the entirety of the benefits provided by these ecosystems and thus strengthens the argument for including blue carbon habitats in the wider MSP framework rather than just within the climate change policy framework. It hence represents a considerable opportunity for blue carbon habitat management and policy inclusion.

4.4.4.2 Assessing Economic Factors Relevant for Policy Development

As briefly mentioned previously, the assessment of economic factors relevant for policy development is carried out by the regulator but as it lies outside the remit of the participants that took part in the interviews, it is not possible to draw detailed information on this from the interview data. Government has a series of economic teams that make these evaluations and do monitoring required for good policymaking; in the case of Marine Scotland, it is the Marine Analytical Unit. And as highlighted previously, the form of the economic assessment that is carried out depends again on the context and the policy that is developed; a small policy would not require the same economic assessment as a big policy development. One of the tools that may be used for these assessments is cost-benefit analysis.

4.4.4.3 Problems Caused by a Focus on Economics

Two major problems of the current economics-focussed approach emerged from the interviews. The Dasgupta review (UK Government, 2021) was highlighted, and it was pointed out that economies need to be embedded more in nature and reflect wider social values and further that they currently do not serve the public good. This point is backed up by information P5 provided from the work they did with communities. They reported that people felt like their culture was being overlooked. P5 further expressed:

> public bodies and the government in particular should take cultural context into consideration much more than they currently do. We're too often guided by the economics of a situation without seeing the bigger picture. And it's interesting that the further you go away from the centres of governance, the less important the money becomes in how people guide their own lifestyles, and it's interesting that islands that are regarded by economists as basket cases[20] are the happiest places in Scotland (P5).

[20] Informal expression for describing a country/region that has a struggling economy (Collins Dictionary, 2022).

Similarly, employment is not the most important factor as perceived by some communities. During community appraisals in the Western Isles, it became apparent that the high-quality environment was a major factor for people's decision to live in the area, and that employment considerations may not be a priority for some. P5 recounted that this local feeling for the environment was unexpected and underestimated by policymakers and concluded that "the politicians are more driven by the employment angle of these peripheral areas than the local people are" (P5).

The second major problem of the economics focussed approach that emerged from the interview was the danger of over-investment in high carbon systems whether for avoiding carbon storage losses or increasing sequestration. With the costs of climate change, investing in mitigation potentials suggests a good cost-benefit ratio. However, with this focus comes the danger of creating carbon monocultures, which could have far-reaching consequences for the environment that are best expressed in P3's comment:

> creating monocultures for carbon […] will of course continue to erode biodiversity and set up a false polarization that you can either have climate policies or you can have nature policies, but you can't have both and actually we need both, it's a false choice to say that you can have one or the other. The use of the land inevitably affects the state of nature, it inevitably affects climate mitigation, and it inevitably affects the capacity of systems to respond to changes in climate which have already locked in, so the land use has to do all three of those things at the same time: mitigation, adaptation and state of nature. And the key to that is diversity (P3).

A diversification of investments and further development of the natural capital approach to encompass as many ES as possible is thus necessary.

4.4.4.4 Economic Considerations for Blue Carbon Habitats

Regarding blue carbon habitats, there was a strong consensus that the focus should not only be on the carbon value but that all benefits the ecosystems provide should be taken into account. Practically, blue carbon habitats are often framed around other benefits they provide, such as biodiversity and reducing coastal erosion and flooding, since there is no avenue yet to recognise them for their carbon storage benefit and as previously mentioned, only the carbon benefits of saltmarshes and seagrass are likely to be included in the UK GHG Inventory. Quite often, blue carbon habitats are designated as PMFs and thus fit into the conservation and MPA commitments, which demonstrates that blue carbon habitat management is not reliant on the inclusion in the GHG Inventory but can also be accomplished within wider MSP. P2 stressed that good policymaking should always consider wider benefits and that when it comes to the environment there is a twin challenge since the environment goes hand in hand with climate change creating the need to look at the two together. In other areas, there might be a scenario where it is possible to focus on one goal and discount other aspects, but this is not the case in the context of the environment. Moreover, it should not be forgotten that some of the blue carbon habitats' ES provide benefits for climate change adaptation, which makes it imperative to consider benefits beyond the carbon storage, so there are multiple stacked benefits that need to be protected. In terms of investment this is already happening, even though there are no formal codes or valuation tools yet, since companies and investors are already convinced of the benefits of these investments from a corporate social responsibility viewpoint.

4.4.5 Information Required for Blue Carbon Policy Inclusion

This section is concerned with the kind of information that is required for policy. Background information on habitats, such as size, location, and condition, was found to be important to very important. This kind of understanding of the habitat is very important to inform the policy regarding what is protected and how this is achieved. The importance of knowing the locational context of a habitat and the situational context (i.e., the kind of policy that is required; e.g., high-level, or more specific and regional) was also highlighted again. For individual habitats detailed information would be important information to the party that carries out the management of the habitat but that if it concerned a high-level policy, it would probably be less important to get high-detail information on individual habitats. A reason for this may be that high-level policies are expected to be overarching and to provide the general framework for regional policies that can take a more focussed approach and would thus also require more detailed information on specific habitats within the region.

4.4.5.1 Information for Policymakers

Two factors emerged on which the kind of information that needs to be provided to policymakers depends. The first factor is the temperament and preferred policy style of policymakers and Ministers themselves. Some prefer to receive only the high-level most important information while others wish to delve into the details to deepen their understanding of an issue. The second factor is the policy that is being drafted. Less detailed information may be required by policymakers to draft a high-level policy just setting out principles. However, if the purpose is to develop or carry out specific interventions or designate a particular site, it is necessary to know the details about the specific habitat.

Lack of information, such as the uncertainty in OC stock described in Chapter 2, can affect the ability to monitor the success of a policy according to a specific baseline. In some instances, though, this problem can be overcome by looking at proxies for monitoring. Fisheries were provided as an example; some habitats are known to be beneficial as juvenile fisheries nursery areas and if pressure on these habitats are removed, the success may be measurable in increased fish productivity. It is also helpful to have information that supports prioritisation based on the biggest overall value (i.e., the sum of all ES) a habitat provides for nature and people to maximise the benefits of the protection, since it is impossible to protect everything. P8 related this point back to the uncertainty that is inherent in scientific evidence and emphasised the importance of taking a balanced approach to gathering data and the necessity to accept practical limitations on gathering more data before making a policy decision.

Furthermore, an interesting point was also raised regarding the kind of information that should be considered. Instead of just taking the different aspects about a specific ecosystem into account, the context of the adjacent land and how the habitat fits into it should also be considered. This highlights once again the importance of context that has emerged numerous times before in the interviews; in this instance locational context in particular.

4.4.5.2 Beneficial Information on Blue Carbon Habitats

In general, the state of information is currently improving for coastal blue carbon habitats and is overall better than for their marine counterparts. However, there are still information gaps and improved knowledge in these areas would improve the inclusion of blue carbon habitats into policy. Figure 4.4 provides an overview of the information that would be beneficial for blue carbon policy inclusion. One participant observed a shift in information that is required on blue carbon habitats in the last two or three years (i.e., since 2018/2019), from basic habitat extent and condition to more detailed information on how the environmental mechanisms within the habitat function. They maintained that it is important to know this detail before deciding which or if any action should be taken. A second participant concurred with the importance of establishing the exact dynamics of how carbon is stored and additionally identified the need to determine the effect that human impacts have on blue carbon habitats, and how much activity or disturbance of these habitats leads to a release of the stored carbon. The importance of knowing the links between habitat condition and pressures and their thresholds for affecting the habitat condition and their ability to store carbon and provide other ES was stressed and that it is important to know which activities exactly are causing the release of carbon. Once this is clear, it is possible to investigate the economic implications of stopping certain high impact activities. Another participant highlighted the need to know more about the storage and sequestration rates of key blue carbon habitats, including saltmarshes and seagrass, and the role of sediments. There is also more to learn about sediment hotspots that store large amounts of carbon and managing pressures to prevent their disturbance and loss of this carbon to maximise the carbon storage. Further, it is currently still unclear what the flux and fate of the carbon resource of donor habitats such as kelp is; more information is needed on where the detritus settles.

Two participants also highlighted the importance of the ability to prioritise key blue carbon habitats. It was clarified that for policy to be developed, it is necessary to know which activities are the most impactful and would thus maximise the outcomes of the policy if they were managed. In line with this it was reported that it is not possible to tackle all blue carbon habitats and that a hierarchy of key ones with the highest relative contribution to the carbon ES is required to target these for reducing pressures or maximising the protection of carbon stores. Figure 4.4 summarises and categorises the information that would be beneficial for blue carbon policy inclusion.

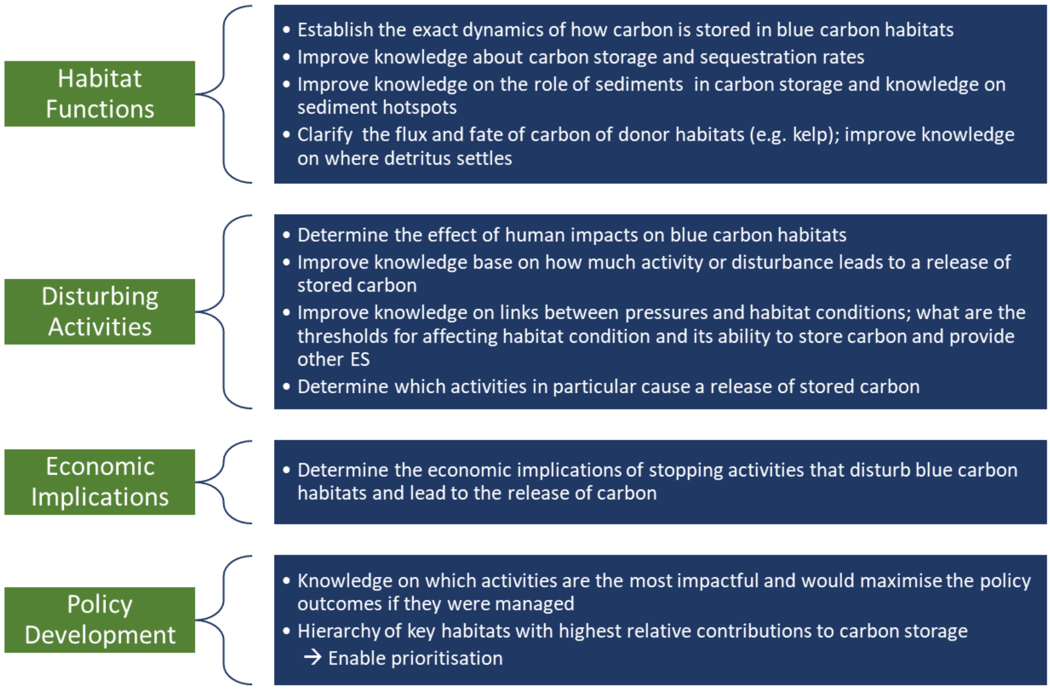


Figure 4.4: Information that would be beneficial for blue carbon inclusion into policy.

4.4.6 Scalability

When asked about the scale of a policy and the importance of the national dimension versus the regional dimension[21], the participants stressed the importance of a consistent overarching national position and policy approach. P2 stressed that from the government's perspective it does not make sense to implement local pilot projects without a national position to set the tone of the work that would be done locally. They stated that this national position is not required to be published in a major strategy document but can take a more informal form. Further reasons that were named in favour of a national policy framework that sets boundaries for regional policy included the importance of consistency in approaches and the potential of conflict within local areas that could be caused by diverging approaches. It was proposed to keep the national policy to high level

[21] 'Regional' refers in this chapter to sub-national Scottish regional policy rather than supranational regions and policy.

objectives and use it to establish boundaries. Local communities could then be encouraged to develop their own approaches that work for their local context within this framework. This would provide space for regional consideration within national policy. In the context of blue carbon this could mean setting guiding principles on the national level, such as objectives for maintaining habitat health and functions; local communities could then develop approaches or projects for individual habitats, such as local saltmarshes, that ensure that habitat health and function is maintained but also allows for grazing for example if this is a locally important factor.

4.4.6.1 Interaction of National-Regional Scales

The interaction of the national with the regional, does not just refer to the regional policy that needs to sit within the national policy. There is also an interaction of the national policy with the local implementation on the ground. Starting with a few pilot projects can help to get feedback how the implementation works before the approach is scaled up to more projects. Moreover, P3 cautioned that direct implementation of national policy can overlook the local complexities of situations, people, and context in favour of simple systems that serve the bureaucracy. This would also be detrimental for nature "because ultimately for nature you want different things doing different things in different places to get the degree of diversity that you need to see in terms of response to local situations, benefits to nature and benefits to people." (P3). An intermediary control such as a regional policy is thus needed to take these local complexities into account, particularly to address the coupled climate nature emergency[22]. This more localised approach also extends to the political-economic aspects since as P3 highlighted, the current political economy is part of the cause of the climate-nature emergency and cannot help to solve it in the form it takes in the present. Attention was called to the fact that there are now several Regional Marine Planning Partnerships that are working on implementing this intermediary level in the form of the RMPs. However, one participant also cautioned that while local initiatives should be supported, they also need to be managed carefully to ensure that they are undertaken in the correct location and that other competing activities are considered. It was also indicated that even from the regional to the local there is a certain degree of flexibility that allows for slight variation. This is the case due to the high possible variation in the habitats or species themselves; in some areas species and habitats may be doing very well, so management could be less restrictive while in others they are not doing too well and may need more stringent management.

4.4.6.2 The Importance of Cross-scale Interactions and Communication

It became clear during the interviews that communication is a very important factor when it comes to national and regional scale policy. It is particularly important to communicate whether certain policies, management or interventions have the desired effect, especially during pilot trials. If this

[22] The coupled climate nature emergency refers to the idea that nature and climate cannot be separated (P3).

is not the case, then it might be a matter of adjusting the local or regional policy, but it could also require an adjustment of the national policy. Additionally, authorities responsible for the local site also need to communicate with the local communities since each site at which activity takes place (e.g., habitat restoration activities) must go through due diligence and engage with the public. Ensuring early communication between the scales is thus an important factor.

4.4.6.3 Blue Carbon Across Policy Scales

In line with the points made on the importance of national and regional and local considerations, blue carbon policy on the local level needs to sit within the boundaries of the national plan. For blue carbon, the relevant policy documents would be the RMP and the NMP. One participant also emphasised though that while regional and local approaches need to be in line with the national plan, they can also decide to take a stronger or more detailed approach or prioritise a particular blue carbon resource. Particularly, the restoration and enhancement of blue carbon ecosystems, such as seagrass planting or native oyster restoration, is often led by community groups and should be supported. Some very important work that enhances understanding of blue carbon habitats can come from citizen science and communities, so it is important to keep people engaged for the purposes of improving evidence and also actioning implementation.

4.4.7 Other Aspects of Interest

There were four other aspects that did not fit neatly within any of the NVivo codes or subcodes and identified themes but that are still of interest. Particularly the first two, could be potential new themes, however, they would require further research to explore them in more detail and are thus at a lower confidence than the themes that were deducted from the literature. Figure 4.1 was updated with these new findings and is presented in its updated form below in Figure 4.5. The third aspect describes a scenario approach that would be relevant for the two emerging themes and the fourth aspect can be highlighted as a potential guiding principle for blue carbon policy. The use of the precautionary principle in the face of uncertainty was highlighted. The causal link between pressure and an activity is not always well established. Hence, under the Habitats Directive, measures must be put in place to avoid significant risk to protected features or species. The precautionary principle shifts the burden of proof; consequently, it needs to be proven that there is no impact. If there is a potential impact, precautions and mitigating measures need to be taken. Another tool that is important in the face of uncertainty is adaptive management. Adaptive management works in several stages and a cyclical fashion. Action is taken under uncertainty and monitoring put in place; dependent on the insights gained from monitoring, the approach is then adjusted, and monitoring continues. An eye should also be kept on practices elsewhere, such as the UK or Europe more widely since it can provide ideas for effective management or help avoid approaches or policies that are ineffective.

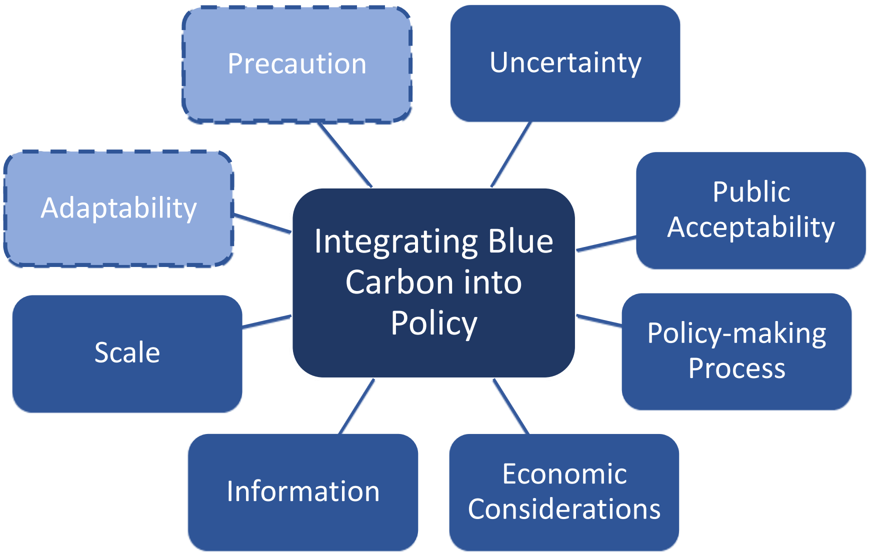


Figure 4.5: Updated themes that are likely to shape blue carbon policy.

Addressing uncertainty can also take the form of considering different perspectives and trade-offs of different policy options. Advice can be considered in worst-case and best-case scenarios; different policy options can then be developed for each of these scenarios and pros and cons compared.

