Article:

Muenzel, Dominic and Martino, Simone orcid.org/0000-0002-4394-6475 (2018) Assessing the feasibility of carbon payments and Payments for Ecosystem Services to reduce livestock grazing pressure on saltmarshes. Journal of Environmental Management. 225. pp. 46-61. ISSN 0301-4797

https://doi.org/10.1016/j.jenvman.2018.07.060

Assessing the feasibility of carbon payments and Payments for Ecosystem Services to reduce livestock grazing pressure on saltmarshes

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Abstract

Saltmarshes provide important services including flood control, climate regulation, and provisioning services when grazed by livestock for agriculture and conservation purposes. Grazing diminishes aboveground carbon, creating a trade-off between these two services. Furthermore, saltmarshes are threatened by overgrazing. To provide saltmarsh protection and ensure the continuing delivery of ecosystem services, there is a need to incentivise land managers to stock environmentally sensible densities. We therefore investigated the possibility of agri-environmental schemes and Payments for Ecosystem Services (PES) to compensate for lost livestock revenue under reduced grazing regimes and provide carbon sequestration and other benefits. This is the first study to consider the benefits arising from a potential carbon market to saltmarshes, although similar schemes exist for peatland and woodland. We calculated the net economic benefit (costs of livestock production are removed from revenue) to farmers obtained from a hectare of grazed saltmarsh under low (0.3 Livestock Units per hectare per year), moderate (0.6), high (1.0) and very high (2.0) stocking densities accounting for livestock revenue, carbon benefits, and agri-environmental subsidies. We repeated the procedure considering additional benefits transferred from the literature in terms of provisioning, regulating and cultural ecosystem services provided by protected saltmarshes. The net benefits were assessed for a range of market carbon prices and social costs of carbon, e.g. the opportunity cost of carbon for society. Applying the model to Scottish saltmarshes we find that the current range of market prices could prompt transitions from high to moderate regimes in areas where livestock value is low, however break-even prices for transitions showed high spatial variability due to spatial variability in livestock values. In some areas of the West Highlands, the break-even carbon price is negative, indicating that the current agri-environmental schemes are able to more than compensate for the lost revenue accruing to farmers by a reduced grazing density. However, in other areas, such as the Outer Hebrides, the break-even carbon price is positive. Private PES schemes or increased public subsidies should then be provided to generate net benefits. It is reasonable to infer that a pure carbon market may have limited scope in incentivising consumers to buy carbon services, especially in areas with limited local number of buyers and corporates of small size. Under this circumstance, a premium carbon market offering bundled ecosystem services may help reduce grazing pressure across a larger number of Scottish saltmarshes, thereby providing globally important climate regulation services and at the same time protecting sensitive habitats.

Keywords

Saltmarsh, grazing management, carbon sequestration, Payments for Ecosystem Services, private and social costs of carbon

Highlights

- Reduced grazing may generate additional ecosystem services in saltmarshes
- Current agri-environmental schemes can compensate forgone low income
- Carbon payments could reduce grazing pressure where opportunity costs are higher
- High spatial variability in carbon price is required to compensate reduced stocking
- Bundling carbon with other ecosystem services might facilitate reduced grazing

1. Introduction

Saltmarshes are coastal habitats dominated by terrestrial plants that experience regular tidal inundation. They provide a wide range of ecosystem services including flood defence and wave attenuation (Pethick, 2002); water filtration and pollutant retention (Barbier et al., 2011); nutrient cycling (Burden et al., 2013); reservoirs for rare and specialist species (Jones et al., 2011); nursery for fish (Laffaille et al., 2000); habitat for grazing livestock (Olsen et al., 2011); and blue carbon sequestration (Beaumont et al., 2014). Despite their importance, considerable human activity has involved land reclamation for agriculture and urban development (Burden et al., 2013). As a result, saltmarsh cover in the UK has declined by 15% since 1945 (Beaumont et al., 2014) and quality has deteriorated, thereby compromising the capacity of saltmarshes to adapt to future sea level rise and climate change (Jones et al., 2011).

In recent years the UK has recognised the importance of protecting saltmarshes and taken steps towards their restoration, including landward retreat of coastal barriers and flooding of reclaimed land (Garbutt et al., 2006). There is general agreement that the value of ecosystem services thus restored far outweighs the cost of restoration, but unless these services are marketed there is a lack of finance for such restoration projects (Environment Bank, 2015). At the same time, there has been growing interest in creating novel markets to allow private investment in ecosystem service provision (Reed et al., 2013). One such option of finance is a scheme of Payments for Ecosystem Services (PES) (Kinzig et al., 2011; Natural Capital Committee, 2015).

PES schemes currently existing in the UK fall under the category of carbon sequestration under the Woodland Carbon Code for newly planted woodland (Forestry Commission, 2017) and carbon sequestration with bundled co-benefits under the Peatland Code for restored peatland (IUCN, 2015; Reed et al., 2013). These Codes provide the verification and accreditation rules replacing voluntary standards such as the Verified Carbon Standard that are often characterised by high transaction costs (Bonn et al., 2014). Different forms of markets for peatlands are reported by Bonn et al. (2014) who classify different schemes according to the services paid, whether they are publically or privately funded, and whether they are international or regional in scope. For example, the UK Peatland Code is a market based on corporate social responsibility (Bonn et al., 2014), covering restoration costs of peatland due to business potential investment. Other transactions are based on public money arising from agri-environmental schemes on carbon emission reduction (Reed et al., 2014) justified on the basis of paying for the fullest possible range of ecosystem service benefits, although they are poorly quantified (Reed et al., 2014). The above mentioned PES applied to peatlands fall in the category of voluntary transactions (private for carbon markets, and government mediated for agri-environmental schemes) between providers and users of resource-use proxies rather than ecosystem services (which are difficult to measure) and have the properties to show conditionality on specific agreed rules amongst parties (Wunder, 2015).

There is no equivalent Code for saltmarshes, although a comparable small-scale scheme is underway in the Deben estuary where funds for services are designed to meet costs of restoring degraded saltmarsh (Environment Bank, 2015). Notably, the current UK action plan for saltmarsh protection does not take into account the value of carbon sequestration services but only the compensation of lost habitat and flood reduction benefits (Natural Capital Committee, 2015). The creation of new saltmarsh for flooding regulation is considered more valuable than managing sustainable agricultural practices of existing saltmarshes. However, considering the valuable service of climate regulation that saltmarshes provide in the face of current climate change threats, there is a potential to establish a PES market to help finance saltmarsh protection and ensure continuing services.

Compared with other vegetated habitats, saltmarshes have a great carbon capture and storage capacity as a result of high primary productivity and tidal trapping of organic matter (Chmura et al., 2003). Saltmarshes in Scotland sequester an estimated 2.35-8.04 t CO_2 ha^{-1} yr^{-1} and have a total carbon stock of 569.7 t ha^{-1} (Beaumont et al., 2014). Unlike freshwater wetlands which are often significant sources of methane, methane emissions from saltmarsh are negligible (Bridgham et al., 2006). Regular inundation promotes anoxic soils that delay decomposition, such that the carbon captured in saltmarsh soils is a long term sink (Yu and Chmura, 2010) well suited for targeting of a carbon emissions reduction market scheme.

In addition to the carbon sequestration service they provide, saltmarshes in the UK are widely grazed for both agricultural purposes and as a conservation tool to enhance floral and faunal biodiversity (Bouchard et al., 2003). Grazing significantly influences vegetation structure and community composition (Yates et al., 2000), thus influencing the services provided. For example, biodiversity is generally maximized under a light grazing regime replicating a natural system of native duck and geese grazing which hosts a wide range of plant, invertebrate and bird species (Adnitt et al., 2007). Light grazing promotes reverse succession and greater species diversity; whereas, ungrazed and intensively grazed regimes tend to produce monocultures (Fleischner, 1994).

The relationship between carbon sequestration and grazing on the other hand is quite complex, resulting from an interaction of stocking density, grazer type, saltmarsh zone, seasonality, and other abiotic parameters. Results from the few existing studies have been contradictory, perhaps exposing geographic differences or the overriding importance of abiotic factors. Grazed saltmarsh in Canada (Yu and Chmura, 2010) and the Netherlands (Elschot et al., 2015) had higher soil carbon contents than ungrazed sites while grazed saltmarsh in Denmark had lower soil organic matter content than ungrazed sites (Morris and Jensen, 1998). A large-scale study in England and Wales found grazing to reduce aboveground vegetation, but to provide relatively small effects on belowground carbon stores (Kingham, 2013). In addition, Kingham (2013) found that effects of grazing on above and belowground carbon stores were generally quite small compared to effects of abiotic factors. This is likely because changes in aboveground carbon stores take time to translate to changes in belowground carbon stores, and mechanisms such as plant compensatory growth do not occur instantly in response to a stressor (Holland et al., 1996).

In this study we investigate the use of livestock grazing as a management tool for saltmarsh climate regulation services, a concept familiar to terrestrial systems (Bhogal et al., 2011; Ma et al., 2006), and the extent to which the monetary valuation of this service can be translated into a financial tool (PES). In contrast to traditional PES schemes implemented in woodland and peatland ecosystems which rely on habitat restoration and creation, the study scenario only involves changing land use. A PES scheme for carbon benefits would operate by compensating farmers for the revenue lost due to reducing stocking density. Providing an economic incentive to farmers who manage saltmarshes to reduce grazing pressure would not only help combat climate change, but ensure that the managers themselves have a vested interest in the protection of the saltmarsh, thus limiting the need for extensive regulation. Nonetheless, a relevant effort in monitoring would be required to show that a saltmarsh managed under a PES project is providing additional benefits compared to the baseline, and fast, reliable and economic tools (such as simple spreadsheet) to do it must be proposed. Grazing management is already widely used as a conservation tool (Bouchard et al., 2003) and is easily adapted to address saltmarsh carbon sequestration, as light levels of grazing have the dual benefit of enhancing biodiversity and minimising the loss of aboveground carbon (Adnitt et al., 2007; Yu and Chmura, 2010).

The aim of this study is to assess the economic conditions or private economic benefits for saltmarsh agriculturists under a range of grazing regimes promoted by a PES scheme for above ground carbon emissions reduction and agri-environmental incentives subsidising reduced grazing pressure. Moreover, social benefits of saltmarshes protected by the SSSI scheme (Sites of Special Scientific Interest that best represent UK natural heritage in terms of flora, fauna, geology and geomorphology) are assessed through benefit transfer approach.

There are several reasons to consider a voluntary carbon PES for saltmarshes. For one, restoring and maintaining climate regulation services of saltmarshes can contribute to Scotland meeting carbon emission reduction targets laid out in the Climate Change (Scotland) Act 2009. At the same time, saltmarsh protection will help meet targets under the Habitats Directive (EC 1992). This is the first study to investigate the use of grazing management in saltmarshes for the purpose of climate regulation services. While the scope is limited to Scotland, as data and values are specific to this country, the methodology are applicable to a wider context.

The paper is structured in two parts: the first (methodology and results) illustrates the economic conditions under which the current voluntary carbon market prices may compensate the opportunity cost of reducing grazing and the social-economic benefits (quantified in monetary terms) arising from a bundle of ecosystem services (using a benefit transfer approach). The second part (discussions and conclusions) describes technical conditions under which a PES scheme can be applied to saltmarshes.

2. Materials and Methods

2.1 Study location

We use Scotland as a case study to explore the feasibility of a carbon market and to investigate geographic differences in associated opportunity costs. A recent survey of Scottish saltmarshes found that 39 of 249 saltmarsh sites were being overgrazed (Figure 1), and 166 failed condition targets to achieve Favourable Condition under the Habitats Directive (Haynes, 2016). One-hundred and twenty-three sites are currently designated as neither SSSI nor Special Areas of Conservation (SAC, from Habitat Directive 92/43/EEC), and it was in these sites that overgrazing was the biggest threat. Non-designated saltmarshes are located mainly on the west coast and offshore islands where enforcement of proper grazing management can be difficult (Haynes, 2016), and it is in these areas where innovative, non-traditional financing mechanisms can be explored to verify additionality to protection.

FIGURE 1

2.2 General methodological approach

To quantify the private economic benefit (money that accrues to farmers) obtained from a hectare of grazed saltmarsh under different grazing regimes, different livestock stocking densities ranging from high intensity agricultural use to minimal grazing enhancing biodiversity and carbon sequestration services are used. The set of benefits chosen account for livestock grazing revenue, carbon benefits, and ad-hoc agri-environmental subsidies for reducing livestock pressures. To make the results applicable to the real world, a range of market carbon prices are used to perform a sensitivity analysis of how benefits change according to different prices. The services with their relative methodology of valuation are described in the following section. Moreover, social economic benefits from reduced carbon emission are proposed by using social costs of carbon instead of carbon market prices. Subsequently, other benefits quantified as willingness to pay for provisioning, regulating and cultural services ecosystem services provided by SSSI protected saltmarshes are considered. The latter are benefits transferred by previous research carried out in the UK (Christie and Rayment, 2012) following a procedure described in section 2.3.4. Finally, section 2.3.5 describes how to calculate the net present value of the total services and the break-even carbon price required to compensate for the loss in agricultural revenue, and whether there are geographic differences across Scotland associated with the opportunity costs of livestock.

2.3 Methods for calculation of benefits from saltmarshes

2.3.1 Livestock benefits

We use livestock prices to calculate profits and opportunity costs associated with changing stocking density. Prices are taken from the June 2016 census of the Scottish agriculture report (RESAS, 2016) provided by the Agricultural Census Analysis Team. The census gives some data on a national level and some data on agricultural parish level. Parishes are geographical units based on Civil Parishes abolished in 1975 which continue to be used for statistical purposes and for the payment of farming grants and subsidies (“Agricultural Parishes,” 2017).

Information provided at national level includes standard output (SO), e.g. the average monetary value (farm-gate worth) in GBP (£) of livestock converted in Table S1 to SO per livestock unit (SO LU^{-1}) by dividing by LU of the livestock type (Chapman, 2007). LU is a reference unit based on feed requirements, where 1 LU is equivalent to an adult dairy cow (“Glossary: Livestock unit (LSU),” 2013), and allows easier analysis of stocking density than if animals are considered by head. However, the absence of information on the real composition (number of units) of the herds at national scale does not allow the computation of the weighted average SO per livestock. To provide the net benefit per head of livestock (e.g. taking the full cost of production into consideration), information from Table B4 of RESAS (2016), reporting income, costs, profits and number of heads of livestock at farm level, is used[1]. Finally, the net benefit per LU is obtained dividing the net benefit by 0.67, the arithmetic mean of the LU of six types of cattle assuming an equal proportion of each type thereby giving the average LU of cattle per hectare, and by 0.12 in the case of sheep, the arithmetic mean of the LU of two types sheep (Table S1).

To provide valuation per parish, we requested livestock units for the parishes of interest to RESAS. The proposed SO LU^{-1} per parish is a combination of cattle and sheep reflecting their proportion in that parish. Therefore, parishes containing a higher proportion of costly livestock have a higher SO LU^{-1}. Although parish SO LU^{-1} should be in the same range as national SO LU^{-1} (Table S1), preliminary analysis revealed for several parishes unrealistically high values. This could be due to errors during census compilation or because unlisted

[1] Looking at Less Favoured Areas (LFA) cattle farming, the average cost per cattle in Scotland is £569, while outcome or standard outputs is £475. Taking into account common agricultural policy subsidies (pillar 1 and 2), the outcome raise to £ 683. Under this circumstance, the net benefit becomes £ 114 per cattle. Subsidies are equal to £208 per cattle. Looking at the LFA sheep farming, average cost per sheep is £125, and revenue is £ 69 per sheep. Including CAP subsidies, revenues raises up to £150 per sheep. The net benefit of a sheep (considering subsidies) is £25. Subsidies are in this case equal to £ 81 per sheep.

agricultural outputs are being counted. We therefore decided to exclude parishes from analysis where calculated total SO differed by more than 50% from reported total SO, resulting in an analysis of 104 rather than 174 parishes. As with sheep only and cattle only scenarios we subtract the cost of inputs, calculated based on the proportion of each livestock type, for each to obtain net benefits from livestock.