The fourth interesting aspect that came up and has the potential to be a guiding principle for blue carbon policy, was protection instead of intervention. It is summarised and illustrated in Figure 4.6. Several participants shared the opinion that removing threats to NbS would be preferable and more beneficial than specific interventions for nature; if this approach were taken, it would only be necessary to remove pressure and let nature develop its own course. It was pointed out that it is not always clear if and what kind of intervention is the right thing to do. Regarding saltmarshes, it is natural for them to go through cycles of accretion and erosion. The question is then whether there should be intervention if erosion is taking place since it means losing some of the stored carbon even though it is part of the natural process. Interventions in one aspect of the environment often have effects on other systems as well, especially in the context of coastal habitats. The focus regarding saltmarshes should not just be on carbon but their multitude of other benefits should also be kept in mind. The more saltmarshes are allowed to operate naturally, the more future-proof these stretches of the coastline are. Future-proofing the coastline is important since the climate change impacts in Scotland will be concentrated on the coast. Natural England's expression of making room or space for nature was pointed out as a fitting way to describe what should happen. In this context, they also make the point that the focus needs to shift from safeguarding just habitat extent to safeguarding habitat structures and functions as is already included in favourable conservation status and the Habitats Directive. Keeping the environment in a good state or letting it recover is within society's interest since the environment is closely linked to wellbeing.

Figure 4.6: Promoting conservation rather than intervention as a potential guiding principle for blue carbon policy.

4.5 Discussion

4.5.1 Connections across the Themes

Identifying connections across the themes is an important step to understanding the policy process and the factors that influence it. A good understanding of the policy process is necessary to ensure successful science-policy integration. Without understanding the integration of all these themes, a low-quality policy could be the consequence due to a misunderstanding of relative importance or ordering of factors. During the interviews and their analysis, it became clear that the Scottish policy process is highly complex with many elements that are closely intertwined. Considering these connections is thus important.

One aspect that came up repeatedly and across all themes, with the exception of public acceptability, was the importance of context. In some cases, it was referred to the situational context, which exists on multiple scales. A (perhaps minority) view is that uncertainty is not very inhibiting anymore when it comes to climate change policy since there is now widespread agreement of its causes and that action needs to be taken. This is a quite high-level situational context with a global reach since climate change policy is a global issue and there are international agreements that need to be adhered to such as the 2015 Paris Agreement. On the other hand, the situational context also refers to more national-level concerns, such as the political climate or even the human factor within government exemplified by Ministers' varying willingness to take risks. All these factors together create a situational context that is important to blue carbon policy. Frequently, the importance of context referred to the policy context, such as the size and scale of the policy (i.e., whether it would be a large policy affecting a high number of people or a small policy only affecting a small number; whether it would be a national or regional policy). Particularly the last factor was identified as important in determining what kind of information policymakers need to develop policies. Overall, these factors also influence what form the economic assessment that is performed for the policy takes or how important reducing uncertainty in evidence is. Moreover, the required policy development steps and the level of clearance for the policy also depend on the policy itself. For small policies that only affect a small number of people and are not contentious, lower-level approval in the government hierarchy may be sufficient. Lastly, the importance of context also referred to the habitat and locational contexts; knowing the size and location of habitats and their ecosystem functions, as well as what land is adjacent to it and how the habitat fits into this context. While these are all factors that are already considered to varying extents, cultural context was identified as important but currently neglected. Together, these different types of contexts form a net of circumstances that stretches across all of the identified themes and influences the making of blue carbon policy. And although the importance of context may not have been explicitly mentioned under the theme of public acceptability, it can be argued that some of the situational and policy contexts described above are connected to it. The ministerial willingness to take risks and the political climate are closely connected to public acceptability.

Risk thus also creates a close connection between uncertainty and public acceptability. The ministerial willingness to take risks on uncertainty and the risk that comes with having to balance conflicting inputs from stakeholders and potentially antagonising one group or making an unpopular decision may seem to be two different kinds of risk, but in a democratic system where the government is elected by the public they are closely intertwined and may influence each other. This is demonstrated by the importance of the political climate in the policy making process as discussed above.

The need for communication is a further strong connector across the themes. It is highly important as is demonstrated by the fact that it emerged as its own subtheme under three out of the six overarching themes, 'uncertainty', 'public acceptability', and 'scale'. A common theme was the importance of transparency, and that communication needs to be conducted with understandable language. When communicating uncertainty, the transparency referred to the uncertainty itself, to which level there is uncertainty and why it is there. In the context of public acceptability, transparency meant communication across the board from all the actors involved in the policy development process and through all available channels from the traditional news outlets to social media and staff that deliver management on the ground. Under the theme 'scale', transparency meant communicating with local communities and feeding back on the efficiency of policies and whether they have the desired effect. Communication should thus be across the scales in addition to within the scales, and also take the form of bottom-up communication in addition to the top-down communication of what outcome is expected of a regional policy. This two-way communication and the benefits of keeping the communication channels open was also highlighted under public acceptability since communities can provide valuable feedback. Communication should thus pervade the whole policy development process and efforts should be made to ensure transparency.

In addition to large-scale links between the themes discussed so far, there are also numerous smaller links between the themes. The regular review processes some policies or plans go through for example is one of the links between the policy process and uncertainty as well as public acceptability. The review process helps to deal with uncertainty since it provides the opportunity to develop a policy or plan without having full clarity on some matters. These can then be adjusted later-on during a review period if the uncertainty has been reduced. The political climate on the other hand and thus public acceptability, can influence how this review is conducted in terms of whether a low-effort review is conducted that just ticks the boxes of performing a review or if new in-depth research regarding an issue is conducted. Another of these links exists between uncertainty and the 'information' theme. Regarding the question how much or what kind of information is required, one answer is enough information to reduce uncertainty to a level where the prioritisation of certain habitats or issues is possible.

4.5.2 Connection to Environmental Management and the Wider Policy Literature

There is not yet an established blue carbon policy analysis literature since blue carbon policy integration is still in its infancy (Lai *et al.*, 2022); however, there are several aspects that emerged from the interviews that should be discussed in the context of the EM framework and the wider policy literature, which provide some verification of the results presented in this chapter. One of the biggest criticisms of EM was its lack of inclusivity and state-centric approach (Bryant and Wilson, 1998). However, this critique was first formulated in the late 1990s and since then EM has undergone further development. According to Barrow (2005), Wilson (2009) and Bennett *et al.* (2018), EM has since become more inclusive with an increasing emphasis on local communities. The findings of the interviews confirm this assessment at least for the Scottish context. The consultation of experts and the public is firmly embedded within the Scottish policy development process. This may be due to the Scottish policy style that puts a strong focus on consultation during both the policy development and implementation phases and can thus not be generalised for other locations. Moreover, the interviews revealed that regional policy frameworks under national oversight are supported in Scottish environmental policy and that there is recognition of the value of flexible local implementation. The Scottish policy process is thus not completely top-down but rather has a strong bottom-up component and is moving away from the state-centric approach that is criticised in EM literature.

Bryant and Wilson (1998) identified a connection between actors and their understanding of the environment. They admit that their classification is simplistic since it does not allow for heterogeneity, but they present the observation that state and non-state actors have potentially divergent ways of understanding the environment. As stated in Bryant and Wilson (1998), state actors have technocentric attitudes following positivist western science, while non-state actors have a more holistic way of viewing human-environment interactions and a detailed but 'non-scientific' understanding of local environmental conditions. Following this logic, the involvement of bottom-up components in Scottish policy should thus entail a shift from focussing on science as underlying evidence to considering a broader understanding of the environment that may also be 'non-scientific'. Yet, this is a point of criticism that emerged in the interviews; evidence for policy is still concentrating on science. However, it also became clear that the holistic ecosystem view and understanding of habitats is considered of high importance. It is therefore not possible to completely agree with Bryant and Wilson's (1998) assessment in the contemporary Scottish context. Additionally, through the participatory character of policy implementation in the Scottish policy style and the apparent endorsement of regional marine policy, it is a high probability that there is flexibility for local knowledge in policy implementation in Scotland. One instance where this local knowledge could be advantageous is the identification of potential managed realignment sites. Austin *et al*. (2022) used a spatial modelling approach in their study to identify potential managed realignment sites; however, this method also has limitations, such as the quality and coverage of digital elevation data for Scotland's coastlines (Austin *et al*., 2022). Local knowledge of the environment could thus present a complimentary source of information that would be beneficial alongside this method.

As mentioned above, throughout the interviews it became clear that the holistic ecosystem view is considered important. Moreover, the interviews also demonstrated that there is an understanding and awareness that interventions in one aspect of the environment can have effects on other systems. This is a very important point, particularly in the context of saltmarshes, which exist, as established, in the coastal zone at the intersection of marine and terrestrial land management. This awareness is necessary to avoid that saltmarshes are siloed. The ecosystem approach as defined in section 1.6.1.4, can be helpful to address this potential issue since it is a strategy to integrate management across land and water. As established, saltmarshes depend, for example, on external sediment supply. This sediment also comes from riverine sources and can thus have a terrestrial origin (Ladd *et al.*, 2019). Hence, there is a very close connection to terrestrial land management. The awareness of the connection between ecosystems that was demonstrated in the interviews is confirmed by Sangiuliano (2019, 51) who found that Marine Scotland was successful in using an ecosystem approach to planning as is set out in the UK Marine Policy Statement.

It also became apparent that a variety of EM tools is used in Scottish environmental policy development. Approaches such as adaptive management, pilots, the use of the precautionary principle, and the scenario approach were named. This is not surprising since, as established, in section 1.6.1.1, sustainable development is a core concept and goal of EM and it is anchored as a general duty in the Marine (Scotland) Act (2010) that covers marine spatial management and thus blue carbon habitats, including saltmarshes. Its principles should hence be firmly embedded in marine policy and planning. However, some further development is required in this area. The current economic system was critiqued as failing to serve the public good and that economics needs to be practiced in ways that reflect wider social values and needs to be embedded in nature. Sustainable development acknowledges a healthy environment is paramount for healthy communities, which is why the triumvirate of economic viability, environmental protection, and the health and happiness of people was defined as its goals. The current economic system that is divorced from the environment is thus not in accordance with these goals. In addition, the cultural context of communities should be taken more into account in policy making, which indicates a shortcoming towards reaching the goal of the health and happiness of people.

Regarding the wider policy development literature, the interviews confirmed Cairney's (2016b) assessment that the simple policy-making cycle is a misleading simplification of the policy-making process. It also strongly confirmed and acknowledged that policymaking operates within a bounded rationality framing. There is an acceptance that the perfect level of information and clarity of evidence may not be reached and that it should not be a barrier to developing policy. This is accompanied by the understanding that there may be stakeholders with competing interests that influence high level decision-makers in the policy development process and that these decision-makers have to take factors beyond the evidence base into account such as the political climate and the impact of a policy on certain stakeholders and sectors. Hence, the overall aim and choice of policymakers is not always clear. All these elements, (i) limited information, (ii) unclear aim, (iii) and unclear choice, are defining factors of bounded rationality (Cairney, 2016b) and the policymakers that act within it. Cairney (2016b) identified the potential issue that comprehensive rationality could be viewed uncritically as the ideal form of policymaking that should be aspired to when its original purpose was only to demonstrate what does not and cannot happen in policymaking systems. This issue was not identified from the interview data.

4.6 Conclusion - Integrating the Carbon Storage Benefit of Saltmarshes into Scottish Policy

Overall, there was a strong consensus during the interviews that effective science-policy integration requires a comprehensive approach to blue carbon habitats that goes beyond the carbon storage ES and that the natural capital approach is a powerful tool that is increasing in importance with the aim to conduct holistic natural capital accounting. Further development of the natural capital approach is necessary though to encompass as many ES as possible. There is a danger of over-investing into high carbon systems that could, in extreme cases, create carbon monocultures with a negative impact on the environment; a diversification of investment and a speedy expansion of the natural capital approach to other ES is thus important. Environmental policy needs to be developed on both the national and regional scale and uncertainty should not be a barrier to policy but in reality, it may still be the case for blue carbon. The policy process is, moreover, as discussed in depth in section 4.5.1, not always the same or straightforward since many aspects of it depend on the situational or policy context.

Saltmarshes in particular, have the potential to be included in the GHG Inventory, but this is not yet the case and it is unclear what the potential timescale is for its inclusion. Moreover, following the notion that a holistic approach should be taken and that as a NbS, the focus should be on the multitude of benefits that saltmarshes provide instead of concentrating just on the carbon benefit, management should go beyond inventories and the climate change plan. This is particularly important since saltmarsh benefits can also contribute to climate change adaptation. The avenue for such holistic management that delivers multiple benefits would be through marine spatial management under the Marine (Scotland) Act (2010) and thus the NMP and RMPs or the Blue Economy Action Plan. Ideally, the carbon storage benefit would be picked up alongside the other benefits in the early stages of a policy plan, such as the initial assessment stage but it can also be added later during a review of the plan. The recent example of the Clyde Marine Plan demonstrates that uncertainty in the evidence may still have an influence on whether or to what extent the carbon storage benefit is included in policy. But, since the inclusion during a review process is possible once more information is available, it is still an attainable goal. Although regional scale policy needs to fit into the overall national plan, there is flexibility to decide to take a stronger approach or to prioritise certain aspects of benefits of a habitat. This local flexibility and possibility for involvement is important since the restoration or enhancement of blue carbon habitats, including saltmarshes, is often led by community groups.

It has been pointed out that the inclusion of blue carbon habitats into policy should not be rushed and that the first objective should be to have a good understanding of these habitats. This research suggests that regarding saltmarshes, enough information is available though to be confident that they do provide a wide range of benefits. One approach for early policy inclusion could thus be to focus on their protection and to ensure that nature is given sufficient space to safeguard habitat extent, functions, and resulting ES. Interventions or measures for enhancement may still be decided on at a later point when more information is available.

5 DISCUSSION AND CONCLUSION

This interdisciplinary thesis investigated the initial assumption that saltmarsh management is best integrated into climate change mitigation policy and comes to the different conclusion that including saltmarshes into the MSP framework may be preferable to be able to capitalise on the additional ES they provide. Scotland was the focus of this study due to the Scottish government's interest in exploring blue carbon as a NbS for climate change mitigation and adaptation and its ambitious emission reduction targets (SBCF, 2022a; Scottish Government, 2022). The thesis aimed to provide a holistic, in-depth study of the Scottish saltmarsh blue carbon resource that connects science and policy. To achieve this aim, the uncertainty regarding the depth of Scottish saltmarshes was investigated, the public's management preferences and WTP for the saltmarsh carbon ES determined, and its incorporation into Scottish policy analysed. To this end, the thesis employed both quantitative and qualitative methods. The following quote by Wilson and Bryant (1997, 7; in Warren, 2009) helps to highlight the importance of this approach: "the central predicament of all environmental managers [is] the quest for predictability in a context of increasing social and environmental uncertainty". The scenario approach investigating different saltmarsh OC stocks according to a variety of depths helps to increase the predictability of saltmarsh carbon storage and allows for swift adjustments if new findings regarding the depth of Scottish saltmarshes come to light. Moreover, knowing the public's management preferences and WTP for these preferences increases the predictability of the acceptability of potential management interventions or policies. The LCA provided insightful information on respondent's cultural and relational values that determined their preferences for saltmarsh management. This represents a strong link between the results of Chapters 3 and 4 since this information can be helpful for policymakers to achieve consensus for proposed policies and interventions, thus reducing controversy. The insight generated by the LCA also demonstrates the importance of researching non-monetary factors since it illustrates the significance of cultural and relational values for the public's preferences and emphasises that there are factors beyond monetary considerations that are impactful and deserve attention. The findings of this thesis are thus valuable for Scottish decision-makers. Moreover, this thesis made three key contributions to the climate change mitigation and adaptation literature: (i) it has demonstrated the relatively small impact saltmarsh blue carbon has on climate change mitigation (based on a Scottish case study) which is in line with IPCC findings (IPCC, 2019); (ii) it has provided evidence that there are strong public preferences to manage saltmarshes for climate change mitigation and adaptation benefits (based on a Scottish cases study); and (iii) it has identified a range of factors that are significant in blue carbon policy inclusion (Scottish case study) and what this means in terms of using blue carbon for mitigation and adaptation. Furthermore, responding to the call for more interdisciplinary studies in the field of sustainability science (Fernandes and Rauen, 2016), this thesis traced the steps from a saltmarsh OC stock assessment to its valuation and policy integration and thus successfully connected science and policy.

5.1 Overview of the Chapter Findings

5.1.1 Uncertainty in the Saltmarsh Carbon Stock

Chapter 2 took a scenario approach to analyse the change in the carbon OC stock with different saltmarsh depths and highlight the uncertainty inherent to saltmarsh data. There were several key conclusions that were drawn from this chapter. Firstly, saltmarsh soil depth was indeed an important factor in the calculation of the OC stock. Secondly, when data of several saltmarshes was combined and an average calculated, the method that was used to calculate the average could influence the OC stock estimation and introduce a further level of uncertainty to the data. Thirdly, using only surficial soil samples may not provide an accurate picture of the Scottish saltmarsh OC stock and distort the importance of some saltmarshes regarding their carbon storage capacity compared to others. Fourthly, it is likely that using OC content and dry bulk density averages present an overestimate of the OC stock for the upper soil layers and an underestimate for the deeper layers.

5.1.2 The Public's Preferences and WTP for Saltmarsh Management

Chapter 3 presented a valuation of the Scottish saltmarsh carbon ES with a stated preference method that allowed the determination of the Scottish public's preferences for saltmarsh management and their WTP for management. The results demonstrated that the management scenarios were preferred over the business-as-usual scenario and that providing more information did not increase respondents' WTP but that it did decrease the randomness of their choices. This highlights the importance of clear communication, which was also stressed on the decision-maker and advisor side in the expert-interviews on blue carbon policy integration (Chapter 4). Importantly, the results demonstrated that there was a preference for an improvement of all included ES, not just the carbon storage ES. The LCA thus revealed that either the choices of the largest group were not entirely rational according to choice theory since they preferred an improvement in all ES irrespective of the price or that participants allocated to this group refused to think in utilitarian terms; the second largest group demonstrated more rational behaviour and prioritised an increase in the carbon storage ES.