Two sets of livestock prices are then used to calculate benefits from livestock: (i) the net national average SO LU^{-1}, split into exclusive cattle or sheep (Table S1) that do not provide any geographic information; and (ii) the SO LU^{-1} per parish, which account for the relative abundance of livestock in each parish that does expose geographic differences (Table S3). The different grazing regimes compared in this research are essentially a range of stocking densities expressed in terms of LU per hectare (ha). A guideline of average annual stocking rates for semi-natural habitats suggests grazing in saltmarsh in the range of 0.25-0.50 LU ha^{-1} yr^{-1} (Chapman, 2007). In accordance with scales employed by Kingham (2013), we assess carbon benefits for four levels of grazing: low (0.3 LU ha^{-1} yr^{-1}), moderate (0.6 LU ha^{-1} yr^{-1}), high (1.0 LU ha^{-1} yr^{-1}), and very high (2.0 LU ha^{-1} yr^{-1}).

2.3.2 Carbon benefits

To calculate carbon benefits, the quantification of how changing grazing pressure affects saltmarsh carbon stocks is needed. Kingham (2013) found that aboveground plant biomass decreased by 2.1 t ha^{-1} LU^{-1} across high, mid, low and pioneer saltmarsh in north Wales and north-west England. Plant biomass is converted to carbon given that plants almost always contain between 45-50% carbon (Magnussen and Reed, 2004). Multiplying by an average of 47.5%, this gives a reduction of aboveground carbon content due to grazing of 0.998 t C ha^{-1} LU^{-1}. We use data from England and Wales (not having more specific data from Scotland) to estimate these effects due to a lack of equivalent studies in Scotland, even though we expect spatial variability.

Additionally, an important component of the carbon budget to consider is the greenhouse gas emissions of livestock itself. Cattle and sheep emit significant amounts of methane, a greenhouse gas 25 times more potent than carbon dioxide, through enteric fermentation (Table S2; Brown et al., 2016). The carbon benefits thus consider two carbon sources: (i) decreased aboveground carbon due to grazing of saltmarsh vegetation, and (ii) carbon equivalent methane emissions of livestock. The carbon flux is then multiplied by market price or social cost of carbon to obtain carbon benefits.

Market prices of carbon in the EU Emissions Trading System have been in the range of £3-£25 t^{-1} C since 2007 and below £10 for the past five years (Marcu et al., 2016). Rather than testing a single market price, we perform a sensitivity analysis using four prices (£1 t^{-1} C, £2 t^{-1} C, £5 t^{-1} C, and £10 t^{-1} C) each at four discount rates (3%, 5%, 10%, and 15%). While market prices are paid to ecosystem service providers in a carbon emissions trading scheme, an alternative way of valuing carbon benefits is to consider the social cost of carbon (SCC). SCC refers to the economic damages incurred by emitting one tonne of carbon and is a cost taken on by society. If caps ensured efficient carbon removal levels, then market price and SCC would be the same (Nordhaus, 2017). A range of social costs at respective discount rates indicative of the current literature are considered (£8.04 t^{-1} C at 5%, £23.38 t^{-1} C at 3%, £37.26 t^{-1} C at 2.5%, and £65.76 t^{-1} C at 3%) (Nordhaus, 2017; Tol, 2009; van den Bergh and Botzen, 2014).

2.3.3 Agri-environmental subsidy

The UK Agri-Environment Climate Scheme compensates farmers providing environmental public goods under Pillar II of the EU Common Agricultural Policy (Reed et al., 2014). In particular, the Wetland Management Option rewards farmers who manage saltmarshes to benefit faunal and floral biodiversity with £90.03 ha^{-1} $year^{-1}$ ("Wetland Management," 2017). Although the scheme does not currently consider the carbon sequestration service of saltmarshes, reducing grazing pressure to benefit biodiversity incidentally ensures that more aboveground carbon is maintained (Kingham, 2013).

A requirement for farmers managing a site is to have an approved grazing regime individualised for each site depending on site-specific conditions. Owing to this, there is no set guideline LU ha^{-1} density below which a site qualifies. Stocking densities can also vary seasonally, for example to minimise conflict with wading birds during breeding season (Bouchard et al., 2003). We make the simplifying assumption that, of our four regimes, low and moderate grazing qualify for the scheme. Several studies have shown that low and moderate grazing increases biodiversity (Bouchard et al., 2003; Fleischner, 1994; Marty, 2005; Norris et al., 1998), suggesting the assumption is reasonable.

2.3.4 Valuation of other saltmarsh ecosystem services

Saltmarshes provide additional services on top of the provisioning service of livestock grazing and the regulating service of carbon sequestration. Using a series of choice experiments, Christie and Rayment (2012) determined that the public (sample from England and Wales) is willing to pay £450 ha^{-1} $year^{-1}$ to secure services currently delivered by SSSI conservation activities in coastal grazing marsh in England and Wales. These services include nature's gifts[2], research and education, climate regulation, water regulation, sense of experience, charismatic species, and non-charismatic species. Excluding climate regulation (to avoid double counting) and water regulation (the degree of which depends on the size of the site) this corresponds to £375.13 ha^{-1} to secure the remaining services. This value is corrected for Scotland applying a willingness to pay transfer with adjustment for income elasticity as given in Pearce et al. (2006):

$$WTP_{Scotland} = WTP_{England+Wales}(Y_{Scotland}/Y_{England+Wales})^{e} \quad (1)$$

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[2] In Christie and Rayment (2012) nature's gifts (mainly natural/spontaneous products of land) are comprehending several provisioning services described also by other ecosystem services approaches (MEA, 2005; TEEB, 2010; CICES, 2018), but do not include livestock grazing, while more recently the literature (see Diaz et al., 2018) see them as synonym of ecosystem services.

where WTP is the willingness to pay; Y is the gross annual pay for employees in England and Wales (£23,426) and Scotland (£22,918) (Aiton, 2016); and e is the income elasticity estimated at 1.3 for British households (Layard et al., 2008). This corresponds to a WTP of £364.58 ha^{-1} in Scotland.

While this value applies to services delivered by SSSI sites, only 49% of Scottish saltmarshes are under such designation. However, in both designated and non-designated sites, grazing is a common tool used to ensure the conservation services of saltmarshes, and where conservation is prioritized, light grazing is preferred as it gives greatest structural and biological diversity (Adnitt et al., 2007; Chatters, 2004). We therefore assume that sites managed under the more stringent low regime warrant the same delivery of services as SSSI designated sites. It is important to remember that moderate grazing still increases biodiversity compared to high and ungrazed systems, which is why we include it in qualifying for subsidy, but it is not the preferred option when conservation is the main goal.

2.3.5 Calculating benefit and break-even carbon prices

We determine the benefit from land use of a hectare of saltmarsh as:

$$NB(t) = B_L(t) - B_C(t) + B_S(t) + B_V(t) \quad (2)$$

where $B_L(t)$ denotes revenue from livestock minus depreciation cost; $B_C(t)$ is the carbon benefit lost due to plant biomass reduction and livestock emissions; $B_S(t)$ is the subsidy gained from agri-environmental schemes; and $B_V(t)$ is the valuation of other saltmarsh services. We do not mix measures of private and public benefits in the analysis: we only include $B_V(t)$ when assessing the social benefits of carbon reduction through the social costs of carbon, as WTP is an economic measure of public preferences for social benefits provided by saltmarsh services rather than a payment at the current market price.