5.1.3 Integrating Blue Carbon into Scottish Policy

Chapter 4 analysed expert interviews on the integration of the saltmarsh carbon storage ES into Scottish policy, particularly which factors are important for a successful integration and how they interact and shape blue carbon policy in the Scottish context. The key conclusion of the analysis was that there is a clear consensus that all saltmarsh ES are important and that there should not be an exclusive focus on the carbon storage ES. The importance of a holistic approach to saltmarsh management was emphasised and that NbS should focus on the multitude of benefits rather than just one. The natural capital approach aiming to conduct holistic natural capital accounting was highlighted as a powerful tool with potential to achieve this. Saltmarsh management should thus go beyond preserving and increasing their carbon storage capacity for climate change mitigation and also realise their potential for climate change adaptation. While saltmarshes have the potential to be included in the GHG Inventory for its mitigation potential, this is not yet the case and within Scottish environmental policy, particularly MSP, the holistic approach can be pursued, and multiple mitigation and adaptation benefits delivered. A further key conclusion was that uncertainty in the available data should not be a barrier for policy, but that in reality this may still be the case for blue carbon habitats like saltmarshes. Nevertheless, enough is known about saltmarsh benefits for an early policy inclusion that focuses on conservation; interventions or other management options could still be decided on at a later point.

These results of Chapter 4 link well with the results of Chapter 3. The LCA provided insightful information on respondent's cultural and relational values that determined their preferences for saltmarsh management. This information can be helpful for policymakers to achieve consensus for proposed policies and interventions, thus reducing controversy. The insight generated by the LCA also demonstrates the importance of researching non-monetary factors since it illustrates the significance of cultural and relational values for the public's preferences and emphasises that there are factors beyond monetary considerations that are impactful and deserve attention. A holistic approach and natural capital accounting would also capture these non-monetary values.

5.2 The Mitigation Potential of Scottish Saltmarshes

Overall, the mitigation potential of Scottish saltmarshes is quite small. In 2019, the Scottish GHG emissions amounted to 47.8 $MtCO_2e$, which is 47,800,000 tonnes CO_2e (Scottish Government, 2021). When the OC stocks of the different scenarios that were estimated in Chapter 2 are converted into CO_2 equivalent (see Table 5.1), it is quite obvious that even for the scenario with the greatest depth, the carbon stored in Scottish saltmarshes is only 10% (Waulkmill Bay and Loch of Stenness data) – 11.5% (top 10 cm Scotland-wide surficial soil data, Ruranska *et al.*, 2020 data) of the 2019 emissions. Since this shows the relation of the entire OC stock that has been accumulated over decades or even millennia to annual Scottish emissions, it is reasonable to deduce that the future sequestration of these existing marshes and the creation of further saltmarsh area will have a negligible mitigation potential.

Table 5.1: Upscaling scenarios with the Loch of Stenness and Waulkmill Bay saltmarsh data and the top 10 cm surficial soil data (Ruranska *et al.*, 2020) converted to CO_2 equivalent.

Scenario	Area (ha)	Depth (m)	Soil OC Stock (tonnes)	Soil OC Stock (tonnes) in CO_2 equivalent
OC content averages of all Loch of Stenness and Waulkmill Bay samples				
20 cm depth	5820.39	0.20	635195 ± 168088	2331166 ± 616883
30 cm depth	5820.39	0.30	952793 ± 252132	3496750 ± 925324
40 cm depth	5820.39	0.40	1270391 ± 336176	4662335 ± 1233766
OC content averages of the top 10 cm surficial soil data				
20 cm depth	5820.39	0.20	749284 ± 253575	2749872 ± 930620
30 cm depth	5820.39	0.30	1123927 ± 380362	4124812 ± 1395929
40 cm depth	5820.39	0.40	1498569 ± 507149	5499748 ± 1861237

This assessment is confirmed by Bradfer-Lawrence *et al.* (2021) who conducted a study on the potential contribution of UK terrestrial NbS to achieving the national net-zero emissions target; saltmarshes were included in their study along with peatlands and woodlands. They conclude that even the cumulative mitigation based on the most ambitious restoration or creation targets of all three of these habitats together would only be equivalent to 3 years of UK emissions. Out of these habitats peatlands represent the biggest mitigation potential while the contribution from saltmarsh creation is limited due to their small area and the limited area for creation.

Saltmarsh creation would take the form of managed realignment (Bradfer-Lawrence *et al.*, 2021), which can only be implemented along suitable stretches of low-lying coastline where artificial structures, which are often remnants of historical land reclamation, prevent the land from being flooded. Only fifteen suitable sites were identified for Scotland in a recent study by Austin *et al.* (2022). Overall, the study predicts that these 15 sites could store an additional 63139 ± 32778 tonnes of carbon in their top 10 cm of soil. However, this does not take relative sea-level rise into account. Depending on the applied emission scenario up to about 50% of this potential could be lost due to sea-level rise. This prediction matches the conclusions drawn above. Additionally, it is also still unclear when managed realignment sites reach a stable state with natural rates of carbon sequestration (Austin *et al.*, 2022), which signifies that it is unclear in which timeframe the mitigation would be delivered. There are studies though that suggest a rapid carbon accumulation in the early years after managed realignment was implemented. Wollenberg *et al.*, (2018) found that six years after implementing managed realignment, a restored saltmarsh in the Bay of Fundy, Canada had a carbon burial rate of more than five times the rate reported for a mature marsh close by. Burden *et al.* (2019) similarly reports that modelling of older realigned sites in the UK shows a rapid carbon accumulation during the first 20 years.

This discussion supports the premise that carbon storage should only be highlighted as one of many benefits that saltmarshes provide since the carbon storage ES and its potential expansion on its own is quite small. This is in line with the idea of NbS that take a holistic view of ecosystems and overlaps with the public's preferences regarding saltmarsh management. As already mentioned in the respective chapter, DCEs can aid the design of socially optimal policies and even though the LCA results indicated that the largest group of participants' choices may not have followed rationality according to choice theory (i.e., the payment seems to have not been a consideration), a clear preference for a holistic approach is still strongly implied and those results should not be disregarded. Overall, the results from the individual chapters thus refute the initial assumption that saltmarsh blue carbon is best integrated into climate change policy due its mitigation potential and the urgency of climate change mitigation. While saltmarshes could play a small role in this context, there is a larger benefit in managing them for all their ES and the additional climate change adaptation benefits within MSP. The research conducted in this thesis thus provides evidence to support the argument made by Merk *et al*., (2022). In their commentary, the authors argue that blue carbon ecosystems can contribute to CO_2 removal but that the value of this service is low compared to the overall value of their total ES. Further, they stress that focussing on the carbon ES could create trade-offs to the detriment of other important saltmarsh ES that could otherwise benefit the local community and that managing blue carbon ecosystems with a carbon focus could thus fail to achieve socially optimal outcomes.

5.3 Saltmarsh Management in Scotland

As mentioned in the introduction, saltmarshes are mentioned under two General Policies in the NMP, which require the reduction of pressures and the safeguarding of ES and call for the consideration of enhancing natural carbon sinks if opportunities arise, and that natural processes and features should be utilised for coastal protection and flood risk management. One of the results of Chapter 4 was that flexibility between the policy scales is possible and that RMPs can decide to take a stronger approach or prioritise certain aspects that are covered under the NMP. RMPs could thus prioritise saltmarshes as coastal features that provide many ES. A further key conclusion from this chapter was also that saltmarsh management should not be hindered by the uncertainty that is inherent to the existing data on the habitat and that sufficient information is available to justify at least a focus on saltmarsh conservation. This approach could also potentially reduce the loss of what Goldstein *et al.* (2020, 287) refer to as 'irrecoverable carbon' which refers to ecosystem carbon stocks that would not be able to "recover within a timescale meaningful to the remaining carbon budget". The recoverability depends on both the sequestration rate and which timeframe is considered as meaningful. Goldstein *et al.* (2020) chose 30 years in line with the IPCC assessment that net-zero emissions have to be achieved by 2050. Saltmarshes are among the irrecoverable carbon since their average time to recover is estimated as 64 years. Managing saltmarshes within the MSP framework also provides advantages for an eventual inclusion into the GHG Inventory. It would ensure that saltmarshes are in good condition and that Scotland is thus one step ahead to realising their full potential for carbon sequestration and storage within the

GHG Inventory if they are included. This is also in line with the leadership role Scotland aspires to regarding climate change mitigation and reaching net-zero emissions by 2045 (SNP, 2022).

One specific existing scheme under which saltmarsh conservation can be achieved are Sites of Special Scientific Interest (SSSI). SSSIs are protected natural features that are considered to represent Scotland's natural heritage in terms of their flora, fauna, geology, or geomorphology. They are designated by NatureScot under the Nature Conservation (Scotland) Act (2004) (NatureScot, 2021a). The majority of saltmarshes are already covered under this scheme and are designated as SSSIs (Austin *et al.*, 2021). They are, however, selected as biological SSSIs (Rees *et al.*, 2019); their carbon storage potential does not play a role in their selection. Activities that are likely to damage the features for which the site is designated require consent from NatureScot before they can be carried out (NatureScot, 2021b). Broadening the criteria for SSSI designation and including carbon could thus be an avenue to include more saltmarshes within this designation and to add a layer of protection for the saltmarshes' carbon stocks.

5.4 Future Research

There is strong potential to build upon the research conducted for all chapters of this thesis in further research. Firstly, to further improve the estimation of the carbon stock and to get a better understanding of the average depth of Scottish saltmarshes, more depth records could be collected in future research studies. While the depth scenarios developed in this thesis provide valuable information for policymakers to reduce the uncertainty of the impacts a policy may have regarding carbon storage or emissions, a more refined average saltmarsh depth would reduce this uncertainty even further and would thus be beneficial, particularly since there are indications that uncertainty may still be inhibiting blue carbon policy inclusion. Secondly, additional analysis could also be performed on the data that was collected with the DCE. Since the focus of the analysis was mainly on the carbon ES, the data could be used to run more analysis on the other ES that were included in the DCE (i.e., the flood defence benefit, the biodiversity benefit, and the recreational benefit of saltmarshes). Regarding the flood defence attribute, for example, it could be analysed if respondents that experienced flooding in the past had a higher preference for improving this attribute than other respondents. It could also be analysed if respondents who had visited saltmarshes before taking the survey showed a higher preference for saltmarsh management to achieve an improvement in the attributes than other respondents. These are just two examples for further analysis that is possible and not an exhaustive list. Third, the interviews that were conducted could be expanded; it was not possible to interview an environmental economist from Scottish Government, so further studies could consider the perspective of that policy dimension on whether blue carbon policy is effective and sustainable. The analysis of this thesis has focussed on the factors that shape blue carbon policy. Moreover, the pool of potential interviewees could be increased by widening the target group to academics who conduct research in this area. This would also increase the sample size but could have the drawback of adding data to the dataset of actors that do not have the same level of influence in policymaking as the actors that were interviewed for this study. To mitigate this effect, a different data collection method could be used. Focus groups or workshops could be facilitated (e.g. at conferences where these actors mix) with participants of diverse backgrounds to collect the data and foster an exchange of ideas and collaboration. Lastly, connecting chapters 2 and 3, a study could be conducted about the Orkney saltmarshes that assesses how much carbon would be lost according to the status quo scenario and what the WTP would be to prevent this loss. Added to the WTP for the other ecosystem services, this would simulate an overall WTP for different management scenarios and thus a budget for alternative policies (i.e. a policy that maintains the status quo and a policy that increases one or several of the ES). However, before this study could be conducted, it would be necessary to consider the local context and to conduct some analysis if the status quo values of the carbon attribute also apply to these specific saltmarshes. This is important since the values for the ES in the choice experiment were based on national averages and as established in section 1.3.2, saltmarshes and their characteristics can vary significantly between different locations.

While the insights of the policy integration chapter may be more applicable for countries with a similar policy style, the scenario approach developed in the second chapter can be transferred to any other jurisdiction with an initial blue carbon dataset. The results from the DCE are not one-on-one transferable since there are a variety of factors such as cultural context that can influence the results. However, this thesis has demonstrated that it is a valuable tool to gain insights regarding the public's preferences and WTP for saltmarsh management and thus an indication of the acceptability of potential policy. Moreover, it provides insights whether the provision of information could be employed to influence the public's preferences and WTP. Overall, the research of this thesis is thus significant beyond the Scottish context since it provides a roadmap for connecting blue carbon science with policy and further similar holistic blue carbon studies in other countries are hence well justified. Blue carbon policy integration is still in its infancy and extending this research to cover further regions could provide further valuable input and guidance for other decision-makers that are at the very beginning of this process.

References

Adam, P., 1990. *Saltmarsh Ecology.* Cambridge: Cambridge University Press.

Adam, P., 2016. Saltmarshes. In: M.J. Kennish, ed. 2016. *Encyclopedia of Estuaries*. Dordrecht: Springer, pp. 515-535.

Adams, C.A. *et al.*, 2012. Nitrous oxide and methane fluxes vs. carbon, nitrogen, and phosphorous burial in new intertidal and saltmarsh sediments. *Science of the Total Environment*, 434, pp. 240-251.

Adnitt, C. *et al.*, 2007. *Saltmarsh Management Manual: Joint Defra/Environmental Agency Flood and Coastal Erosion Risk Management R&D Technical Report SC030220.* Bristol: Environment Agency.

Alberini, A. *et al.*, 2018. Preferences for energy efficiency vs. renewables: what is the willingness to pay to reduce CO2 emissions? *Ecological Economics,* 144, pp. 171-185.

Applegate, J.S., 2000. The precautionary preference: An American perspective on the precautionary principle. *Human and Ecological Risk Assessment*, 6(3), pp. 413-443.

Armitage, D. *et al*., 2008. Adaptive co-management and the paradox of learning. *Global Environmental Change*, 18(1), pp. 86-98.

Atkinson, G. *et al*., 2018. *Cost benefit Analysis and the Environment: Further Developments and Policy Use*. Paris: OECD Publishing.

Austin, W. *et al.*, 2021. Blue carbon stock in Scottish saltmarsh soils. *Scottish Marine and Freshwater Science*, 12(13), pp. 1-39.

Austin, W. *et al.*, 2022. Scottish saltmarsh, sea-level rise, and the potential for managed realignment to deliver blue carbon gains. *ClimateXChange Publications*, [online] Available at: <https://era.ed.ac.uk/handle/1842/39119> [Accessed 11 August 2022].

Barbier, E.B. *et al.*, 2011. The value of estuarine and coastal ecosystem services. *Ecological Monographs,* 81(2), pp. 169-193.

Barlow, N.L.M. *et al*., 2014. Salt-marsh reconstructions of relative sea-level change in the North Atlantic during the last 2000 years. *Quaternary Science Reviews*, 99, pp.1-16.

Barrow, C.J., 1999. *Environmental Management: Principles and Practice*. London: Routledge.

Barrow, C.J., 2005. *Environmental Management and Development*. New York: Routledge.

Bateman, I.J. *et al.*, 2002. *Economic Valuation with Stated Preference Techniques: A Manual*. Cheltenham: Edward Elgar.

Bauer, D.M. *et al.*, 2004. Public preferences for compensatory mitigation of salt marsh losses: A contingent choice of alternatives. *Conservation Biology*, 18(2), pp. 401-411.

Beaumont, N.J. *et al.*, 2014. The value of carbon sequestration and storage in coastal habitats. *Estuarine, Coastal and Shelf Science*, 137(1), pp. 32-40.

BEIS, 2021a. *Policy Paper - Valuation of greenhouse gas emissions: for policy appraisal and evaluation*. [online] Available at: <https://www.gov.uk/government/publications/valuing-greenhouse-gas-emissions-in-policy-appraisal/valuation-of-greenhouse-gas-emissions-for-policy-appraisal-and-evaluation> [Accessed 22 June 2022].

BEIS, 2021b. *UK greenhouse gas emissions: other technical reports*. [online] Available at: <https://www.gov.uk/government/publications/uk-greenhouse-gas-emissions-explanatory-notes> [Accessed 18 April 2022].

BEIS, 2022. *UK greenhouse gas emission statistics: planned methodology changes*. [online] Available at: <https://www.gov.uk/government/publications/planned-methodology-changes-for-uk-greenhouse-gas-emissions> [Accessed 18 April 2022].

Bennett, J. and Adamowicz, V., 2001. Some Fundamentals of Environmental Choice Modelling. In: J. Bennett and R. Blamey, eds. 2001. *The Choice Modelling Approach to Environmental Valuation.* Cheltenham: Edward Elgar. pp. 37-69.

Bennett, J. and Blamey, R., 2001. Introduction. In: J. Bennett and R. Blamey, eds. 2001. *The Choice Modelling Approach to Environmental Valuation.* Cheltenham: Edward Elgar. pp. 1-10.

Bennett, N. *et al*., 2018. Environmental stewardship: A conceptual review and analytical framework. *Environmental Management*, 61(4), pp. 597-614.

Bergstrom, J. and Dillman, B., 1985. Public environmental amenity benefits of private land: The case of prime agricultural land. *Southern Journal of Agricultural Economics*, 17 (1), pp. 139-150.

Bergstrom, J.C. *et al.*, 1989. Information effects in contingent markets. *American Journal of Agricultural Economics,* 71(3), pp. 685-691.

Birol, E. and Cox, V., 2007. Using choice experiments to design wetland management programmes: The case of Severn Estuary wetland, UK. *Journal of Environment Planning and Management*, 50(3), pp. 363-380.

Bodansky D., 1991. Scientific uncertainty and the precautionary principle. *Environment*, 33(7), pp.4-44.

Boorman, L.A. *et al.*, 2001. The effect of rates of sedimentation and tidal submersion regimes on the growth of salt marsh plants. *Continental Shelf Research*, 21(18-19), pp. 2155-2165.

Borges, E.C. *et al.*, 2021. Evaluation of the spatial variability of ecosystem services and natural capital: the urban land cover change impacts on carbon stocks. *International Journal of Sustainable Development & World Ecology*, 28(4), pp. 339-349.

Boyle, K.J., 1989. Commodity specification and the framing of contingent-valuation questions. *Land Economics*, 65(1), pp. 57-63.

Boyle, K.J. *et al.,* 1990. Species substitution and question sequencing in contingent valuation surveys evaluating the hunting of several types of wildlife. *Leisure Sciences*, 12(1), pp.103-118.

Bradfer-Lawrence, T. *et al.*, 2021. The potential contribution of terrestrial nature-based solutions to a national 'net zero' climate target. *Journal of Applied Ecology*, 58(11), pp. 2349-2360.