We then calculate the benefits over a period of 30 years by discounting them into net present value with the formula:

$$\mathrm{NPV} = \sum_{t=0}^{T} \frac{NB(t)}{(1+i)^t} \quad (3)$$

where *t* is time and *i* are the discount rates chosen to reflect a range of social preferences to high rate of return achievable in agriculture (3%, 5%, 10%, and 15% for market prices, 2.5%, 3%, and 5% for social costs, see section 2.4). We employ a timeframe of 30 years rather than a longer one, as a scheme involving grazing management does not need to rebuild carbon in soil, as happens for drastic changes in planting systems, but rather affects changes in aboveground carbon which are faster (Palmer and Silber, 2012).

In addition to the net benefit, we calculate the break-even carbon price required to compensate the foregone revenue from livestock and subsidy at different stocking densities. The break-even carbon price will differ depending on the specific change in regime, such that we end up with six prices for six permutations of regime changes: moderate to low, high to low, very high to low, high to moderate, very high to moderate and very high to high.

Lastly, we compare how changes in benefits and break-even carbon prices differ across Scottish parishes. The break-even price relates to the livestock value (SO LU^{-1}) we calculated per parish (reflecting the real composition of the herd), as the price of carbon to offset loss would need to be higher the more revenue can be gained from livestock. To visualise which of the 174 parishes containing saltmarsh (Figure 1) have higher costs, we map changes in net benefits and break-even carbon prices per parish using ArcGIS 10.4.

3. Results

3.1 Carbon benefits

Carbon benefits over 30 years for various market carbon prices and discount rate for 1 LU ha^{-1} yr^{-1} are shown in Table 1. These represent the monetary value of (i) carbon emission avoided and (ii) the methane equivalent carbon emission avoided by livestock if grazing intensity is reduced by 1 LU ha^{-1} yr^{-1}. It is possible to estimate relative values as arising from changes in grazing regimes from the difference in benefits when transitioning from one regime to another. For example, the carbon benefits lost under a very high regime (2 LU ha^{-1} yr^{-1}) are the benefits in Table 1 multiplied by two, as the grazing intensity is twice as high. Thus, a transition from a very high to high grazing regime will result in carbon benefits equivalent to the values presented in Table 1. Insofar as these values represent the amount of money generated if a carbon market provided payments for reduced emissions due to reduction from very high to high grazing regime, these carbon benefits are quite low.

TABLE 1

Within each discount rate bracket, there is a linear relationship between carbon benefits and carbon price. Thus, the benefits lost for a carbon price of £10 t^{-1} C are ten times higher than the benefits lost for a carbon price of £1 t^{-1} C. As we would expect, the greater the carbon price, the greater the benefits lost. Cattle grazing incurred slightly higher carbon loss as 1 LU of cattle emits 51% more methane than 1 LU of sheep (Table S2). Notably, the benefits lost for cattle are only around 13% higher than sheep, since carbon lost due to reduction in vegetation is greater than the carbon emitted by livestock (see Section 2.4).

The carbon benefits lost using social costs of carbon (Table 2) were higher than when using market prices, which we expect given that the social costs we used were greater. Once again, the carbon benefits lost were higher when saltmarshes were grazed exclusively by cattle rather than sheep, as the former emit greater amounts of methane.

TABLE 2

3.2 Change in net benefits when altering grazing regimes

Table 3 illustrates how reductions in grazing regimes result in either loss or gain of net economic benefit from a hectare of saltmarsh over a period of 30 years. For a majority of cases where sheep and cattle grazing intensity is reduced from a higher regime to a lower regime, land managers can expect to lose money under the current range of market carbon prices (Table 3). This is because the value of carbon benefits lost at higher stocking density is much lower than the revenue gained from livestock. The greatest losses are incurred under a transition from very high to low regimes, as this is when the greatest change in stocking density occurs at a reduction of 1.7 LU ha^{-1} yr^{-1}. The reduction in net benefit is lower with higher carbon prices, as these equate to higher carbon benefits to offset revenue loss.

TABLE 3

However, in cases where the transition in sheep and cattle grazing is from high to moderate, land managers could actually gain net benefit (Table 3). According to these results, there is an economic incentive to reduce to a moderate grazing regime if grazing is currently on a high regime. The reason for this is that under our current study assumptions such a change allows a saltmarsh site to qualify for an additional £90.03 ha^{-1} $year^{-1}$ from the Wetland Management subsidy. The income from subsidy and carbon benefits is sufficient to offset the loss in revenue arising from a reduction in grazing of 0.4 LU ha^{-1} yr^{-1}, the difference between high and moderate regimes.

On the other hand, when we use social costs of carbon to assess changes in net benefits of different grazing regimes, there are more instances where it is economically feasible to reduce grazing pressure (Table 4). Without additional bundled ecosystem service co-benefits, decreasing grazing pressure is profitable under a high to moderate transition as before. When we include other benefits (valued at £364.58 ha^{-1}) to take into account the broad range of ecosystem services provided by protected SSSI sites, benefits increase under all transitions except very high to high and very high to moderate, although not under very high to moderate for cattle at highest social costs of carbon.

TABLE 4

3.3 Livestock opportunity costs

Examining the transition from high to moderate stocking density further, as this is the most favourable transition having the lowest opportunity cost and greatest increase in net benefit (Tables 3 and 4), the loss in livestock revenue that is incurred each year is given in Table 5. The figures represent the amount of money farmers would lose for reducing stocking density by 0.4 LU ha^{-1} yr^{-1}. Conversely, this is also the amount of compensation needed to incentivise a reduction in grazing pressure, whether provided through ecosystem service benefits or subsidiary aid.

TABLE 5

3.4 Break-even carbon prices

Break-even carbon prices were overall higher for sheep than for cattle (Table 6), as the former are more valuable when considered per livestock unit. Negative break-even price occurred under a transition from high to moderate sheep and cattle stocking densities, as sites then qualified for the additional £90.03 ha^{-1} compensation scheme which drove down the required carbon benefit to offset loss of agricultural revenue. The fact that these values are negative indicates that land managers are better off under the less intense regimes. In other cases, the positive break-even prices indicate the market price that carbon would have to be at to incentivise managers to reduce stocking density. The range of positive carbon prices in Table 6 are higher than the range of market prices we explored, but comparable to the range of social costs of carbon indicative of the literature. Therefore, if carbon removal levels were efficient and offset occurred at the full social cost, these would be sufficient to prompt a decrease in grazing pressure. The highest break-even prices occurred under a transition from moderate to low and very high to high stocking densities (Table 6). The prices of these two transitions are identical because both involve halving of stocking density and no addition or removal of subsidy.

TABLE 6

3.5 Geographic differences in opportunity costs

The calculated livestock value (standard output) per parish, under a composition of livestock in each parish as given in the agricultural census and accounting for cost for 104 parishes containing saltmarsh ranged from £112.68 LU^{-1} in Stenness to £1,527.70 LU^{-1} in Monkton & Prestwick[3] (Table S3). The big range between these values is determined mainly by the different compositions of herds. In some parishes, mainly in the lowlands, the presence of a high number of highly valuable resources causes the SO to be very high.

Transitioning from high to low stocking density at a carbon price of £1 t^{-1} C and 5% discount rate resulted in gain of net benefit in 4 out of 104 parishes (Figure 2), and 7 out of 104 at a higher price of £10 t^{-1} C (Figure S1). Thus, the range of market carbon prices we explore could potentially prompt reductions in select parishes with low value livestock.

FIGURE 2

Transitioning from high to moderate regimes instead, the number of parishes with benefit gains increases. At carbon price of £1 t^{-1} C and 5% discount rate with real herd compositions, 42 out of 104 parishes showed gains in net benefit (Figure 3). Considering the upper limit of market carbon prices of £10 t^{-1} C, 46 of the 104 parishes show gains in net benefit (Figure S2).