Brown, B.J. *et al*., 1987. Global sustainability: toward definition. *Environmental Management*, 11(6), pp. 713-719.

Brundtland, G.H., 1987. *Our Common Future: Report of the World Commission on Environment and Development. Geneva, UN-Document A/42/427*. [online] Available at: <http://www.un-documents.net/ocf-ov.htm> [Accessed 28 June 2022].

Brunyeel, S., 2009. Thinking down or acting up? Governmental and grassroots environmental thought and action in the transboundary Northern Plains. *Environments,* 36(3), pp. 55-72.

Bryant, R.L. and Wilson, G.A., 1998. Rethinking environmental management. *Progress in Human Geography*, 22(3), pp. 321-343.

Burden, A. *et al.*, 2019. Effect of restoration on saltmarsh carbon accumulation in Eastern England. *Biology Letters*, 15(1), pp. 1-4.

Burrows, M. *et al.*, 2014. *Assessment of carbon budgets and potential blue carbon stores in Scotland's coastal and marine environment. Project Report. Scottish Natural Heritage Commissioned Report No. 761* [pdf] SNH. Available at: <https://www.nature.scot/snh-commissioned-report-761-assessment-carbon-budgets-and-potential-blue-carbon-stores-scotlands> [Accessed 19 April 2020].

Cairney, P., 2008. Has devolution changed the 'British Policy Style'? *British Politics*, 3(3), pp. 350-372.

Cairney, P., 2014. The territorialisation of interest representation in Scotland: Did devolution produce a new form of group-government relations? *Territory, Politics, Governance,* 2(3), pp. 303-321.

Cairney, P., 2016a. The future of Scottish government and public policy: A distinctive Scottish style? In D. McTavish: Politics in Scotland? In: D. McTavish, ed. 2016. *Politics in Scotland*. London: Routledge, Taylor & Francis Group. pp. 213-228.

Cairney, P., 2016b. *The Politics of Evidence-based Policy Making*. Basingstoke: Palgrave Macmillan.

Cameron, T.A. and Englin, J., 1997. Respondent experience and contingent valuation of environmental goods. *Journal of Environmental Economics and Management*, 33(3), pp. 296-313.

Cannell, M.G.R. *et al.*, 1999. National inventories of terrestrial carbon sources and sinks: The U.K. Experience. *Climatic Change*, 42(3), pp. 505-530.

Ceci, C., 2017. *Benefits of coastal managed realignment for society*. [online] Available at: <https://community.rspb.org.uk/ourwork/b/science/posts/benefits-of-coastal-managed-realignment-for-society> [Accessed 10 August 2022].

Champ, P.A. *et al.*, 2017. *A Primer on Non-Market Valuation*. 2nd ed. Dordrecht: Springer.

Chapman, V.J., 1960. *Salt Marshes and Salt Deserts of the World*. London: Leonard Hill.

Chmura, G.L. *et al.*, 2003. Global carbon sequestration in tidal, saline wetland soils. *Global Biogeochemical Cycles*, 17(4), pp. 1-12.

Chmura, G.L., 2013. What do we need to assess the sustainability of the tidal salt marsh carbon sink? *Ocean & Coastal Management*, 83, pp. 25-31.

Choice Metrics, 2018. *Ngene User Manual and Reference Guide* [pdf] Choice Metrics. Available at: <http://www.choice-metrics.com/download.html#manual> [Accessed 5 March 2020].

Climate Change (Emissions Reduction Targets) (Scotland) Act 2019. [online] Available at: <https://www.legislation.gov.uk/asp/2019/15/contents/enacted> [Accessed 05 August 2022].

Climate Change (Scotland) Act 2009. [online] Available at: <https://www.legislation.gov.uk/asp/2009/12/contents> [Accessed 05 August 2022].

Clyde Marine Planning Partnership, 2018. *Clyde Regional Marine Plan: Pre-Consultation Draft*. [online] Available at: <https://www.clydemarineplan.scot/marine-planning/clyde-regional-marine-plan/#assessment> [Accessed 10 August 2022].

Codden, C.J. *et al*., 2022. Non-conservative behavior of dissolved organic carbon in a Georgia salt marsh creek indicates summer outwelling. *Estuarine, Coastal and Shelf Science*, 265, 107709, pp. 1-12.

Collins Dictionary, 2022. *Definition of Basket Case*. [online] Available at: <https://www.collinsdictionary.com/de/worterbuch/englisch/basket-case> [Accessed 14 August 2022].

Convention on Biological Biodiversity, 2004. The Ecosystem Approach. [online] Available at: <https://www.cbd.int/doc/publications/ea-text-en.pdf> [Accessed 7 February 2023].

Cope, M., 2021. Organizing, Coding, and Analyzing Qualitative Data, In: I. Hay and M. Cope, ed. 2021. *Qualitative Research Methods in Human Geography*. Don Mills: Oxford University Press. pp. 355-375.

Costanza, R. *et al.* 1997. The value of the world's ecosystem services and natural capital. *Nature*, 387, pp. 253-260.

Costanza, R. *et al.*, 2014. Changes in the global value of ecosystem services. *Global Environmental Change*, 26, pp. 152-158.

Costanza, R. *et al.*, 2017. Twenty years of ecosystem services: How far have we come and how far do we still need to go? *Ecosystem Services*, 28(A), pp. 1-16.

Czajkowski, M. and Budziński, W., 2019. Simulation error in maximum likelihood estimation of discrete choice models. *Journal of Choice Modelling*, 31, pp. 73-85.

Czajkowski, M. *et al.*, 2014. Learning and fatigue effects revisited: Investigating the effects of accounting for unobservable preference and scale heterogeneity. *Land Economics*, 90(2), pp.324-351.

Czajkowski, M. *et al.*, 2015. The effects of experience on preferences: Theory and empirics for environmental public goods. *American Journal of Agricultural Economics*, 97(1), pp. 333-351.

Czajkowski, M. *et al.*, 2016a. Controlling for the effects of information in a public goods discrete choice model. *Environmental and Resource Economics*, 63(3), pp. 523-544.

Czajkowski, M. *et al.*, 2016b. Information and learning in stated-preference studies. Working Paper 20/2016 (211), Faculty of Economic Sciences, University of Warsaw.

Dadey, K.A. *et al*., 1992. Dry-bulk density: its use and determination. *Proceedings of the Ocean Drilling Program, Scientific Results*, 126, pp. 551-554.

Davidson, K.E. *et al.*, 2017. Livestock grazing alters multiple ecosystem properties and services in salt marshes: a meta-analysis. *Journal of Applied Ecology*, 54(5), pp. 1395-1405.

DECC, 2009. *Carbon valuation in UK policy appraisal: a revised approach.* [online] Available at: <https://www.gov.uk/government/publications/carbon-valuation-in-uk-policy-appraisal-a-revised-approach> [Accessed 13 August 2022].

DEFRA, 2007. *The Social Cost of Carbon and the Shadow Price of Carbon: What They Are, And How to Use Them in Economic Appraisal in the UK.* Available at: <https://www.gov.uk/government/publications/shadow-price-of-carbon-economic-appraisal-in-the-uk> [Accessed 14 November 2019].

De Groot, R. *et al.*, 2017. 2.1. A short history of the ecosystem services concept Rudolf. In: B. Burkhard and J. Maes, (eds.) 2017. *Mapping Ecosystem Services*, Sofia: Pensoft Publishers, pp. 31-34.

Dencer-Brown, A.M. *et al.*, 2022. Integrating blue: How do we make nationally determined contributions work for both blue carbon and local coastal communities? *Ambio,* 51(9), pp. 1978-1993.

Diederich, J. and Goeschl, T., 2011. *Willingness to pay for individual greenhouse gas emissions reductions: evidence from a large field experiment*. Discussion paper no 517. Department of Economics, Heidelberg University. Available at: <http://archiv.ub.uni-heidelberg.de/volltextserver/12543/> [Accessed 19 April 2020].

Döll, P., 2004. Qualitative-quantitative scenarios as a means to support sustainability-oriented regional planning. In: A. Marquina, (ed) 2004. *Environmental Challenges in the Mediterranean 2000–2050*. NATO Science Series, vol 37. Springer, Dordrecht, pp. 47-60.

Doody, J. P., 2013. Coastal squeeze and managed realignment in southeast England, does it tell us anything about the future? *Ocean and Coastal Management*, 79, pp. 34-41.

Drupp, M. A. *et al.*, 2018. Discounting disentangled. *American Economic Journal*, 10(4), pp. 109-134.

Duarte, C.M. *et al.*, 2005. Major role of marine vegetation on the oceanic carbon cycle. *Biogeosciences*, 2(1), pp. 1-8.

Duarte, C.M. *et al.*, 2013. The role of coastal plant communities for climate change mitigation and adaptation. *Nature Climate Change*, 3, pp. 961-968.

Dunlap, R.E. *et al.*, 2000. Measuring endorsement of the new ecological paradigm: A revised NEP scale. *Journal of Social Issues*, 56(3), pp. 425-442.

Dunlap, R.E. and Van Liere, K.D., 1978. The “new environmental paradigm”: A proposed measuring instrument and preliminary results. *Journal of Environmental Education*, 9, pp. 10-19.
Dunn, K., 2021. Engaging Interviews, In: I. Hay and M. Cope, ed. 2021. *Qualitative Research Methods in Human Geography*. Don Mills: Oxford University Press. pp. 148-185.

Ehler, C.N., 2012. Perspective: 13 Myths of marine spatial planning. *Marine Ecosystems and Management*, 5(5), pp. 1-3.

Ehler, C.N. and Douvere, F., 2009. Marine spatial planning: A step-by-step approach toward ecosystem-based management. Intergovernmental Oceanographic Commission and Man and the Biosphere Programme. iOC Manual and Guides no. 53, iCaM Dossier no. 6. Paris: UNESCO.

Ekstrand, E.R. and Loomis, J., 1998. Incorporating respondent uncertainty when estimating willingness to pay for protecting critical habitat for threatened and endangered fish. *Water Resources Research*, 34(11), pp. 3149-3155.

Elementar, year unknown. *Elementar Vario EL Cube*. [online] Available at: <http://www.spektronika.ru/sites/default/files/vario_el_cube.pdf> [Accessed 25 May 2022].

European Environment Agency, 2023. *Quantitative scenarios*. [online] Available at: <https://www.eea.europa.eu/help/glossary/eea-glossary/quantitative-scenarios> [Accessed 07 February 2023].

Everitt, B.S. *et al.* 2011. *Cluster Analysis*. 5th ed. Chichester: Wiley.

Faccioli, M. *et al.*, 2019. Stated preferences for conservation policies under uncertainty: Insights on the effect of individuals' risk attitudes in the environmental domain. *Environmental and Resource Economics*, 73(2), pp. 627-659.

Farber, S.C. *et al.*, 2002. Economic and ecological concepts for valuing ecosystem services. *Ecological Economics*, 41(3), pp. 375-392.

Fávero, L.P. and Belfiore, P., 2019. *Data Science for Business and Decision Making*. London: Academic Press, an imprint of Elsevier.

Fernandes, V. and Rauen, W.B., 2016. Sustainability: an interdisciplinary field. *Fronteiras: Journal of Social, Technological and Environmental Science*, 5(3), pp. 188-204.

Fest, B.J., 2022. A review of sediment carbon sampling methods in mangroves and their broader impacts on stock estimates for blue carbon ecosystems. *Science of the Total Environment*, 816, pp. 1-12.

Fiebig, D. *et al.*, 2010. The generalized multinomial logit model: Accounting for scale and coefficient heterogeneity. *Marketing Science*, 29(3), pp. 393-421.

Frazão Santos, C. *et al.*, 2019. Marine Spatial Planning. In: C. Sheppard (ed.) 2019. *World Seas: An Environmental Evaluation - Volume III: Ecological Issues and Environmental Impacts*. London: Academic Press. pp. 571-592.

Fuller, R.J., 2010. *Bird Habitats in Britain*. Digital ed. London: T & AD Poyser.
Greenberg, R. *et al.*, 2014. The distribution and conservation of birds of coastal salt marshes. In: B. Maslo and J.L Lockwood, ed. 2014. *Coastal Conservation.* Cambridge: Cambridge University Press. pp. 180-242.

Goldstein, A. *et al.*, 2020. Protecting irrecoverable carbon in Earth's ecosystems. *Nature Climate Change*, 10(4), pp. 287-295.

González-García, V. *et al.*, 2014. Comparing environmental impacts of different forest management scenarios for maritime pine biomass production in France. *Journal of Cleaner Production*, 64, pp. 356-367.

Greenberg, R. *et al.*, 2014. The distribution and conservation of birds of coastal salt marshes. In: B. Maslo and J.L Lockwood, (ed.) 2014. *Coastal Conservation*. Cambridge: Cambridge University Press. pp. 180-242.

Greene, W.H. and Hensher, D.A., 2010. Does scale heterogeneity across individuals matter? An empirical assessment of alternative logit models. *Transportation*, 37(3), pp. 413-428.

Greenhill, L. *et al*., 2020. Does marine planning enable progress towards adaptive governance in marine systems? Lessons from Scotland's regional marine planning process. *Maritime Studies*, 19(3), pp. 299-315.

Greer, S. and Jarman, H., 2008. Policy Styles and Devolution. In: A. Trench, ed. 2008. *The State of the Nations*. London: Imprint Academics. pp. 167-196.

Grilli, G. *et al.*, 2022. The role of choice experiments in natural capital accounting approaches: fast track versus simulated exchange value in the Deben Estuary saltmarshes. *Journal of Environmental Planning and Management*, 65(7), pp. 1281-1300.

Grip, K. and Blomqvist, S., 2021. Marine spatial planning: Coordinating divergent marine interests. *Ambio*, 50(6)., pp. 1172-1183.

Grunewald, K. and Bastian, O., 2015. Development and Fundamentals of the ES Approach. In: K. Grunewald and O. Bastian, eds. 2015. *Ecosystem Services: Concept, Methods and Case Studies*. Berlin: Springer. pp. 13-34.

Gulev, S.K. *et al.*, 2021. Changing State of the Climate System. In: V. Masson-Delmotte *et al.* (eds.) 2021. *Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge: Cambridge University Press. pp. 287-422.

Halkidi, M. *et al*., 2016. Method-Independent Indices for Cluster Validation and Estimating the Number of Clusters. In: C. Hennig *et al*. ed. 2016. *Handbook of Cluster Analysis*. Boca Raton: CRC Press, Taylor and Francis Group. pp. 595-618.

Hanley, N. and Barbier, E.B., 2009. *Pricing Nature - Cost-Benefit Analysis and Environmental Policy*. Cheltenham: Edward Elgar.

Hanley, N. and Czajkowski, M., 2019. The role of stated preference valuation methods in understanding choices and informing policy. *Review of Environmental Economics and Policy*, 13(2), pp. 248-266.

Hanley, N. and Munro, A., 1992. *The Effects of Information in Contingent Markets for Environmental Goods*. Queen's University Discussion Papers, Queen's University, Ontario.

Hanley, N. *et al.*, 2007. *Environmental Economics in Theory and Practice*. 2nd ed. Basingstoke: Palgrave Macmillan.

Hanley, N. *et al.*, 2013. *Introduction to Environmental Economics*. 2nd ed. Oxford: Oxford University Press.

Harvey, R.J. *et al.*, 2019. No detectable broad-scale effect of livestock grazing on soil blue-carbon stock in salt marshes. *Frontiers in Ecology and Evolution*, 7, pp. 1-12.

Haynes, T.A., 2016a. *Scottish saltmarsh survey national report*. Scottish Natural Heritage Commissioned Report No. 786.

Haynes, T.A., 2016b. *Saltmarsh Survey of Scotland*. [online] Available at: <https://data.gov.uk/dataset/5ed5df7a-a81f-4dba-9cd0-7e03c437aa71/saltmarsh-survey-of-scotland> [Accessed 05 May 2019].

Hennig, C. and Meila, M., 2016. Cluster Analysis: An Overview. In: C. Hennig *et al.* ed. 2016. *Handbook of Cluster Analysis*. Boca Raton: CRC Press, Taylor and Francis Group. pp. 1-20.

Hensher, D. A. *et al.*, 2015. *Applied Choice Analysis*. 2nd ed. Cambridge: Cambridge University Press.

Herr, D. *et al.*, 2019. An analysis of the potential positive and negative livelihood impacts of coastal carbon offset projects. *Journal of Environmental Management*, 235, pp. 463-479.

Hess, S. and Palma, D., 2019. Apollo: a flexible, powerful and customisable freeware package for choice model estimation and application. *Journal of Choice Modelling*, 32, pp. 1-26.

Hess, S. and Palma, D., 2022. *Apollo: A flexible, powerful and customisable freeware package for choice model estimation and application*. [online] Available at: <http://www.apollochoicemodelling.com/manual.html> [Accessed 01 June 2022].

Hess, S. and Train, K., 2017. Correlation and scale in mixed logit models. *Journal of Choice Modelling*, 23, pp. 1-8.

Hilmi, N. *et al*., 2021. The role of blue carbon in climate change mitigation and carbon stock conservation. *Frontiers in Climate*, 3, pp. 1-18.

Himes-Cornell *et al.*, 2018. Valuing ecosystem services from blue forests: A systematic review of the valuation of salt marshes, sea grass beds and mangrove forests. *Ecosystem Services*, 30(A), pp. 36-48.

HM Treasury, 2013. *The Green Book: Appraisal and Evaluation in Central Government*. [online] Available at: <https://www.gov.uk/government/publications/the-green-book-appraisal-and-evaluation-in-central-governent> [Accessed 13 August 2022].

HM Treasury, 2020. *The Green Book. Central Government Guidance on Appraisal and Evaluation*. [online] Available at: <https://www.gov.uk/government/publications/the-green-book-appraisal-and-evaluation-in-central-governent> [Accessed 16 August 2021].

Hole, A.R. and Kolstad, J.R., 2012. Mixed logit estimation of willingness to pay distributions: A comparison of models in preference and WTP space using data from a health-related choice experiment. *Empirical Economics*, 42(2), pp. 445-469.

Houghton, John, 2015. *Global Warming: The complete briefing.* 5th ed. Cambridge: Cambridge University Press.

Howard, J. *et al.* (eds.), 2014. *Coastal Blue Carbon: Methods for assessing carbon stocks and emissions factors in mangroves, tidal salt marshes, and seagrass meadows.* [online] Available at: <https://www.cifor.org/library/5095/coastal-blue-carbon-methods-for-assessing-carbon-stocks-and-emissions-factors-in-mangroves-tidal-salt-marshes-and-seagrasses/> [Accessed 16 August 2021].

Howard, J. *et al.*, 2017. Clarifying the role of coastal and marine systems in climate mitigation. *Frontiers in Ecology and the Environment*, 15(1), pp. 42-50.

Imamura, K. *et al.*, 2020. Valuation of coral reefs in Japan: Willingness to pay for conservation and the effect of information. *Ecosystem Services*, 46, pp. 1-13.

Instrumental Criteria Sub-committee, 2008. CHNS Elemental Analysers. *AMC Technical Briefs*, 29, pp. 1-2.

Interis, M.G. and Petrolia, D.R., 2016. Location, location, habitat: How the value of ecosystem services varies across location and by habitat. *Land Economics*, 92(2), pp. 292-307.

IPCC, 2001. *Climate Change 2001: Synthesis Report. A Contribution of Working Groups I, II, and III to the Third Assessment Report of the Intergovernmental Panel on Climate Change*. [online] Available at: <https://www.ipcc.ch/report/ar3/syr/> [Accessed 13 August 2022].

IPCC, 2010. *Guidance Note for Lead Authors of the IPCC Fifth Assessment Report on Consistent Treatment of Uncertainties.* [online] Available at: <https://www.ipcc.ch/site/assets/uploads/2017/08/AR5_Uncertainty_Guidance_Note.pdf> [Accessed 13 August 2022].

IPCC, 2013. *Climate Change 2013: The Physical Science Basis*. *Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change.* [online] Available at: <https://www.ipcc.ch/report/ar5/wg1/> [Accessed 24 May 2022].

IPCC, 2014a. *2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands.* [online] Available at: <https://www.ipcc-nggip.iges.or.jp/public/wetlands/> [Accessed 03 May 2019].

IPCC, 2014b. *Climate Change 2014 Synthesis Report: Fifth Assessment Report*. [online] Available at: <https://ar5-syr.ipcc.ch/index.php> [Accessed 24 May 2022].

IPCC, 2018. *Global Warming of 1.5°C: Proposed outline of the special report in 2018 on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change sustainable development, and efforts to eradicate poverty*. [online] Available at: <https://www.ipcc.ch/sr15/> [Accessed 19 April 2020].

IPCC, 2019. *IPCC Special Report on the Ocean and Cryosphere in a Changing Climate*. [online] Available at: <https://www.ipcc.ch/srocc/download/#pub-full> [Accessed 27 August 2022].

IPCC, 2021. Summary for Policymakers. In: *Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*. [online] Available at: <https://www.ipcc.ch/report/ar6/wg1/> [Accessed 27 August 2022].

IUCN, 2022. *Nature-based Solutions*. [online] Available at: <https://www.iucn.org/theme/nature-based-solutions> [Accessed 13 August 2022].

Jessoe, L. and Rapson, D., 2014. Knowledge is (less) power: experimental evidence from residential energy use. *American Economic Review*. 104(4), 1417-1438.

Johnston, R.J. *et al.*, 2017. Contemporary guidance for stated preference studies. *Journal of the Association of Environmental and Resource Economists*, 4(2), pp. 319-405.

Johnston, R.J. *et al.*, 2018. Using ecosystem service values to evaluate tradeoffs in coastal hazard adaptation. *Coastal Management*, 46(4), pp. 259-277.

Jones, L. *et al.*, 2011. *UK National Ecosystem Assessment – Technical Report: Chapter 11: Coastal Margins*. [online] Available at: <http://uknea.unep-wcmc.org/Resources/tabid/82/Default.aspx> [Accessed 22 June 2022].

Keating, M., 2005. *The Government of Scotland: Policy Making after Devolution*. 2nd ed. Edinburgh: Edinburgh University Press.

Kershaw, J. *et al.*, 2022. *Briefing: Blue Carbon*. [online] Available at: <https://www.theccc.org.uk/wp-content/uploads/2022/03/CCC-Briefing-Blue-Carbon-FINAL.pdf> [Accessed 7 June 2022].

King, S.E. and Lester, J.N., 1995. The value of salt marsh as a sea defence. *Marine Pollution Bulletin*, 30(3), pp. 180-189.

Kirwan, M.L. *et al.*, 2016. Overestimation of marsh vulnerability to sea level rise. *Nature Climate Change*, 6(3), pp. 253-260.

Klenert, D. *et al.*, 2018. Making carbon pricing work for citizens. *Nature Climate Change,* 8, pp. 669-677.

Kotchen, M. *et al.*, 2013. Willingness-to-pay and policy-instrument choice for climate-change policy in the United States. *Energy Policy*, 55, pp. 617-625.

Kuwae, T. *et al.*, 2022. Implementation of blue carbon offset crediting for seagrass meadows, macroalgal beds, and macroalgae farming in Japan. *Marine Policy*, 138, pp. 1-11.

Ladd, C.J.T. *et al.*, 2019. Sediment supply explains long-term and large-scale patterns in salt marsh lateral expansion and erosion. *Geophysical Research Letters*, 46(20), pp. 11178-11187.

Lagarde, M., 2013. Investigating attribute non-attendance and its consequences in choice experiments with latent class models. *Health Economics*, 22(5), pp. 554-567.

Lai, Q. *et al*., 2022. Research Development, Current Hotspots, and Future Directions of Blue Carbon: A Bibliometric Analysis. *Water*, 14(8), pp. 1-17.

Lancaster, K.J., 1966. A new approach to consumer theory. *Journal of Political Economy*, 74(2), pp. 132–157.

Lancsar, E. *et al.*, 2017. Discrete choice experiments: A guide to model specification, estimation and software. *PharmacoEconomics*, 35(7), pp. 697-716.

LaRiviere, J. *et al.*, 2014. The value of familiarity: Effects of knowledge and objective signals on willingness to pay for a public good. *Journal of Environmental Economics and Management*, 68(2), pp. 376-389.

Lewis, C.J.E. *et al.*, 2017. Variability and vulnerability of coastal 'Blue Carbon' stocks: A case study from southeast australia. *Ecosystems*, 21(2), pp. 263-279.

Lockwood, B. and Drakeford, B.M., 2021. The value of carbon sequestration by saltmarsh in Chichester Harbour, United Kingdom. *Journal of Environmental Economics and Policy*, pp. 1-15.

Lockwood, J.L and Maslo, B., 2014. A research agenda for coastal biodiversity and conservation. In: B. Maslo and J.L Lockwood, ed. 2014. *Coastal Conservation*. Cambridge: Cambridge University Press. pp. 371-379.

Louviere, J.J., 2001. Choice Experiments: An Overview of Concepts and Issues. In: J. Bennett and R. Blamey, eds. 2001. *The Choice Modelling Approach to Environmental Valuation.* Cheltenham: Edward Elgar. pp. 13-36.

Louviere, J.J. *et al.*, 2000. *Stated Choice Methods: Analysis and Applications*. Cambridge: Cambridge University Press.

Luce, R.D., 1959. *Individual Choice Behaviour.* New York: Wiley.

Luisetti, T. *et al.*, 2011. Coastal and marine ecosystem services valuation for policy and management: Managed realignment case studies in England. *Ocean* & *Coastal Management*, 54(3), pp. 212-224.

Luisetti, T. *et al.*, 2013. Valuing the European ‘coastal blue carbon’ storage benefit. *Marine Pollution Bulletin*, 71(1-2), pp. 101-106.

Luisetti, T. *et al.*, 2014. Coastal zone ecosystem services: From science to values and decision making; a case study. *Science of the Total Environment*, 493, pp. 682-693.

Luisetti, T. *et al.*, 2015. Why Value ‘Blue Carbon’? In: R.K. Turner and M. Schaafsma, eds. 2015. *Coastal zones ecosystem services: from science to values and decision making*. Cham: Springer. pp. 191-206.

Lüthi, D. *et al.*, 2008. High-resolution carbon dioxide concentration record 650,000-800,000 years before present. *Nature*, 453, pp. 379-382.

MacDonald, M.A. *et al.*, 2017. Benefits of coastal managed realignment for society: Evidence from ecosystem service assessments in two UK regions. *Estuarine, Coastal and Shelf Science*, 244, pp. 1-11.

Macreadie, P.I. *et al.*, 2012. Paleoreconstruction of estuarine sediments reveal human-induced weakening of coastal carbon sinks. *Global Change Biology*, 18(3), pp. 891-901.

Mariel, P. *et al.* 2021. *Environmental Valuation with Discrete Choice Experiments: Guidance on Design, Implementation and Data Analysis*. Cham: Springer Nature.

Marine and Coastal Access Act 2009. [online] Available at: <https://www.legislation.gov.uk/ukpga/2009/23/contents> [Accessed 05 August 2022].

Marine (Scotland) Act 2010. [online] Available at: <https://www.legislation.gov.uk/asp/2010/5/pdfs/asp_20100005_en.pdf> [Accessed 4 April 2022].

Marine Scotland, 2021. *National Marine Plan Review 2021: Three Year Report on the effectiveness of Scotland's National Marine Plan*. [online] Available at: https://marine.gov.scot/data/national-marine-plan-review-2021 [Accessed 14 August 2022].

Markandya, A., 2005. *Economic Principles and Overview of Valuation Methods for Environmental Impacts*. [online] Available at: <https://www.semanticscholar.org/paper/Economic-Principles-and-Overview-of-Valuation-for-Markandya/835c88626e3428c8b8a5dc7b28de31b828031451> [Accessed 29 August 2022].

Marshall, C. and Rossman, G.B., 2006. *Designing Qualitative Research*. 4th ed. Thousand Oaks: Sage Publications.

Martino, S. and Kenter, J.O., 2021. The ecosystem approach. In: S. Martino and J.O. Kenter. ed. 2021. *Sustainable governance of marine and coastal heritage: Methods, tools, and approaches.* Brussels: PERICLES, European Commission.

McCauley, D.J., 2006. Selling out on nature. *Nature*, 443, pp. 27-28.

McFadden, D., 1973. Conditional logit analysis of qualitative choice behaviour. In: P. Zarembka, ed. 1974. *Frontiers in Econometrics*. New York: Academic Press. pp. 105-142.

McKinley, E. *et al.*, 2020. Uses and management of saltmarshes: A global survey. *Estuarine, Coastal and Shelf Science*, 243, 106840, pp. 1-20.

McLeod, E. *et al.*, 2011. A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO2. *Frontiers in Ecology and the Environment*, 9(10), pp. 552-560.

Mcowen, C.J. *et al.*, 2017. A global map of saltmarshes. *Biodiversity Data Journal*, 5, pp. 1-13,

Merk, C. *et al*., 2022. The need for local governance of global commons: The example of blue carbon ecosystems. *Ecological Economics*, 201, pp. 1-8.

Millennium Ecosystem Assessment (MEA), 2005. *Millennium Ecosystem Assessment: Ecosystems and Human Well-being: Synthesis.* Washington, DC: Island Press.

Miller, B.W. and Morisette, J.T., 2014. Integrating research tools to support the management of social-ecological systems under climate change. *Ecology and Society*, 19(3), pp. 1-12.

Miller, L.C. *et al*. Carbon accumulation and storage across contrasting saltmarshes of Scotland [Manuscript In Revision]. School of Geography and Sustainable Development, University of St Andrews.

Mills, F. *et al.*, 2017. *Clyde Marine Region Assessment*. [online] Available at: https://www.clydemarineplan.scot/marine-planning/clyde-regional-marine-plan/#assessment [Accessed 10 August 2022].

Mitchell, B., 2002. *Resource and Environmental Management*. 2nd ed. Harlow: Prentice Hall.

Muddiman, S., 2019. *Ecosystem Services: Economics and Policy*. Cham: Palgrave Macmillan.

Munro, A. and Hanley, N.D., 2001. Information, Uncertainty, and Contingent Valuation. In: I.J. Bateman and K.G. Willis. ed. 2011. *Valuing Environmental Preferences: Theory and Practice of the Contingent Valuation Method in the US, EU, and Developing Countries*. Oxford University Press, Oxford, pp. 258–274.

Narayan, S. *et al.*, 2016. The effectiveness, costs and coastal protection benefits of natural and nature-based defences. *PLoS ONE*, 11(5), pp. 1-17.

NatureScot, 2021a. *Sites of Special Scientific Interest (SSSIs)*. [online] Available at: <https://www.nature.scot/professional-advice/protected-areas-and-species/protected-areas/national-designations/sites-special-scientific-interest-sssis> [Accessed 1 August 2022].

NatureScot, 2021b. *Sites of Special Scientific Interest (SSSIs) – Consents*. [online] Available at: <https://www.nature.scot/professional-advice/protected-areas-and-species/protected-areas/national-designations/sites-special-scientific-interest-sssis-consents> [Accessed 1 August 2022].

NatureScot, 2022. *NatureScot: Scotland's Nature Agency.* [online] Available at: <https://www.nature.scot/> [Accessed 6 May 2022].

NatureScot, 2023. *Saltmarsh: Saltmarsh and Climate Change*. [online] Available at: <https://www.nature.scot/landscapes-and-habitats/habitat-types/coast-and-seas/coastal-habitats/saltmarsh> [Accessed 7 February 2023].

Needham, K. *et al.*, 2018. What is the causal impact of information and knowledge in stated preference studies? *Resource and Energy Economics*, 54, pp. 69-89.

NERC C-SIDE, 2022. *The C-SIDE project*. Online. Available at: <https://www.c-side.org/about-1> [Accessed 30 May 2022].

Newing, H., 2011. *Conducting Research in Conservation: Social Science Methods and Practice.* London: Routledge.

Nieuwenhuize, J. *et al.*, 1994. Rapid analysis of organic carbon and nitrogen in particulate materials, *Marine Chemistry*, 45(3), pp. 217-244.

NMP, 2015. *Scotland's National Marine Plan*. [online] Available at: https://www.gov.scot/publications/scotlands-national-marine-plan/ [Accessed 14 August].

Nordhaus, W.D., 2017. Revisiting the social cost of carbon. *Proceedings of the National Academy of Sciences of the United States of America (PNAS)*, 114(7), pp. 1518-1523.

Nylund-Gibson, K. and Choi, A.Y., 2018. Ten frequently asked questions about latent class analysis. *Translational Issues in Psychological Science*, 4(4), pp. 440-461.

Oberski, D. 2016. Mixture Models: Latent Profile and Latent Class Analysis. In: J. Robertson and M. Kaptein, eds. 2016. *Modern Statistical Methods for HCI*. Cham: Springer. pp. 275-287.

Ogden, J.C. *et al.*, 1999. Environmental management scenarios: Ecological implications. *Urban Ecosystems*, 3(3), pp. 279-303.

Olsen, E. *et al.*, 2014. Integration at the round table: Marine spatial planning in multi-stakeholder settings. *PLoS ONE*, 9(10), pp. 1-9.

Pechanec, V. *et al*., 2022. Selected aspects of carbon stock assessment in aboveground biomass. *Land*, 11(1), pp. 1-16.

Pendleton, L. *et al.*, 2012. Estimating global ‘‘Blue Carbon’’ emissions from conversion and degradation of vegetated coastal ecosystems. *PLoS ONE*, 7(9), pp. 1-7.

Perni A. and Martínez-Paz, J.M., 2017. Measuring conflicts in the management of anthropized ecosystems: Evidence from a choice experiment in a human-created Mediterranean wetland. *Journal of Environmental Management*, 203(1), pp. 40-50.

Peterson, G.D. *et al*., 2003. Scenario planning: a tool for conservation in an uncertain world. *Conservation Biology*, 17(2), pp. 358-366.

Petrolia, D.R. *et al.*, 2014. America's wetland? A national survey of willingness to pay for restoration of Louisiana's coastal wetlands. *Marine Resource Economics*, 29(1), pp. 17-37.

Pill, M., 2022. Towards a funding mechanism for loss and damage from climate change impacts. *Climate Risk Management*, 35, pp. 1-10.

Porter, J.S. *et al.*, 2020. Blue carbon audit of Scottish waters. *Scottish Marine and Freshwater Science*, 11(3), pp. 1-96.

Purvis, B. *et al.*, 2019. Three pillars of sustainability: in search of conceptual origins. *Sustainability Science*, 14(3), pp. 681-695.

Reed, M., 2009. A civic sort-of science: Addressing environmental managerialism in Canadian biosphere reserves. *Environments*, 36(3), pp. 17-36.

Remoundou, K. *et al.*, 2015. Valuing climate change mitigation: A choice experiment on a coastal and marine ecosystem. *Ecosystem Services*, 11, pp. 87-94.

Rendón, O. *et al.*, 2022. Heterogeneity of values for coastal flood risk management with nature-based solutions. *Journal of Environmental Management*, 304, pp. 1-10.

Richards, L., 2015. *Handling Qualitative Data: A Practical Guide*. 3rd ed. Los Angeles: SAGE.

Ritchie, H. and Roser, M., 2019. *Our World in Data: Land Use*. [online] Available at: <https://ourworldindata.org/land-use> [Accessed 1 September 2022].

Robinson, D.A. *et al*., 2022. Analytical modelling of soil porosity and bulk density across the soil organic matter and land-use continuum. *Scientific Reports*, 12, pp. 1-14.

Robson, C., McCartan K., 2016. *Real World Research*. 4th ed. Chichester: John Wiley & Sons.

Rogers, K. *et al*., 2019. Wetland carbon storage controlled by millennial-scale variation in relative sea-level rise. *Nature*, 567, pp. 91-95.

Ruranska, P. *et al*., 2020. *Dry bulk density, loss on ignition and organic carbon content of surficial soils from Scottish salt marshes, 2018-2019*. NERC Environmental Information Data Centre. (Dataset). https://doi.org/10.5285/81a1301f-e5e2-44f9-afe0-0ea5bb08010f.

Samples, K. *et al.*, 1986. Information disclosure and endangered species valuation. *Land Economics*, 62(3), pp. 306-312.