FIGURE 3

At the same time, we consider the required price of market carbon to induce reductions in stocking of parish herds from high to low (Figure 4) and high to moderate (Figure 5) transitions, the same scenarios considered in Figures 2 and 3. For high to low transitions, break-even price ranged from £-10.97 t^{-1} C to £963.53 t^{-1} C across parishes. For high to moderate transitions, which were also the most favourable transitions with lowest break-even price, price ranged from £-77.40 t^{-1} C to £897.10 t^{-1} C. As before, while negative carbon prices make little

[3] Differences in values are given by the different composition of livestock across the regions

sense in a market context, they indicate that moderate regimes are automatically preferable when the gain from subsidy is greater than the incremental loss from fewer livestock in parishes where value of livestock is low.

FIGURE 4

FIGURE 5

Geographic comparison of parishes shows that opportunity cost of reducing stocking density and required break-even carbon price were highest in the Solway Firth, Moray Firth and Firth of Clyde regions. Lowest values occurred on parts of the west coast and offshore islands, including the Inner Hebrides and small sections of the Shetland Islands and Orkney. Incidentally, these parishes with lower break-even carbon prices overlap with sites identified by the 2016 national saltmarsh survey as being threatened from overgrazing and lacking designated protection (Haynes, 2016). Although saltmarshes are also present on the east coast (Figure 1), Figures 2-5 do not show parishes due to lack of realistic data on livestock values.

4. Discussion

This section reviews results for the proposition of a carbon market (5.1) and contextualises them into a potential regional PES scheme (5.2). Limits of this study are finally introduced to show the validity of our results (5.3).

4.1 Viability of findings for a pure carbon market in mitigating saltmarsh grazing pressure

We set out to investigate whether a pure carbon market could prompt a reduction of grazing pressure on Scottish saltmarshes by calculating net benefits of land use and break-even carbon prices to compensate revenue loss. The break-even prices were directly related to opportunity costs, such that prices were lower for cattle grazing regimes than for more valuable sheep grazing regimes, and had both negative and positive values. Negative break-even prices were calculated under certain scenarios, particularly the most favourable high to moderate regime change, due to gains from subsidies being greater than the relatively small reduction in stocking density. This illustrates the role that agri-environment subsidies have in promoting less intense grazing management practice, and that there is no need for further compensation schemes when livestock value is low.

The greatest break-even prices of £143.46 t^{-1} C and £102.40 t^{-1} C for sheep and cattle respectively (Table 6) occur when there is no effect from public subsidy, such as in moderate to low and very high to high transitions. In such a case, the carbon price needs to compensate wholly for the value of livestock lost due to reduced stocking density. With these carbon prices in the range of social costs of carbon estimated by the literature, a carbon market in such a scenario would only be possible if market prices of carbon reflected true costs of carbon.

The results demonstrate the important role of subsidies and of a reduction of voluntary schemes, either due to exclusion from the Common Agricultural Policy because of Brexit or because grazing intensity is not reduced sufficiently. Under these circumstances, it would be more difficult for a pure carbon market to prompt a decrease in grazing pressure. On the other hand, if policy were rectified to recognize the importance of saltmarsh climate regulation services, either by establishing new agri-environmental schemes that farmers can apply for or by increasing the amount of money provided under the Wetland Management option, this would further drive down the required break-even carbon price. In this case, an increase in subsidiary aid could make further transitions in stocking density feasible.

In our analysis of livestock values based on real parish compositions (Figures 4-5), the lowest break-even carbon price was around £-77 t^{-1} C. However, parishes displayed a huge variability in break-even prices, such that even in the most favourable transition from high to moderate some parishes had break-even prices around £960 t^{-1} C. These results illustrate that a subset of parishes could well have pure carbon markets with price of carbon in international market ranges while still managing to incentivize reductions in grazing pressure, while in other parishes this is largely unfeasible due to high value of livestock. There is some promise in that lower break-even prices are found on the west coast and offshore islands (Figure 2), areas that are identified by Haynes (2016) as containing saltmarsh sites which lack designation and are threatened by overgrazing. These parishes then, having lower opportunity costs, will more easily elicit decreases in grazing intensity through the introduction of a PES scheme selling carbon emission reduction and other services. In the majority of parishes however, it appears rather unfeasible that a pure carbon market can provide carbon benefits at a high enough value to incentivise saltmarsh agriculturists to decrease grazing intensity. Moreover, the number of potential carbon credits that can be exchanged in this local market would be very limited and insufficient to compensate corporates emissions.

4.2 Towards a voluntary carbon PES scheme

There is a growing recognition that coastal agricultural landscapes provide important services other than food provision. This is not limited to the UK, but has been studied extensively worldwide (Power, 2010; Wylie et al., 2016). Moreover, although common agro-industrial practices are degrading ecological conditions and ecosystem services in the pursuit of food provision (Foley et al., 2005), society's demand and need for supporting and cultural services, aesthetic value and cultural identity are increasing (Reed et al., 2014). The benefit transfer analysis employed in this study indicates that adding to livestock revenue and carbon sequestration the benefits valued by Christie and Raymant (2012) significantly drives up the net benefit society can get from saltmarshes. At a WTP of £364.58 ha^{-1} (see section 2.3.4), the collective value of these services can compensate the negative benefits generated by a reduction of grazing as shown in the Figure 3.

Notably, perceptions on the importance of saltmarsh ecosystem services are highly influenced by people's perspective and level of education, and whether services are directly experienced and affect local livelihoods (Soy-massoni et al., 2016). This requires a specific analysis of the demand of the services more appreciated by stakeholders under different management regimes, as occurred for peatlands (Bonn et al, 2014). The valuation of services provided by a saltmarsh site is therefore highly influenced by local perceptions and conditions. When we considered the benefit transfer of services provided by SSSI sites, we simplified this by adapting the perceived social benefit from English and Welsh saltmarshes from Christie and Rayment (2012). To have a more accurate picture, however, the next step would be to assess social perception and willingness to pay for services in our study sites from different potential buyers, focusing possibly on broad domains with similar typology of farming rather than attempting to assess values for each parish individually.

To outline some possibilities, saltmarsh sites with high numbers of grazing geese may attract visitors such as bird-watchers or hunters who may be charged an entrance or hunting fee (Macmillan et al., 2004). Water or aquaculture companies benefitting from water filtration services may contribute to PES to avoid the need of artificial water filters (Locatelli et al., 2014). Communities protected from storms and waves may diverge funds otherwise needed to build hard sea defences (King and Lestert, 1995). As these examples indicate, the beneficiaries and the particular services that are valued will vary in different places and geographical contexts.

Once beneficiaries and the main services demanded are identified, it is possible to figure out a specific market for saltmarshes services. Here we focus on voluntary carbon market. Compared to the compliance carbon markets developed under the Kyoto protocol, voluntary markets have more opportunities to apply to local land use projects and higher flexibility to bundle benefits. Moreover, schemes under the UN framework such as Joint Implementation, Clean Development Mechanisms and Reduced Emissions from Deforestation and Degradation work directly with and through national government processes and usually require a threshold of thousands of carbon credits, difficult to reach for coastal project (Wylie et al., 2016). Conversely, international voluntary standards for blue carbon projects are usually based on tailored methodologies (usually validated by Plan Vivo and Verified Carbon Standard -VCS) and prove to be easier to implement due to the higher flexibility and lower costs of the required carbon accounting, verification and certification. VCS has so far provided general standards for land-based climate change mitigation projects. Moreover, VCS VM0024 provides GHGs assessment for coastal wetland creation, while the VM 0033 (2015) for tidal wetland restoration. Recently, adaptations to VCS standard have been proposed by Reed et al. (2013) to verify changes in GHGs fluxes resulting from peatland restoration. However, verification and accreditation remain financially feasible only for large-scale projects and big investors (Wylie et al., 2016). To overcome these difficulties, regional voluntary agreements are emerging; they may be national or more localised markets targeting local/national investors under new or adapted existing standards to tailor specifically to local restoration schemes (Kosoy and Guigon, 2012). Examples for the UK are the peatland and woodland restoration projects (Forestry Commission, 2017; IUCN, 2015). An advantage of these markets is that the standards proposed are often based on VCS guidance, but use their own empirical or modelled measurement to verify carbon emission reduction: they are rigorous for local investors, but more cost effective than the VCS standard. Finally, they often target more services, not having the possibility to reach a critical threshold for only one (for example high volume of carbon credits sold).