Sanderson, I., 2010. Evidence, learning and intelligent government: Reflections on developments in Scotland. *German Policy Studies*, 6(2), pp. 53-85.

Sandorf, E.D. *et al.*, 2016. Valuing unfamiliar and complex environ- mental goods: a comparison of valuation workshops and internet panel surveys with videos. *Ecological Economics*, 129, pp. 50-61.

Sangiuliano, S.J., 2019. Analysing the prevalence of ecosystem services in the objectives and policies of Scotland's National Marine Plan. *Marine Policy*, 104, pp. 37-52.

SBCF, 2022a. *About the Scottish Blue Carbon Forum.* [online] Available at: <https://www.bluecarbon.scot/about> [Accessed 24 August 2022].

SBCF, 2022b. *What is Blue Carbon?* [online] Available at: <https://www.bluecarbon.scot/what-is-blue-carbon> [Accessed 10 June 2022].

Ščasný, M. *et al.*, 2017. Public acceptability of climate change mitigation policies: a discrete choice experiment. *Climate Policy*, 17(S1), pp. 111-130.

Schuerch, M. *et al.*, 2018. Future response of global coastal wetlands to sea-level rise. *Nature*, 561, pp. 231-234.

Scotland Act 1998. [online] Available at: <https://www.legislation.gov.uk/ukpga/1998/46/contents> [Accessed 05 August 2022].

Scott, D.B. and Greenberg, D.A., 1983. Relative sea-level rise and tidal development in the Fundy tidal system. *Canadian Journal of Earth Sciences*, 20(10), pp. 1554-1564.

Scottish Government, 2018a. *National Marine Plan Review 2018: three-year report*. [online] Available at: <https://www.gov.scot/publications/national-marine-plan-review-2018-three-year-report-implementation-scotlands/> [Accessed 14 August 2022].

Scottish Government, 2018b. *Scottish Natural Heritage: factsheet.* [online] <https://www.gov.scot/publications/scottish-natural-heritage-factsheet/> [Accessed 6 May 2022].

Scottish Government, 2020a. *Scottish Planning Policy*. [online] Available at: <https://www.gov.scot/publications/scottish-planning-policy/pages/11/#climate_mig> [Accessed 14 August 2022].

Scottish Government, 2020b. *EU Exit: marine environmental legislation in Scotland*. [online] Available at: <https://www.gov.scot/publications/eu-exit-marine-environmental-legislation-scotland-2/pages/4/> [Accessed 14 August 2022].

Scottish Government, 2020c. *Scotland's People Annual Report: Scottish Household Survey*. [online] Available at: <https://www.gov.scot/publications/scotlands-people-annual-report-results-2018-scottish-household-survey/pages/7/> [Accessed 31 July 2020].

Scottish Government, 2020d. *Scotland and the sustainable development goals: a national review to drive action*. [online] Available at: <https://www.gov.scot/publications/scotland-sustainable-development-goals-national-review-drive-action/documents/> [Accessed 29 April 2022].

Scottish Government, 2021. *Scottish Greenhouse Gas statistics: 1990-2019*. [online] Available at: <https://www.gov.scot/publications/scottish-greenhouse-gas-statistics-1990-2019/> [Accessed 14 August 2022].

Scottish Government, 2022. *Scotland. Let's do net zero.* [online] Available at: <https://www.netzeronation.scot/> [Accessed 15 April 2022].

Secor, A.J., 2010. Social Surveys, Interviews, and Focus Groups. In: B. Gomez and J.P. Jones III, ed. 2010. *Research Methods in Geography*. Chichester: Wiley-Blackwell. pp. 194-205.

Seddon, N. *et al*., 2020. Understanding the value and limits of nature-based solutions to climate change and other global challenges. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 375(1794), pp. 1-12.

SEFARI, 2023. *Natural Asset Inventory and Natural Capital Accounts*. [online] Available at: <https://sefari.scot/research/objectives/natural-asset-inventory-and-natural-capital-accounts> [Accessed, 24. February 2023].

Segal, D., 2022. Who Will Profit from Saving Scotland's Bogs? *The New York Times* [online] Available at: <https://www.nytimes.com/interactive/2022/05/05/headway/scotland-peatlands-climate-change.html> [Accessed 22 June 2022].

Shapansky, B. *et al.*, 2008. Assessing information provision and respondent involvement effects on preferences. *Ecological Economics*, 65(3), pp. 626-635.

Sheehan, L. *et al.*, 2019. Blue carbon: An additional driver for restoring and preserving ecological services of coastal wetlands in Tampa Bay (Florida, USA). *Wetlands*, 39(6), pp. 1317-1328.

Shoyama, K. *et al.*, 2013. Public preferences for biodiversity conservation and climate-change mitigation: A choice experiment using ecosystem services indicators. *Land Use Policy*, 34, pp. 282-293.

Shreedhar, G. and Mourato, S., 2019. Experimental evidence on the impact of biodiversity conservation videos on charitable donations. *Ecological Economics*, 158, pp. 180-193.

Silliman, B.R., 2014. Salt marshes. *Current Biology*, 24(9), pp. R348-R350.

Slater, A. and Claydon, J., 2020. Marine spatial planning in the UK: A review of the progress and effectiveness of the plans and their policies. *Environmental Law Review*, 22(2), pp. 85-107.

Smeaton, C. *et al.*, 2020. Coring and compaction: Best practice in blue carbon stock and burial estimations. *Geoderma*, 364, pp. 1-7.

Smeaton, C. *et al.*, 2021. *Organic carbon density of surficial soils across Scottish saltmarshes*. Online. Available at: <https://data.marine.gov.scot/dataset/organic-carbon-density-surficial-soils-across-scottish-saltmarshes> [Accessed 30 May 2022].

Smith, H.D. *et al.*, 2011. The integration of land and marine spatial planning. *Journal of Coastal Conservation*, 15(2), pp. 291-303.

SNP, 2022. What actions will the SNP take to combat climate change? [online] Available at: <https://www.snp.org/policies/what-actions-will-the-snp-take-to-combat-climate-change/> [Accessed 1 August 2022].

Spivak, A. *et al.*, 2019. Global-change controls on soil-carbon accumulation and loss in coastal vegetated ecosystems. *Nature Geoscience*, 12(9), pp. 685-692.

Stata, 2019. *Multivariate Statistics Reference Manual*. [pdf] StataCorp LLC. Available at: <https://www.stata.com/bookstore/multivariate-statistics-reference-manual/> [Accessed 17 September 2020].

Stern, N., 2006. *The Economics of Climate Change: The Stern Review*. Cambridge: Cambridge University Press.

Stratford, E. and Bradshaw, M., 2021. Rigorous and Trustworthy: Qualitative Research Design, In: I. Hay and M. Cope, ed. 2021. *Qualitative Research Methods in Human Geography*. Don Mills: Oxford University Press. pp. 92-106.

Sutton-Grier, A.E. *et al*., 2014. Incorporating ecosystem services into the implementation of existing U.S. natural resource management regulations: Operationalizing carbon sequestration and storage. *Marine Policy*, 43, pp. 246-253.

Swait J. and Louviere J., 1993. The role of the scale parameter in the estimation and comparison of multinomial logit models. *Journal of Marketing Research*, 30(3), pp. 305-314.

The Blue Carbon Initiative, 2022. *Global Distribution of Blue Carbon Ecosystems*. [online] Available at: <https://www.thebluecarboninitiative.org/> [Accessed 7 June 2022].

The Economics of Ecosystems and Biodiversity Project (TEEB), 2010. *Mainstreaming the Economics of Nature: A Synthesis of the Approach, Conclusions and Recommendations of TEEB*. London and Washington: Earthscan.

The World Bank, 2018. *Data: Forest area (% of land area).* [online] Available at: <https://data.worldbank.org/indicator/AG.LND.FRST.ZS> [Accessed 10 May 2018].

The World Bank, 2021. *State and Trends of Carbon Pricing 2021*. [online] Available at: <https://openknowledge.worldbank.org/handle/10986/35620> [Accessed 14 August 2022].

Tienhaara, A. *et al.*, 2021. Information use and its effects on the valuation of agricultural genetic resources. *Land Economics*, 98(2), 337-354.

Tkac, J., 1998. The effects of information on willingness-to-pay values of endangered species. *American Journal of Agricultural Economics*, 80(5), pp. 1214-1220.

Tol, R., 2011. The social cost of carbon. *Annual Review of Resource Economics*, 3, pp. 419-443.

Tol, R.S.J., 2014. *Climate Economics: Economic Analysis of Climate, Climate Change and Climate Policy*. Cheltenham: Edward Elgar.

Train, K. and Weeks, M., 2005. Discrete Choice Models in Preference Space and Willing-to-pay Space. In: R. Scarpa and A. Alberini, eds. 2005. *Applications of Simulation Methods in Environmental and Resource Economics*. Dordrecht: Springer Netherlands. pp. 1-16.

Tröels-Smith, J., 1955. *Characterization of unconsolidated sediments*. Dansk Geologiske Undersøgelse, Reitzels Forlag.

UK Government, 2021. *Final Report – The Economics of Biodiversity: The Dasgupta Review*. [online] Available at: <https://www.gov.uk/government/publications/final-report-the-economics-of-biodiversity-the-dasgupta-review> [Accessed 31 August 2022].

UNFCCC, 2022a. *The Paris Agreement*. [online] Available at: <https://unfccc.int/process-and-meetings/the-paris-agreement/the-paris-agreement> [Accessed 14 August 2022].

UNFCCC, 2022b. *The carbon market*. [online] Available at: <https://unfccc.int/process/conferences/pastconferences/bali-climate-change-conference-december-2007/statements-and-resources/the-carbon-market> [Accessed 14 August 2022].

Valiela, I. *et al.*, 2018. Transient coastal landscapes: Rising sea level threatens salt marshes. *Science of the Total Environment*, 640-641, pp. 1148-1156.

van Ardenne, L.B. *et al.*, 2018. The importance of geomorphic context for estimating the carbon stock of salt marshes. *Geoderma*, 330, pp. 264-275.

Verardo, D.J. *et al.*, 1990. Determination of organic carbon and nitrogen in marine sediments using the Carlo Erba NA-1500 analyzer. *Deep Sea Research Part A. Oceanographic Research Papers*, 37(1), pp. 157-165.

Warren, C., 2009. *Managing Scotland's Environment*. 2nd ed. Edinburgh: Edinburgh University Press.

Watson, A., 2021. "Placing" Participant Observation, In: I. Hay and M. Cope, ed. 2021. *Qualitative Research Methods in Human Geography*. Don Mills: Oxford University Press. pp. 125-147.

Watson, S.C.L. *et al.*, 2020. Assessing the natural capital value of water quality and climate regulation in temperate marine systems using a EUNIS biotope classification approach. *Science of the Total Environment*, 744, pp. 1-11.

Wedding, L.M. *et al.*, 2021. Incorporating blue carbon sequestration benefits into sub-national climate policies. *Global Environmental Change*, 69, pp. 1-9.

Weller, B.E. *et al.*, 2020. Latent class analysis: A guide to best practice. *Journal of Black Psychology*, 46(4), pp. 287-311.

Whitehead, J. and Blomquist, G., 1991. Measuring contingent values for wetlands: Effects of information about related environmental goods. *Water Resources Research*, 27(10), pp. 2523-2531.

Willis, K.G., 2014. The Use of Stated Preference Methods to Value Cultural Heritage. In: V.A. Ginsburgh and D. Throsby, eds. 2014. *Handbook of the Economics of Art and Culture*. Oxford: Elsevier. Volume 2, pp. 145-181.

Wilson, G.A., 2009. Rethinking environmental management - Ten years later: A view from the author. *Environments*, 36(3), pp. 3-15.

Wilson, G.A. and Bryant, R.L., 1997. *Environmental Management: New directions for the Twenty-First Century*. London: UCL Press.

Wollenberg, J.T. *et al.*, 2018. Rapid carbon accumulation following managed realignment on the Bay of Fundy. *PLoS ONE*, 13(3), pp. 1-14.

Wylie, L. *et al.*, 2016. Keys to successful blue carbon projects: Lessons learned from global case studies. *Marine Policy*, 65, pp. 76-84.

Yoo, H.I, 2020. lclogit2: An enhanced command to fit latent class conditional logit models. *Stata Journal*, 20(2), pp. 405-425.

Ziegler, A., 2017. Political orientation, environmental values, and climate change beliefs and attitudes: an empirical cross country analysis. *Energy Economics*, 63, pp. 144-153.

APPENDIX A

Table A.1: Loch of Stenness and Waulkmill Bay saltmarsh soil stratigraphy according to the Tröels-Smith (1995) classification scheme. Peat and transitional layers represent saltmarsh soil indicating the saltmarsh depth.

Core	Depth Interval (cm)	Description
	Loch of Stenness	
STEN01	1-10	Dark peat
	10-21	Organic silt, humified peat
	21-28	Sandy Mud
STEN02	1-13	Peat
	13-20	Silty organics
	20-25	Sandy mud
STEN03	0-8	Peat
	8-22	Organic silt/silty organics
	22-32	Sandy mud with organics
STEN04	0-6	Peat
	6-10	Transition
	10-30	Silty organics organic rich
STEN05	0-4	Peat/silty peat
	4-28	Silty organics
STEN06	0-30	Peat
	20-30	Dark humified peat
STEN07	0-21	Peat
	21-38	Sandy mud, some organics
STEN08	0-7	Peat
	7-15	Transition
	15-18	Sandy mud
	18-22	Stone
	22-23	Sandy mud
STEN09	0-24	Peat, organic
STEN10	0-8	Peat
	8-22	Transition
	22-30	Sandy mud with some organics
STEN11	0-12	Peat
	12-18	Sandy mud with some organics
STEN12	0-6	Peat
	6-23	Transition
	23-27	Grey clay
STEN13	0-12	Peat
	12-16	Transition
	16-46	Sandy mud, some organics
STEN14	0-14	Peat
	14-23	Blue silty clay
STEN15	0-9	Peat
	9-34	Silt with organics
STEN16	0-51	Peat
STEN17	0-21	Peat
	21-26	Blue silt
	Waulkmill Bay	
WAULK01	0-69	Peat, sandier with increasing depth
WAULK02	0-29	Peat

	29-52	Sandier, pocket of sand at 52 cm depth
WAULK03	0-44 44-51 51-59	Peat Grey clay Peat, sandier
WAULK04	0-12 12-18 18-51	Peat Transition Grey clay
WAULK05	0-9 9-22 22-44	Peat Transition with organics Sandy silt with organics
WAULK06	0-11 11-35	Peat Sandy silt with organics
WAULK07	0-10 10-48	Peat Sandy silt
WAULK08	0-10 10-48	Peat Sandy silt
WAULK09	0-18 18-34 34-40 40-67	Peat Humified peat Transition Silty sand
WAULK10	0-22 22-54	Peat Peat with sand and clay
WAULK11	0-20 20-33 33-55	Peat Transition, some organics, silt, and sand Silty sand

APPENDIX B

Table B.1: Balancing Tests to test whether the random sampling was successful. The Chi-2 test results indicate that the randomisation was successful since there is no significant relationship between any variable with the treatment or control group ($p > 0.05$).

Variable Name	**Variable Descriptor**	**Chi-2 test result**	**Variable Name**	**Variable Descriptor**	**Chi-2 test result**
Sex	q26	p = 0.677	Heard about saltmarshes	q5	P = 0.819
Age	q27	p = 0.942	Have visited a saltmarsh	q6	P = 0.531
Education	q28	p = 0.397	NEP scale item 1	q22_1	P = 0.165
Marital Status	q29	p = 0.108	NEP scale item 2	q22_2	P = 0.920
Children	q30	p = 0.620	NEP scale item 3	q22_3	P = 0.738
Income	q31	p = 0.491	NEP scale item 4	q22_4	P = 0.700
Employment	q32	p = 0.231	NEP scale item 5	q22_5	P = 0.945
Taxpayer	q18	p = 0.493	NEP scale item 6	q22_6	P = 0.396
Election Participation	q34	p = 0.556	NEP scale item 7	q22_7	P = 0.419
Likert-scale: knowledge biodiversity	q14_1	p = 0.976	Recycling	q23_1	P = 0.155
Likert-scale: knowledge flood protection	q14_3	p = 0.815	Donations	q23_2	P = 0.617
Likert-scale: knowledge carbon storage	q14_4	p = 0.485	Buy organic products	q23_3	P = 0.231
Likert-scale: knowledge recreation	q14_5	p = 0.660	Risk scale	q24	P = 0.179
Affected by flooding in the past	q13	p = 0.724	Discounting scale	q25	P = 0.394

Table B.2: Mixed logit model in preference and WTP space, no interaction with treatment.

	Preference Space		WTP space	
Number of observations		10,536		10,536
Number of parameters		18		18
Log likelihood		-2734.01		-2881.86
AIC		5504.02		5799.73
BIC		5614.97		5910.68
Choice	**Mean (St. error)**	**Standard deviation (St. error)**	**Mean (St. error)**	**Standard deviation (St. error)**
Maintaining current biodiversity level	0.657*** (0.0717)	-0.135 (0.312)	100.946*** (12.772)	-32.622*** (12.163)
Increasing biodiversity level	1.0177*** (0.0897)	-0.112 (0.376)	196.741*** (13.948)	15.635 (14.825)
Maintaining current flood defence level	0.4667*** (0.0707)	0.011 (0.231)	73.945*** (12.266)	-75.634*** (13.358)
Increasing flood defence level	1.0697*** (0.094)	0.6599*** (0.146)	211.636*** (15.363)	-80.626*** (14.996)
Marginal increase in carbon storage	0.051*** (0.004)	0.039*** (0.006)	8.938*** (0.657)	4.827*** (0.667)
Providing bridges and boardwalks	0.499*** (0.072)	-0.193 (0.250)	70.417*** (11.433)	16.312 (12.997)
Providing bridges, boardwalks, and bird hides	0.345*** (0.074)	0.362* (0.197)	56.633*** (10.449)	-73.717*** (12.285)
asc (alternative specific constant)	3.336*** (0.348)	2.917*** (0.392)	739.199*** (94.679)	810.0299*** (91.841)
Increase in income tax for 10 years	-0.033*** (0.012)	0.5299 (0.463)	-0.018*** (0.005)	0.045** (0.025)

The sign of the estimated standard deviations is irrelevant: interpret them as being positive
***, ** and * indicate 1,5 and 10% significance levels respectively

Table B.3: Significant correlations of the mixed logit model in preference space with correlation between all attributes specified. This model provides evidence that respondents' choices are correlated and not independent.

	No interaction with treatment		**Interaction of treatment with the carbon attribute**	
Number of observations		10,536		10,536
Number of parameters		54		65
Log likelihood		-2689.36		-2686
AIC		5486.72		5501.99
BIC		5819.57		5902.65
Choice - Correlation	**Estimate**	**Std. Error**	**Estimate**	**Std. Error**
biodiv0_biodiv1	1.068***	0.229	1.022***	0.248
biodiv0_flood0	0.385	0.243	0.193	0.232
biodiv1_flood0	0.199	0.231	0.403**	0.201
biodiv0_flood1	0.768***	0.2997	0.597*	0.319
biodiv1_flood1	0.980***	0.346	1.153***	0.337
flood0_flood1	0.854***	0.287	0.783**	0.319
biodiv0_carbon	0.0095	0.010	x	x
biodiv1_carbon	0.028**	0.012	x	x
flood0_carbon	0.013	0.014	x	x
flood1_carbon	-0.015	0.015	x	x
biodiv0_carbonShort	x	x	-0.011	0.016
biodiv1_carbonShort	x	x	0.034**	0.015
flood0_carbonShort	x	x	-0.001	0.018
flood1_carbonShort	x	x	-0.018	0.022
biodiv0_carbonLong	x	x	0.012	0.013
biodiv1_carbonLong	x	x	0.042***	0.016
flood0_carbonLong	x	x	-0.0001	0.023
flood1_carbonLong	x	x	-0.019	0.023
carbonShort_carbonLong	x	x	0.0244	0.034
biodiv0_recreation1	-0.378*	0.221	-0.358	0.239
biodiv1_recreation1	0.814***	0.198	0.682***	0.194
flood0_recreation1	-0.042	0.340	-0.438	0.327
flood1_recreation1	-0.090	0.331	0.243	0.359
carbon_receration1	-0.354	0.258	x	x
carbonShort_recreation1	x	x	-0.501*	0.272
carbonLong_recreation1	x	x	-0.179	0.374
biodiv0_recreation2	-0.082	0.242	-0.159	0.262
biodiv1_recreation2	0.710***	0.226	0.668***	0.214
flood0_recreation2	0.009	0.359	-0.219	0.373
flood1_recreation2	-0.399	0.326	-0.174	0.329
carbon_recreation2	-0.524	0.320	x	x
carbonShort_recreation2	x	x	-0.615*	0.318
carbonLong_recreation2	x	x	-0.041	0.492
recreation1_recreation2	-0.027	0.522	-0.5895*	0.343
biodiv0_asc	-0.224	0.493	-0.507	0.412
biodiv1_asc	-0.2595	0.426	0.213	0.312
flood0_asc	0.858*	0.515	0.066	0.436
flood1_asc	0.146	0.492	-0.038	0.413
carbon_asc	0.658	0.458	x	x
carbonShort_asc	x	x	0.281	0.434
carbonLong_asc	x	x	0.108	0.586
recreation1_asc	1.279**	0.622	1.645***	0.440
recreation2_asc	-1.205	0.892	-0.292	0.479
biodiv0_payment	-0.389***	0.121	-0.026	0.117
biodiv1_payment	0.388**	0.154	0.014	0.0998
flood0_payment	0.253	0.167	-0.442***	0.112
flood1_payment	0.813***	0.179	0.428***	0.126
carbon_payment	0.028	0.127	x	x

carbonShort_payment	x	x	-0.401***	0.135
carbonLong_payment	x	x	-0.963***	0.140
recreation1_payment	-0.264**	0.127	1.3195***	0.135
recreation2_payment	-0.005	0.356	0.647***	0.126
asc_payment	1.573***	0.182	0.941***	0.123

The sign of the estimated standard deviations is irrelevant: interpret them as being positive
***, ** and * indicate 1,5 and 10% significance levels respectively

Table B.4: Comparison of the final models run in Stata and R providing details on differences in the results.

Model	Description	Stata		R	
MixlPrefC	Mixed logit model in preference space, no interaction with treatment	LL Wald Mean Wald SD	-2732.85 x x	LL Wald Mean Wald SD	-2734.01 x x
MixlPrefT-C	Mixed logit model in preference space, interaction treatment - carbon	LL Wald Mean Wald SD	-2732.62 p=0.4208 p=0.0000	LL Wald Mean Wald SD	-2736.7 p=0.32199 p=0.0000
MixlPrefT-A	Mixed logit model in preference space, interaction treatment - all	LL Wald Mean Wald SD	-2722.05 p=0.2413 p=0.0404	x x x	
MixlPrefCcorr	Mixed logit model in preference space, no interaction with treatment, allowing for all correlation between attributes	LL Wald Mean Wald SD	-2689.23 x x	LL Wald Mean Wald SD	-2689.36 x x
MixlPrefT-Ccorr	Mixed logit model in preference space, interaction treatment - carbon, allowing for all correlation between attributes	LL Wald Mean Wald SD	-2679.79 p=0.8201 p=0.4176	LL Wald Mean Wald SD	-2686 p=0.3928 p=0.0116
MixlPrefT-Acorr	Mixed logit model in preference space, interaction treatment - all, allowing for all correlation between attributes	x		x	
MixlWTPC	Mixed logit model in WTP space, no interaction with treatment	x		LL Wald Mean Wald SD	-2881.86 x x
MixlWTPT-C	Mixed logit model in WTP space, interaction treatment - carbon	x		LL Wald Mean Wald SD	-2876.58 p=0.1074 p=0.0014

Figure B.1: Choice Experiment Questionnaire with Treatment (i.e., increased information).

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Management Options for Scotland's Coastal Marshlands in the face of Climate Change

Management Options for Scotland's Coastal Marshlands in the face of Climate Change

You are being invited to participate in a research study about future coastal management. This study is conducted by Simone Riegel from the School of Geography and Sustainable Development at the University of St Andrews as part of her PhD research.

(Waulkmill Bay Saltmarsh, Orkney; ©Simone Riegel)

What is this survey about?
I am researching different management options for saltmarshes. Saltmarshes are scattered along the Scottish coastline and can provide benefits to society. Unfortunately, saltmarshes are threatened by human activity and sea level rise. However, they can be managed, which means that measures can be taken to preserve saltmarshes and their benefits. With this questionnaire I would like to ask for your opinion on different management options for saltmarshes. This will help me to gain an understanding of the wider Scottish population's preferred management option. I anticipate that filling in this questionnaire will take about 20 minutes.

Funding
This study is undertaken with the support of Marine Scotland. The results will be shared with the Scottish Government through Marine Scotland.

Your Data
The data collected in this survey will only be used for the purposes of this study and will be strictly anonymous.
If you have any questions, please contact Toluna; your queries will be forwarded to the researcher. Your participation is entirely voluntary, and you can withdraw at any time while filling in the questionnaire. Completed questionnaires cannot be retracted since the data is anonymous and cannot be matched to you.

Thank you in advance for taking the time to complete this questionnaire.

☐ **I consent to participating in this survey (tick box to proceed)**

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What is you biological sex?

- ○ Male
- ○ Female
- ○ Prefer not to say

What is your age?

- ○ under 18
- ○ 18-24
- ○ 25-34
- ○ 35-44
- ○ 45-54
- ○ 55-64
- ○ 65 and over
- ○ Prefer not to say

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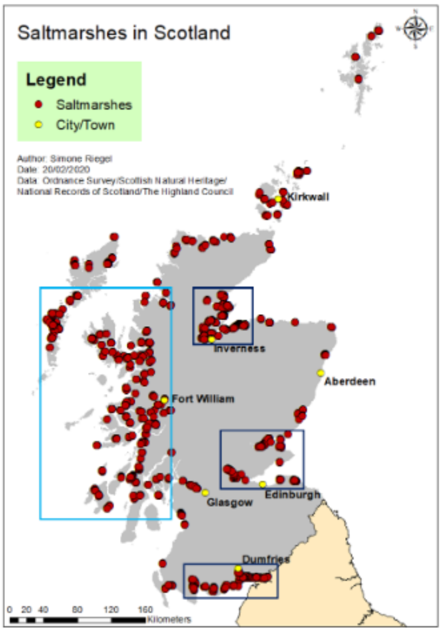
What is a saltmarsh and why is it important?

Saltmarshes are located in sheltered places (e.g. river estuaries) and are regularly flooded by the sea at high tide. They have higher areas (upper marsh, landwards) that are only flooded during spring tides or storms, and lower areas (lower marsh, seawards), which are flooded more regularly.

(left: saltmarsh at low tide; right: the same saltmarsh flooded at high tide; ©Simone Riegel)

Large extents of undisturbed mud by the sea can become saltmarshes when grass or seaweed start growing on them. Since the sea regularly floods saltmarshes, their soil is saturated with saltwater. Pools of water are frequent on the marsh and natural ditches (tidal creeks) formed by the water cross it all the way to the sea.

Unfortunately, saltmarshes are very threatened by sea-level rise and human activity; about 50% of saltmarshes have been lost globally. In the last 75 years (since 1945), Scotland has lost about 13% of its saltmarshes and will lose a further 12% in the next 40 years (until 2060) if nothing is done to prevent this from happening.



(Concentrated saltmarsh areas are primarily located on the north-eastern, eastern and south-western coast [e.g. in estuaries, dark blue boxes]; on the western and north-western coast saltmarshes are usually smaller and more scattered [e.g. at the head of sea lochs and behind barrier islands, light blue box])

In the last 75 years, Scotland has lost:

- ○ 13% of its saltmarshes
- ○ 26% of its saltmarshes
- ○ 50% of its saltmarshes

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Have you heard about saltmarshes before?

- ○ Yes
- ○ No

Have you been to a saltmarsh in Scotland before?

- ○ Yes
- ○ Yes, I have been to several different saltmarshes in Scotland
- ○ No
- ○ No, but I have been to a saltmarsh outside of Scotland

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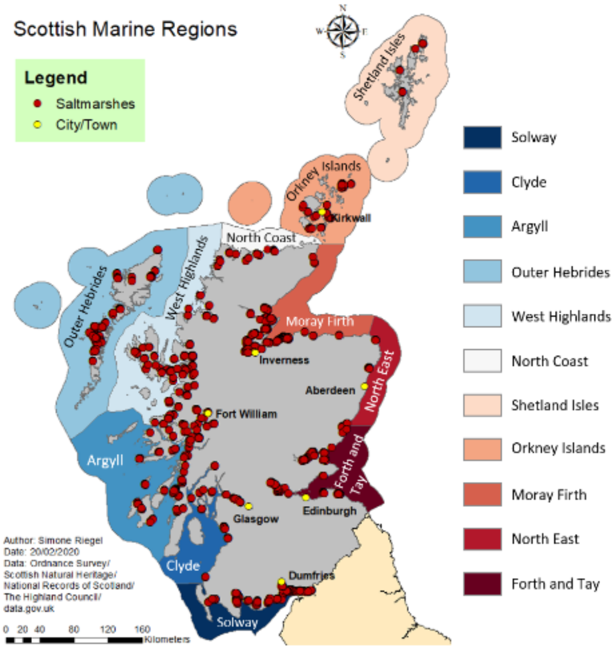
Where is/are the saltmarsh(es) you have visited?



- ☐ Solway
- ☐ Clyde
- ☐ Argyll
- ☐ Outer Hebrides
- ☐ West Highlands
- ☐ North Coast
- ☐ Shetland Isles
- ☐ Orkney Islands
- ☐ Moray Firth
- ☐ North East
- ☐ Forth and Tay
- ☐ Unsure

How often do you visit saltmarshes?

- ○ Almost every day
- ○ 1 to 2 times per week
- ○ 1 to 2 times per month
- ○ 1 to 2 times per year
- ○ Less often
- ○ Just once

Why do you visit saltmarshes? Please use the textbox in the last option if you have other interests when you visit saltmarshes and the provided options do not fit.

- ☐ To go for a walk
- ☐ Dog walking
- ☐ Wildfowling
- ☐ To enjoy nature
- ☐ For educational purposes (e.g. research)
- ☐ Birdwatching
- ☐ Other, please specify... ______

How would you describe the state of the last Scottish saltmarsh you visited?

	Strongly disagree	Disagree	Unsure	Agree	Strongly agree
The saltmarsh is easily accessible (e.g. boardwalks, parking, fences etc.)	○	○	○	○	○
The saltmarsh appears to be undisturbed (e.g. no artificial trenches, etc.)	○	○	○	○	○
The saltmarsh is grazed (e.g. by cattle or sheep)	○	○	○	○	○
The saltmarsh appears to expand and has a pioneer zone (the marsh is expanding further seawards; e.g. see picture below) 	○	○	○	○	○
The saltmarsh shows signs of erosion (e.g. see picture below) 	○	○	○	○	○
There is wildlife on the marsh (e.g. birds)	○	○	○	○	○

In which region is the saltmarsh you just described?

- ○ Solway
- ○ Clyde
- ○ Argyll
- ○ Outer Hebrides
- ○ West Highlands
- ○ North Coast
- ○ Shetland Isles
- ○ Orkney Islands
- ○ Moray Firth
- ○ North East
- ○ Forth and Tay
- ○ Unsure

Do you agree with the following statements?

	Strongly disagree	Disagree	Unsure	Agree	Strongly agree
Saltmarshes support a variety of plants and animals (biodiversity)	○	○	○	○	○
Saltmarshes have a negative impact on the fishing industry close by because fish can hide in the creeks during high tide	○	○	○	○	○
Saltmarshes provide protection from flooding in coastal areas	○	○	○	○	○
Saltmarshes take carbon dioxide (a greenhouse gas) out of the atmosphere and can store the carbon long-term (e.g. centuries)	○	○	○	○	○
Saltmarshes support a variety of recreational activities	○	○	○	○	○

Have you been affected by a coastal flooding event in the past (e.g. flooding by sea water during a storm)?

- ○ Yes
- ○ Yes, several times
- ○ No

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Below is some more information about different benefits saltmarshes can provide. Please read through it carefully; it will be helpful later on, when we ask you to make choices about your preferences for the future management of saltmarshes.

Biodiversity

- Describes the variety of life found in a place (e.g. animals and plants, etc.)
- There are not many different species of plants and animals that can be found on saltmarshes, but a lot of the species that are there don't exist in many other places
- The number of different bird species that breed on the saltmarsh can tell us whether the saltmarsh has a healthy level of biodiversity

Sea Defence

- Saltmarshes are natural flood defences and can reduce the height of damaging waves during a storm surge by almost 20%. This reduces the risk of erosion and flooding at the coast
- The saltmarsh plants reduce the strength of waves before they hit the land
- Having a saltmarsh on the coast can reduce the need for high and costly hard-structure seawalls

Carbon Storage

- Plants, including saltmarsh plants, remove carbon dioxide (CO_2) from the atmosphere during photosynthesis to use it for growth. CO_2 is one of the greenhouse gases that cause global climate change; the more CO_2 in the atmosphere, the hotter the climate on Earth
- Some of the carbon ends up in the saltmarsh roots and then stays trapped in the soil, which makes saltmarshes very effective carbon stores as long as the soil is not disturbed
- The saltmarsh vegetation also traps sediment that was transported to the marsh from other locations (e.g. by a river or by flooding through the tide). These sediments also contain carbon, which then gets locked away by the saltmarsh and is not released into the atmosphere
- Forests also store away carbon in this way, but saltmarshes are about 10 times more efficient
- If the saltmarsh (or portions of it) is destroyed, the stored carbon is released and contributes to global climate change

Recreation

- Saltmarshes provide recreational opportunities
- Scenic views from the water (e.g. sailing in estuaries) and the tidal creeks of bigger saltmarshes can be explored with a paddle board or smaller boats; artistic appeal of the dynamic marsh landscape and wildlife (e.g. birds, otters, etc.)
- Saltmarshes attract many bird species, so they are a prime spot for birdwatching and wildfowling (shooting geese and ducks)
- Some saltmarshes are more accessible than others (e.g. wooden walkways, bridges over wide creeks, etc.) and are used for everyday recreational activities like dog-walking


(A bridge and boardwalk on the saltmarsh close to Port Appin at the west coast of Scotland; ©Simone Riegel)

Please choose the correct answer:

○ Saltmarshes are bad for carbon storage

○ The different plant species found on saltmarshes are very common and can also be found in many other places

○ Saltmarshes are natural flood defences and protect the coastline from erosion and flooding

○ Saltmarshes are not very appealing and cannot be used for any recreational activities

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Management Options for Scotland's Coastal Marshlands in the face of Climate Change

On the next page we will present you with a range of different management options for saltmarshes. Each option will influence the various benefits saltmarshes can provide (Biodiversity/Flood Defence/Carbon Storage/Recreational Infrastructure) in a different way.

An example of such a management option could be to plant saltmarsh plants. This can help regenerate an existing marsh that has previously been eroded or can help restore a saltmarsh that has previously been completely destroyed.
Another option to increase saltmarsh quality would be to fill in artificial ditches. These ditches were originally created to drain the saltmarsh to achieve a small improvement in the land's suitability for livestock grazing. At the same time, the ditches damage the marsh, which, for example, negatively influences its ability to support biodiversity and to store carbon.

The different management options come at a cost and will be funded by the Scottish Government. Since this money comes from taxes, we are interested in which options you, as the general public, prefer and prioritise and how much you would be willing to pay to support management options that increase the benefits saltmarshes provide.

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In this section, we ask you to **choose your preferred management option** among several different choices. The management options would be **implemented by the local authorities and the central government (i.e. Scottish Government).** These management options would have different impacts on:

The number of bird species on Scottish saltmarsh, which could increase by 15% (+15%) or remain the same or decrease by 15% (-15%) if nothing is done.

The length of the coastline protected by saltmarshes, which could increase by 14km (+14km) or remain the same if managed. If nothing is done, due to sea level rise, the length of coastline protected by saltmarshes would decrease by 14km (-14km).