We propose in order to promote a reduction in grazing pressure a combination of private transactions and subsidiary aid to compensate for the loss in agricultural revenue. This can be seen as a renovated agro-environmental scheme as proposed by the Welsh Assembly based on spatially targeted approach to land management (Reed et al., 2014) and voluntary payments from corporates as proposed under the Peatland Code. Payments would include a public subsidy for the reduction of grazing (to moderate or low pressure), and then, according to the cost opportunity of grazing, a variable price for carbon voluntarily paid by companies to compensate the forgone income. Differently from conventional agro-environmental schemes there is the introduction here of outcome-based (carbon regulation) payments that would also be able to provide additional benefits as those quantified for the SSSI protected saltmarshes.

Bundling carbon sequestration with other services requires identifying whether low levels of grazing conflict with other supporting, regulating, and cultural services. As already mentioned, there is good evidence that low levels of saltmarsh grazing benefits biodiversity (Adnitt et al., 2007; Jones et al., 2011). There is some evidence that nutrient cycling in coastal grasslands is maximised under ungrazed regimes (Ford et al., 2012) and that grazing slows down microbial turnover (Olsen et al., 2011), although classic theory dictates that intensive grazing promotes faster nutrient cycling than light or ungrazed regimes (Bardgett et al., 1998). Flood control is diminished with high levels of grazing as large herbivores such as cattle cause soil compaction and decrease soil infiltration rates (Carroll et al., 2004). This suggests that perhaps low levels of small herbivore grazing such as sheep may have smaller or negligible negative effects. Grazing effects on cultural services depend somewhat on the cultural aspect being valued. For example, the cultural service of environmental appreciation of forb and flower abundance have been found to be maximised under extensively grazed regimes (Ford, 2012), but extensive grazing is known to decrease abundance of wading birds (Chapman, 2007). There is the need, therefore, before valuing bundled ecosystem services, for further research to identify the services that trade-off and those that complement when aiming for a lessening of grazing intensity.

Other than considering co-benefits, several other aspects must be addressed to achieve a well-functioning PES scheme. Incentives that encourage production on one service may have adverse effects on others (for example a reduction in biodiversity- Kinzing et al., 2011; Chan et al., 2017). For instance, trade-offs caused by intensity of grazing between biodiversity (Ford et al., 2012; Sharps et al., 2016), fruition of the saltmarshes (van Zenten et al., 2016) and appreciation of the landscape and tourist attraction (Ryan 2011) are documented. Incentives might drive also the displacement of activities (leakage) damaging environmental service provision to areas outside the geographical zone of PES intervention (Engel et al., 2008); and finally incentives might induce the argument that polluters must be paid for conservation reducing intrinsic or altruistic motivations (Chan et al., 2017).

Success of PES initiatives is then reliant upon filling the gap between management aspects and scientific knowledge of the ecosystem services (Kinzig et al., 2011) and the need to establish standards, develop tools, metrics and methods to reduce transaction costs (Naeem et al., 2015). Here we want to address some of these issues, focusing attention on, (i) additionality, (ii) how to ensure permanence by reducing external stressors, and (iii) monitoring. Under (i) it is necessary to show that the project would not occur without private investments. This would be applicable in those areas where there is not yet established legal protection (for example the Outer Hebrides); where protection under the EU Habitat Directive is in force, but significant grazing pressure and physical degraded soil is evident (Reed et al., 2014); or, in case of legal test fulfilled, where companies are able to significantly support public subsidies. The contribution from private sponsors can be decided in the rules of the market and must not necessarily be a high amount (for example in the case of the Peatland Code this is set at 15% of project cost-Reed et al., 2013). According to the results reported in Figures 4 and 5, many saltmarshes in the Highlands and Outer Hebrides could be supported towards a transition to a low grazing regime with a carbon price within 100 $t^{-1}C$. Although this price is far from being a private market price, but close to the social cost of carbon, this is equivalent to £ 40 ha^{-1} for each ton of carbon emission reduced, an amount which small companies can invest to raise social responsibility. For those areas characterised by higher opportunity costs (see Figures 4 and 5), where the break-even carbon price is set at £500-1,000 $t^{-1}C$, this can be translated in a support of £ 200-400 ha^{-1}, an amount that can be exchanged by involving bigger companies, trusts, NGOs and more support from public bodies.

PES must account for risks to permanence (point ii), and in particular natural risks caused by climate change and different patterns of erosion that might reduce the area dedicate to grazing. A series of strategies must be adopted to minimise these risks, from the evaluation of exposure and vulnerability of saltmarshes to external stressors, to their mitigations provided by a larger scale of participants; from the formulation of landscape-scale PES schemes to the promotion of financial instruments such as insurance or PES credit buffers to replace or make alternative payments for a failed project (Friess et al., 2015). Other forms of risks are internal to the project management such as lack of sufficient funds to meet the opportunity costs of reduced grazing or reversal to higher intensive use in habitats outside the area of intervention. This is one of the seven concerns raised by Chan et al (2017) upon a PES scheme, where any new markets or system of incentives creates new externalities. However, there is building evidence that the latter can be addressed by promoting the roles of farmers as stewards of the countryside, and rewarding patterns of behaviour (Newtown, 2011) rather than environmental outcomes. This could facilitate the attestation of long-term commitments to reduced grazing effort that subsides paying the full economic costs cannot address, contrarily to what is assumed by mainstream economics (Goldman-Benner et al., 2012; Wunder, 2013; Ferrraro and Kiss, 2002). An alternative externality to a PES applied to saltmarsh grazing is the relocation of grazing from natural pasture to stable production, removing the benefits of methane emissions reduction from reducing ruminant. The additionality of this service, at least internally to the country, can be guaranteed under a national cap system (quotas) that reduces the total number of herds. However, it seems impossible to guarantee additionality if the reduction of grazing in the UK is compensated by an equivalent increase in another country.

Finally, to guarantee cost-effective monitoring of GHGs emissions reduction (point iii), it is essential to develop a protocol that facilitates indirect valuation of GHGs to reduce the costs of direct measurements or process-based computational models (Bonn et al., 2014), expected to be relevant in those contexts (such as saltmarsh habitats) where carbon emissions show high variability over space and time. Proxy variables can be specifically developed for GHGs emissions and tailored to the regional context as proposed for the valuation of carbon emissions change of aquatic restored habitats[4]. For our specific case, it is possible to propose an empirical model fitting data from UK studies on management of saltmarshes, biogeochemical parameters (water table level, soil moisture, soil temperature, etc.), and ecological gradients (species composition).

The above three points align the saltmarsh PES scheme to the recent PES description suggested by Wunder (2015) as a voluntary agreement between service users and providers showing the property to generate outputs conditional on agreed rules; to provide monitoring by simple tools (reckoner) at reasonable transaction costs; and to look to effects *offsite* the market intervention taking into account and tackling environmental externalities.

A final consideration is about the importance of valuation to establish a PES. Although valuation is not always needed to create a market (Vatn, 2010), a potential cause of PES failure could be due to the difficulties associated with the agreement of prices of the services to be exchanged. A sound valuation of ecosystem services, under deliberated approach (Spash, 2008; Kenter et al., 2017) or reverse auction (Stoneham et al., 2003), could facilitate agreement between the parties, reducing transaction costs, and providing fair prices (usually lower than those paid in a fixed-price system) that reflect cost of management in a context of intrinsic motivation (Chan et al., 2017). Valuation is not relevant also when trading is not the mechanism chosen to carry out a transaction of ecosystem services (Wunder, 2015). This is true for the Peatland Code (Reed et al., 2013) and the PES scheme proposed in this paper, where private financial transactions are supposed to be made by corporates for enhancing social responsibility rather than trading carbon credits for offsetting CO2 emissions. Moreover, in the case of saltmarshes, where the ecosystem provides a wide range of services that can vary across space and time, it is often difficult and time- and resource-consuming to provide accurate valuations

[4] Examples of rapid reckoners for GHG emissions are under development for coastal wetland in the US. https://www.mass.gov/blue-carbon-calculator

and to disentangle one service from another (Environment Bank, 2015). Because of this difficulty, the Deben Estuary PES scheme has opted for the cost of saltmarsh restoration rather than valuation of saltmarsh services (Environment Bank, 2015). The Peatland Code allows also companies to sponsor emissions reductions with prices reflecting costs of restoring degraded peatland, the costs differing depending on how degraded the habitat is (IUCN, 2015).