The amount of carbon that would additionally be absorbed from the atmosphere and stored away, which could increase by the equivalent of the annual emission of 4,000 cars, 10,000 cars, 16,000 cars or 20,000 cars or be maintained to the current level. If nothing is done, saltmarshes would release carbon to the atmosphere, this release would be equivalent to the annual emissions of 10,000 cars (-10,000). **The increased storage is a benefit that would not be immediate but come into effect over time.**

The recreational infrastructure available: boardwalks () and bridges over creeks () could be constructed, as well as shelters for birdwatchers (). Currently none of these infrastructures are available on most of the saltmarshes.

50 The increase in income tax per year (between £25 - £300) that would be necessary to fund such management. The income tax would be increased once by this factor for the next 10 years. The additional money would be protected and only be used for saltmarsh management. Reminder: currently, for an income of £30,000 you would pay £3,500 per year income tax according to current tax bands. For this example, the tax would increase from £3,500 to £3,525-£3,800 per year depending on which management option is chosen.

While stopping the degradation of saltmarshes and maintaining current benefits they provide could be achieved in a matter of a few years, increasing the benefits as described could take up to 30-50 years.

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Management Options for Scotland's Coastal Marshlands in the face of Climate Change

On the following pages, we present the outcomes of three different management options. The first two are management options that can be realised through a financial contribution. The last one represents the current state of saltmarshes and requires no financial contribution.

For each question, we ask that you **choose the management option that you personally prefer the most even if it is not your ideal management option**. If you choose the current situation ('Business as usual' option) the increase in income tax will be zero. If you think that the price matching the proposed management options is too high, please select the current situation. **Please consider your personal budget and what you would have to give up in order to pay for the increase in income tax when making the choices**.

Below is an example of the choices you will have to make on the next pages. **Overall, you will be asked 6 times to choose between three different options.** The option columns of the table describe how exactly Biodiversity, Flood Defence, Carbon Storage, and Recreational Infrastructure will be influenced. At the bottom of each column you can see the additional amount of income tax you would have to pay. The 'Business as usual' option will always stay the same and has no additional cost. If you prefer option 2, please tick the corresponding box as shown on the image below.

	Option 1	**Option 2**	**Business as usual**
Biodiversity (breeding bird species on saltmarsh)	+15% bird species breeding on saltmarsh	-15% bird species breeding on saltmarsh	-15% bird species breeding on saltmarsh
Flood Defence (protected coastline)	-14 km protected	+14 km protected	-14 km protected
Carbon Storage (released or additionally stored; measured in equivalent yearly car emissions)	carbon **released**; equivalent to yearly emission of **10,000** cars	additional carbon **stored**; equivalent to yearly emission of **20,000** cars	carbon **released**; equivalent to yearly emission of **10,000** cars
Recreational Infrastructure (provision of new infrastructure)	No Infrastructure	pathways and bridges	No Infrastructure
Price/Payment per year (£)	50	150	0

Option 1 ○ Option 2 ◉ Business as usual ○

In the next section you will be shown several varying and numbered tables and asked to make a choice for each table; how many choices are you asked to make in the next section?

- ○ 4 choices (the tables are numbered 1-4)
- ○ 6 choices (the tables are numbered 1-6)
- ○ 7 choices (the tables are numbered 1-7)

Do you pay income tax?

- ○ Yes
- ○ No

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Table 1: Please choose your preferred option amongst the 3 options below.

	Option 1	Option 2	Business as usual
Biodiversity (breeding bird species on saltmarsh)	+15% bird species breeding on saltmarsh	no change in bird species breeding on saltmarsh	-15% bird species breeding on saltmarsh
Flood Defence (protected coastline)	no change in km protected	+14 km protected	-14 km protected
Carbon Storage (released or additionally stored; measured in equivalent yearly car emissions)	additional carbon **stored**; equivalent to yearly emission of **20,000** cars	carbon **released**; equivalent to yearly emission of **10,000** cars	carbon **released**; equivalent to yearly emission of **10,000** cars
Recreational Infrastructure (provision of new infrastructure)	pathways, bridges and bird hides	pathways and bridges	No Infrastructure
Price/Payment per year (£)	150	50	0

Option 1 ○ Option 2 ○ Business as usual ○

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Table 2: Please choose your preferred option amongst the 3 options below.

	Option 1	Option 2	Business as usual
Biodiversity (breeding bird species on saltmarsh)	-15% bird species breeding on saltmarsh	no change in bird species breeding on saltmarsh	-15% bird species breeding on saltmarsh
Flood Defence (protected coastline)	-14 km protected	no change in km protected	-14 km protected
Carbon Storage (released or additionally stored; measured in equivalent yearly car emissions)	additional carbon **stored**; equivalent to yearly emission of **4,000** cars	additional carbon **stored**; equivalent to yearly emission of **4,000** cars	carbon **released**; equivalent to yearly emission of **10,000** cars
Recreational Infrastructure (provision of new infrastructure)	pathways and bridges	No Infrastructure	No Infrastructure
Price/Payment per year (£)	300	300	0

Option 1 ○ Option 2 ○ Business as usual ○

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Table 3: Please choose your preferred option amongst the 3 options below.

	Option 1	Option 2	Business as usual
Biodiversity (breeding bird species on saltmarsh)	no change in bird species breeding on saltmarsh	-15% bird species breeding on saltmarsh	-15% bird species breeding on saltmarsh
Flood Defence (protected coastline)	+14 km protected	-14 km protected	-14 km protected
Carbon Storage (released or additionally stored; measured in equivalent yearly car emissions)	additional carbon **stored**; equivalent to yearly emission of **16,000** cars	current level of carbon storage	carbon **released**; equivalent to yearly emission of **10,000** cars
Recreational Infrastructure (provision of new infrastructure)	No Infrastructure	pathways, bridges and bird hides	No Infrastructure
Price/Payment per year (£)	200	25	0

Option 1 ○ Option 2 ○ Business as usual ○

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Table 4: Please choose your preferred option amongst the 3 options below.

	Option 1	Option 2	Business as usual
Biodiversity (breeding bird species on saltmarsh)	no change in bird species breeding on saltmarsh	+15% bird species breeding on saltmarsh	-15% bird species breeding on saltmarsh
Flood Defence (protected coastline)	-14 km protected	+14 km protected	-14 km protected
Carbon Storage (released or additionally stored; measured in equivalent yearly car emissions)	additional carbon **stored**; equivalent to yearly emission of **20,000** cars	carbon **released**; equivalent to yearly emission of **10,000** cars	carbon **released**; equivalent to yearly emission of **10,000** cars
Recreational Infrastructure (provision of new infrastructure)	No Infrastructure	pathways, bridges and bird hides	No Infrastructure
Price/Payment per year (£)	50	100	0

Option 1 ○ Option 2 ○ Business as usual ○

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Table 5: Please choose your preferred option amongst the 3 options below.

	Option 1	Option 2	Business as usual
Biodiversity (breeding bird species on saltmarsh)	no change in bird species breeding on saltmarsh	-15% bird species breeding on saltmarsh	-15% bird species breeding on saltmarsh
Flood Defence (protected coastline)	no change in km protected	+14 km protected	-14 km protected
Carbon Storage (released or additionally stored; measured in equivalent yearly car emissions)	additional carbon **stored**; equivalent to yearly emission of **20,000** cars	carbon **released**; equivalent to yearly emission of **10,000** cars	carbon **released**; equivalent to yearly emission of **10,000** cars
Recreational Infrastructure (provision of new infrastructure)	pathways and bridges	No Infrastructure	No Infrastructure
Price/Payment per year (£)	100	100	0

Option 1 ○ Option 2 ○ Business as usual ○

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Table 6: Please choose your preferred option amongst the 3 options below.

	Option 1	Option 2	Business as usual
Biodiversity (breeding bird species on saltmarsh)	+15% bird species breeding on saltmarsh	-15% bird species breeding on saltmarsh	-15% bird species breeding on saltmarsh
Flood Defence (protected coastline)	-14 km protected	no change in km protected	-14 km protected
Carbon Storage (released or additionally stored; measured in equivalent yearly car emissions)	current level of carbon storage	additional carbon **stored**; equivalent to yearly emission of **20,000** cars	carbon **released**; equivalent to yearly emission of **10,000** cars
Recreational Infrastructure (provision of new infrastructure)	pathways, bridges and bird hides	pathways and bridges	No Infrastructure
Price/Payment per year (£)	100	50	0

Option 1 ○ Option 2 ○ Business as usual ○

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You see this question if you chose the 'Business as usual' option (= £0) at least once. Why did you choose the 'Business as usual' option? Please choose the statements that apply

- ☐ Government/local authority should pay
- ☐ Taxes and fees are already too high, so there should not be an additional financial burden
- ☐ I am a non-tax payer/I am not working
- ☐ I think it is better to ask experts about how saltmarshes should be managed
- ☐ I am not concerned about the condition of saltmarshes
- ☐ I cannot afford to pay any more in taxes
- ☐ The positive change of the saltmarsh condition is not significant enough to pay for it
- ☐ It is not a priority/other things are more important
- ☐ Other, please specify...

How comfortable were you with the choices you made in the choice card questions?

Very uncomfortable	Quite uncomfortable	Neither comfortable nor uncomfortable	Quite comfortable	Very comfortable
○	○	○	○	○

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Why were you very/quite uncomfortable with your choice?

- ○ None of the scenarios represented my preference
- ○ I do not care about saltmarshes and their benefits
- ○ I was unsure which scenario to choose out of the three possibilities
- ○ I was unsure if I would end up having to pay the selected price
- ○ I could not represent my preference with the combination of choices I made
- ○ Other, please specify...

Do you agree or disagree with the following statements?

	Strongly Disagree	Disagree	Unsure	Agree	Strongly agree
We are approaching the limit of the number of people the earth can support	○	○	○	○	○
When humans interfere with nature it often produces disastrous consequences	○	○	○	○	○
Humans are severely abusing the environment	○	○	○	○	○
Plants and animals have as much right as humans to exist	○	○	○	○	○
If things continue on their present course, we will soon experience a major ecological catastrophe	○	○	○	○	○
We need to support a political focus on the environment and environmental protection	○	○	○	○	○
We should consider moving infrastructure (such as roads), so the coastline can naturally adapt to sea-level rise	○	○	○	○	○

How often...

	Never	Hardly ever	Sometimes	Often	Whenever possible
Do you recycle?	○	○	○	○	○
Do you donate to conservation associations?	○	○	○	○	○
Do you buy organic products?	○	○	○	○	○

How willing or unwilling are you, in general, to take risks. Please use a scale from 0 to 10, where 0 means you are "completely unwilling to take risks" and a 10 means you are "very willing to take risks".

0	1	2	3	4	5	6	7	8	9	10
○	○	○	○	○	○	○	○	○	○	○

How willing are you to give up something that is beneficial for you today in order to benefit more from that in the future? Please again indicate your answer on a scale from 0 to 10, where 0 means you are "completely unwilling to do so" and a 10 means you are "very willing to do so"

0	1	2	3	4	5	6	7	8	9	10
○	○	○	○	○	○	○	○	○	○	○

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University of St Andrews

Management Options for Scotland's Coastal Marshlands in the face of Climate Change

What is the highest degree or level of education you have completed (if you have a non-UK education, please choose the option most similar)?

- ○ High School Qualifications
- ○ College Qualifications (e.g. HNC/HND)
- ○ Bachelor's Degree
- ○ Master's Degree
- ○ PhD or higher
- ○ Technical Qualifications (e.g. apprenticeship, etc.)
- ○ Prefer not to say

What is your marital status?

- ○ Single
- ○ Married
- ○ Civil Partnership
- ○ Divorced
- ○ Widowed
- ○ Other, please specify...
- ○ Prefer not to say

How many children do you have?

- None
- 1
- 2
- 3
- 4 or more
- Prefer not to say

What is your annual household income (before paying taxes)?

- £0-£12,500/year
- £12,501-£20,000/year
- £20,001-£30,000/year
- £30,001-£40,000/year
- £40,001-£50,000/year
- Over 50,000/year
- Prefer not to say

What is your current employment status?

- Employed Full-Time
- Employed Part-Time
- Self-Employed
- Homemaker
- Seeking opportunities
- Student
- Retired
- Other, please specify...
- Prefer not to say

What are the first four letters and digits of your postcode (e.g. KY16)?

Did you participate in the last election?

- Yes
- No
- Prefer not to say

Do you have any comments regarding this questionnaire or saltmarshes?

Thank you very much for participating in this survey. Please make sure to click on the 'Submit' button at the end of this page to submit the questionnaire.

If you want to get in touch and find out more about the research, please contact Toluna; all queries will be forwarded to the researcher.

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Submit

APPENDIX C

Table C.1: Key interview themes and research questions that were addressed.

Theme	Research Question
Uncertainty	How is uncertainty in scientific evidence that supports a policy draft dealt with?
Public Acceptability	How does the acceptability of policies play a role in setting a policy?
Policy-making Process	What are the steps between scientific evidence and a policy draft?
Economic Considerations	What are the economic considerations that are taken into account for policy?
Information	How much and what kind of information is needed about a habitat?
Scale	Which scale should the policy take, national or regional?

Figure C.1: Interview Schedule.

Interview Schedule

Introduction

Hi, my name is Simone, and I am a PhD student at the University of St Andrews. My PhD project is about the potential of Scottish saltmarshes to help with the efforts of containing and reducing climate change and the consequences that come with it. The interview today is going to help me with my PhD thesis where I want to connect this potential to policy.

Before we start, I want to, first of all, thank you very much for taking the time to meet with me today!

You indicated in the consent email that you are happy to be recorded, I just started the recording/You indicated in the consent for that you are not happy to be recorded. I will take notes during the interviews, so I want to apologise in advance that I might be turning away from the screen slightly sometimes to take notes. Do you have any questions about the interview or the consent form I asked you to sign?

Before we start with the interview questions I prepared, it would be great if you could briefly introduce yourself and tell me a bit more about your position and your role within Government/NatureScot.

…

Thank you very much for giving me this overview of your role. On a scale from 1-10, with 1 representing science and 10 representing policy, where would you locate yourself?

Questions RE ‘Uncertainty’

It is quite common that there is uncertainty attached to scientific data. One example is the different greenhouse gas emission scenarios that lead to different estimates of sea-level rise and depending on which scenario comes to pass, Scotland will be affected to different extents. For example, for the different emission scenarios, the projected sea-level rise for the Orkney Islands lies between 0.4 and 0.7m by 2100, which can make a significant difference. My first topic is about this uncertainty.

1. How do you deal with uncertainty in scientific data that supports a policy draft?
2. How would you rank the importance of reducing this uncertainty for setting a policy? Very Important/Important/Somewhat Important/Not Important

Follow-up Questions

3. When does uncertainty become an inhibiting factor for setting policy? Is there a tipping point?[And why? Follow up question if not mentioned in reply] Can you give some examples from your previous work?
4. From your experience, in which phase of the policy process is uncertainty the most detrimental?
5. Are there currently specific uncertainties that you are monitoring about Blue Carbon in your department?

Questions RE ‘Public Acceptability’

Thank you very much for sharing your insights about uncertainties with me, the next topic I want to talk about is ‘Public Acceptability’. In the past years, environmental policy has received a lot of attention from the media and the public. The Paris Agreement, for example has been highly publicised and it seems like there is an increased interest in environmental issues like climate change.

6. How would you say the acceptability of policies plays a role in you setting a policy?
7. How would you rank the importance of public acceptability in setting a policy? Very Important/Important/Somewhat Important/Not Important

Follow-up Questions

8. Does the Scottish Government gauge the public’s acceptance of a policy? If yes, how?
9. Are there different strategies regarding public acceptability for the development and then the implementation phase of policy?

10. What role does communication play with public acceptability regarding environmental policy?

<u>Questions RE 'Procedural Validity'</u>

Another area I am very interested in is the details of the policy development process.

11. How would you describe the steps between receiving scientific evidence and a first policy draft?

Follow-up Question

12. Who is involved in those steps?

13. What are the review procedures for a new policy before it is introduced as a bill to Parliament?

Follow-up Question

14. Are they internal or external?

15. In your experience, what are the requirements to make this process work well?
16. How would you rank the importance of review procedures in setting a policy? Very Important/Important/Somewhat Important/Not Important
17. How can Blue Carbon be integrated into existing frameworks? And in which development step is this decided?

<u>Questions RE 'Economic Feasibility'</u>

Another important aspect that comes to my mind when I think about policy is economic considerations.

18. What are the economic considerations that you take into account when you set a policy?

Follow-up Question

19. Are these the same for all policies or are there any differences?
20. When you set a policy with the aim to increase carbon storage in a habitat, such as forests or peatlands for example, do you only consider the value of the carbon benefit, or do you also consider the value of other potential benefits that can be achieved as side effects of the policy?

21. How would you rank the importance of economic considerations in setting a policy? Very Important/Important/Somewhat Important/Not Important

Question(s) RE 'Functional Calculations'

An important point that we haven't covered yet, is the kind of information that you need to draft policy.

22. How much information do you need about the habitat for your part in drafting policy (e.g. location/size/condition)? Do you want more high-level information or detailed data on Blue Carbon habitats to draft policy?
23. How would you rank the importance of background information about the habitat? Very Important/Important/Somewhat Important/Not Important
24. What key additional scientific evidence is required to set Blue Carbon Policy?

Question RE 'Scalability'

The last topic I want to talk about is the question of scale. There are existing frameworks for environmental policy on the national level but then there is also the advantage of local knowledge about the habitats that are close by. I'm very interested in what your experience is in this area.

25. Should policy on Blue Carbon be set on the Scottish national level and then implemented locally or would it be better to decide locally on the most fitting policy?
26. How would you rank the importance of thinking about national versus local scale in setting policy? Very Important/Important/Somewhat Important/Not Important

End of Interview Questions

We have come to the end of the questions that I prepared for this interview. Thank you very much for your time and for sharing your experience with me! You shared great insights with me, which is very helpful for my work. Before we end this conversation, I want to ask you if…

27. There are any important aspects about drafting and implementing Blue Carbon policy that we didn't touch upon, but that you would like to raise?
28. Is there anyone you could think of that I should also have this conversation with?

If there are no further points you want to add, we can conclude this interview here. Thank you very much again and please don't hesitate to send me an email if you have any questions or want to add anything to our conversation.