In the case of the peatland scheme, companies can receive deferred indirect economic benefits by improving corporate social responsibility rather than direct financial benefits. In our case study, where we explore changes in land use in the form of grazing management rather than saltmarsh restoration through managed realignment or other practices, there are no equivalent costs of restoration. However, if it can be established that the summative value of services provided is higher for lightly grazed than for more intensively grazed saltmarsh, the price of the payment could reflect the opportunity cost of reducing grazing. The values in Figures 4 and 5 give the amount of money required to compensate a transition from high to low and high to moderate stocking density, respectively. As long as a combination of subsidies and PES provide that amount, it does not necessarily matter what proportion of the amount is from the value of carbon sequestration versus from other services. In a way, this is comparable to prices reflecting restoration costs in peatland projects.

4.3 Assumptions and limitations

It is important to highlight that the results given here are highly dependent on a series of assumptions introduced to simplify complex real-world grazing practices into a format that could apply more generally to typical saltmarshes. The reason why a transition from high to moderate and low stocking density was feasible is because we assumed sites stocked at a density < 0.6 LU ha^{-1} yr^{-1} can obtain financial support from the Wetland Management agri-environment option. It may be argued that such an assumption is precarious, especially since grazer type, local conditions and seasonal variation determine whether a particular grazing intensity is harmful for biodiversity at a particular site. We could not obtain the average stocking density of sites currently managed under this option due to an absence of data. However, even with this assumption the underlying logic for the trends we observe and discuss holds true. Essentially, if a decrease in grazing pressure is sufficient to allow farmers to claim subsidies, this drives down the required carbon price to a value that might guarantee private voluntary payments able to compensate forgone income and guarantee a bundle of benefits. The specific value of this carbon price will depend on local conditions (opportunity cost in each parish) and on the degree to which grazing intensity is reduced.

Net benefits from reduced grazing in the model are influenced by reduction in methane emissions from ruminants and results hold (e.g. can be considered additional) assuming that relocation of the reduced herds from the field to stable is avoided.

We assume the full ecosystem services of SSSI protected sites are provided by sites grazed under low grazing regimes, even if these sites are not currently under SSSI designation. We did this to obtain an approximation of the social benefit that saltmarsh sites provide, but future work will have to investigate how the value of services differs between protected and non-protected sites. We also assume that the value of livestock grazed on saltmarsh is equivalent to national average livestock values. Moreover, it is important to keep in mind that the enforcement of any stocking density on a saltmarsh site can be more effective in either privately owned land or saltmarsh with clear fenced designations. PES is not intended as a silver bullet that can address any environmental problem. Where it is missing the authority to manage ecosystems, a prior condition to the success of PES is to enforce property rights (Engel et al., 2008). Sites where any agriculturist can freely graze their livestock cannot receive PES unless there is a record of total grazing pressure over the year.

There are some limitations to the grazing effect on carbon sequestration we adapted from a study of English and Welsh saltmarshes (Kingham, 2013). The reduction in aboveground vegetation due to grazing (for further detail see section 2.2) was a single effect without a confidence interval or probability distribution, preventing us from using a statistical approach. Saltmarshes in the UK are classified into four bio-geographical regions with distinct vegetation communities: western, south-eastern, western Scottish and eastern Scottish (Adam, 1990). The effect of grazing on Scottish saltmarsh may therefore differ slightly from the effects observed in the English and Welsh region. As there is currently no study which has explored this for Scotland specifically, there is a need for further research in this regard. Such a study must consider effects on both aboveground and belowground stores, considering soil carbon in coastal ecosystems is a very important component of blue carbon accounting (Pendleton et al., 2012). The subsequent model of grazing effects should then be validated with on the ground collection of primary data. Further socio-economic research, for example via use of questionnaires, could be used to validate the assumptions proposed by the model in the regions that show promise in incorporating alternative financing schemes.

5. Conclusions

This study has proposed for the first time the possibility to introduce a PES scheme to support the provision of ecosystem services provided by saltmarshes under reduced intensive agricultural use. In particular, it has focuses on three major aspects: 1) the quantification of the impact that carbon price have on addressing the change in grazing regime; 2) the impact of the provision of other public services on the possibility to reduce livestock grazing ; and 3) considerations to be addressed to implement a sub-national scale PES for saltmarshes.

This study, then, other than describing the economics behind the possibility to reduce grazing pressure in saltmarshes, contributes to the growing body of work investigating the creation of new markets to enable private investments in ecosystem services provision. Across the globe, various management tools using an ecosystem services framework are being used for the purposes of conservation and sustainable resource use, including marine spatial planning, ecosystem-based management, and integrated coastal zone management (Granek et al., 2009;

Post and Lundin, 1996). As a part of this toolset, Payments for Ecosystem Services provide an incentive-based mechanism promoting nature conservation (Lau, 2013). Blue carbon initiatives are becoming common under the Kyoto Protocol Clean Development Mechanism, whereby climate mitigation projects are carried out in developing countries (Palmer and Silber, 2012). Blue carbon restoration projects of mangrove, seagrass and saltmarsh habitats in countries such as Kenya, Vietnam, Madagascar and India help alleviate poverty, finance conservation, and ensure the continuing provision of climate mitigation services (Hejnowicz et al., 2015; Lau, 2013; Wylie et al., 2016). While our study focused on saltmarsh grazing in Scotland, the management of saltmarsh grazers for carbon sequestration purposes could be investigated in other countries using the same approach, both developing and developed, for blue carbon initiatives.

While there is currently a Peatland Code and Woodland Code for carbon payments towards habitat restoration in the UK, our study provides some initial information that may help contribute to the establishment of a Saltmarsh Code. To date, PES schemes have largely ignored saltmarshes, despite their suitability given the variety of ecosystem services they provide. We investigated one aspect of a potential saltmarsh PES scheme, involving managing livestock stocking density and the provision of carbon benefits in exchange for lessening grazing pressure. Where only the change in climate regulation service is considered, the price of carbon required to compensate a decrease in grazing is higher than can be expected in voluntary carbon markets. However, it is possible that regional markets specifically tailored for saltmarshes could establish much higher prices (this must be explored studying the demand for saltmarshes services) or that common voluntary carbon prices can be driven down by the addition of agri-environment subsidies and by bundling payments of other ecosystem services. Finally, converting break-even price of a ton of carbon to costs per hectare when grazing is reduced from 1 to 0.6 LU ha^{-1} shows to be a not so high cost, especially in those areas characterised by a limited opportunity cost such as Western Scotland, and some internal uplands areas of the

Highlands and the Hebrides. Notwithstanding the possibility to address some interventions in these rural areas, the area affected and consequently the herd number is limited.

Under the scenario of grazing reduction from 1 to 0.6 LU ha^{-1} we expect to have a reduction of 9,260 heads of which 51% cattle (4,759 units) and 49% sheep (4,501 units). These reduced cattle units correspond only to 1.3% of the total production in the 174 parishes where saltmarshes are present, and to 0.26% of the entire Scottish production. Moreover, the reduction in sheep is 0.35% of the production in the 174 parishes where saltmarshes are present and 0.067% of the global Scottish production. Although it is not aim of this research to quantify the economic impact on the agricultural sector, considering the very small reduction in livestock units, it is unlikely that these figures will have a relevant impact on trade balance, negative effect on food security and distortion on prices. Conversely, the rewards by public subsidies and private incentives can help sustain income and reduce pressure on a sensitive habitat favouring the production of environmental benefits and contributing to provide a new role to farmers as local stewards of nature rather than mere producers of market commodities.

In its early stages, the PES saltmarsh market is unlikely to function as a formal carbon offsetting tool. Instead, it allows third parties to demonstrate corporate and public social responsibility and to benefit from services which could lessen costs of, for example, water filtration or sea defences. As saltmarsh carbon projects become more common, they may more easily gain verification and accreditation, to name one example with the Verified Carbon Standard (VCS, 2017), allowing the transfer of carbon emission reduction credits in an emissions trading scheme. The Woodland Code started in a similar fashion and now allows projects to generate tradable carbon, while the Peatland Code is on its way to this stage (IUCN, 2015). In this way, although a pure carbon market appears unfeasible, a premium carbon market offering bundled ecosystem services may help reduce grazing pressure on Scottish saltmarshes, thereby providing globally important climate regulation services and at the same time protecting sensitive habitats.

7. Acknowledgements

The study proposed is described in Dominic Muenzel's MSc thesis submitted as a partial fulfilment for the degree of MSc Ecosystem-Based Management of Marine Systems, University of St Andrews (August 2017).

The authors thank Paul Gavin and the Agricultural Census Analysis Team for providing the census data from the 2016 Scottish agricultural report and 4 anonymous reviewers whose comments have strengthened the content and the quality of the paper.

Table 1 Carbon benefits lost under a range of market prices│ Value of carbon emission avoided from 2015-2045 from 1 ha of saltmarsh when grazing is reduced by 1 LU ha^{-1} (high grazing regime) using a range of market carbon prices. Benefits for low, moderate, and very high regimes are calculated by multiplying values reported in Table 1 by respective stocking densities.

Discount rate (%)	Carbon price (£ t^{-1} C)	Carbon benefits of sheep grazing (£)	Carbon benefits of cattle grazing (£)
3	1	29.91	33.93
	2	59.83	67.86
	5	149.56	169.66
	10	299.13	339.31
5	1	23.77	26.97
	2	47.55	53.93
	5	118.87	134.84
	10	237.74	269.67
10	1	15.14	17.17
	2	30.28	34.35
	5	75.70	85.87
	10	151.40	171.74
15	1	10.99	12.46
	2	21.97	24.92
	5	54.93	62.31
	10	109.86	124.62

Table 2 Carbon benefits lost under a range of social costs of carbon│ Value of carbon emission avoided from 2015-2045 from 1 ha of saltmarsh when grazing is reduced by 1 LU ha^{-1} (high grazing regime) using a range of social costs of carbon. Benefits for low, moderate, and very high regimes are calculated by multiplying values reported in Table 2 by respective stocking densities. Carbon prices and discount rate are from Nordhaus (2017); Tol (2009); van den Bergh and Botzen (2014).

Discount rate (%)	Carbon price (£ t^{-1} C)	Carbon benefits of sheep grazing (£)	Carbon benefits of cattle grazing (£)
5	8.04	191	217
3	23.38	699	793
2.5	37.26	1,187	1,346
3	65.76	1,967	2,231

Table 3 Opportunity costs of changing grazing regimes with market carbon prices

Change in net benefits (in GBP) over 30 years caused by a change in sheep and cattle grazing regimes. Positive values in bold indicate gains in net benefit.

Discount rate (%)	Carbon price (£ t^{-1} C)	Change in grazing regime: moderate to low	high to low	very high to low	high to moderate	very high to moderate	very high to high
Sheep							
3	1	-1,279	-1,129	-5,390	**150**	-4,112	-4,262
	2	-1,270	-1,108	-5,340	**162**	-4,070	-4,232
	5	-1,243	-1,045	-5,187	**198**	-3,944	-4,142
	10	-1,198	-940	-4,933	**258**	-3,735	-3,993
5	1	-1,016	-897	-4,284	**119**	-3,268	-3,387
	2	-1,009	-880	-4,244	**129**	-3,235	-3,363
	5	-988	-830	-4,122	**157**	-3,135	-3,292
	10	-952	-747	-3,920	**205**	-2,968	-3,173
10	1	-647	-571	-2,728	**76**	-2,081	-2,157
	2	-643	-561	-2,703	**82**	-2,060	-2,142
	5	-629	-529	-2,625	**100**	-1,996	-2,097
	10	-606	-476	-2,497	**130**	-1,890	-2,021
15	1	-470	-414	-1,980	**55**	-1,510	-1,565
	2	-466	-407	-1,961	**59**	-1,495	-1,554
	5	-456	-384	-1,905	**73**	-1,449	-1,521
	10	-440	-345	-1,812	**95**	-1,372	-1,466
Cattle							
3	1	-1,032	-554	-3,994	**478**	-2,962	-3,441
	2	-1,022	-530	-3,937	**492**	-2,915	-3,407
	5	-991	-459	-3,764	**533**	-2,772	-3,305
	10	-941	-340	-3,475	**601**	-2,535	-3,135
5	1	-820	-440	-3,174	**380**	-2,354	-2,734
	2	-812	-421	-3,129	**391**	-2,316	-2,707
	5	-788	-365	-2,991	**423**	-2,203	-2,627
	10	-748	-270	-2,762	**477**	-2,014	-2,492

10	1	-522	-280	-2,022	**242**	-1,499	-1,741
	2	-517	-268	-1,992	**249**	-1,475	-1,724
	5	-502	-232	-1,905	**270**	-1,403	-1,673
	10	-476	-172	-1,759	**304**	-1,283	-1,587
15	1	-379	-203	-1,467	**176**	-1,088	-1,264
	2	-375	-195	-1,446	**181**	-1,070	-1,251
	5	-364	-168	-1,382	**196**	-1,018	-1,214
	10	-345	-125	-1,276	**221**	-931	-1,151

Table 4 Change in net social benefit of grazing regimes | Change in net benefits (in GBP) over 30 years when sheep and cattle grazing regimes are changed using social costs of carbon. For each livestock type, scenarios without additional service valuation and with SSSI valuation from Christie and Rayment (2012) are presented.

Discount rate (%)	Carbon price (£ t^{-1} C)	Change in grazing regime: moderate to low	high to low	very high to low	high to moderate	very high to moderate	very high to high
Sheep							
without SSSI valuation							
5	8.04	-966	-780	-4,000	**186**	-3,034	-3,220
3	23.38	-1,078	-660	-4,252	**418**	-3,175	-3,592
2.5	37.26	-1,015	-393	-3,775	**622**	-2,761	-3,382
3	65.76	-697	227	-2,097	**925**	-1,400	-2,325
with SSSI valuation							
5	8.04	**5,003**	**5,189**	**1,969**	**186**	-3,034	-3,220
3	23.38	**6,433**	**6,851**	**3,258**	**418**	-3,175	-3,592
2.5	37.26	**6,981**	**7,602**	**4,220**	**622**	-2,761	-3,382
3	65.76	**6,813**	**7,738**	**5,413**	**925**	-1,400	-2,325
Cattle							
without SSSI valuation							
5	8.04	-763	-307	-2,852	**456**	-2,088	-2,545
3	23.38	-804	-22	-2,703	**782**	-1,899	-2,681
2.5	37.26	-706	327	-2,025	**1,033**	-1,319	-2,353
3	65.76	-373	984	-259	**1,357**	**114**	-1,243
with SSSI valuation							
5	8.04	**5,206**	**5,662**	**3,117**	**456**	-2,088	-2,545
3	23.38	**6,706**	**7,488**	**4,807**	**782**	-1,899	-2,681
2.5	37.26	**7,290**	**8,323**	**5,970**	**1,033**	-1,319	-2,353
3	65.76	**7,138**	**8,495**	**7,252**	**1,357**	**114**	-1,243

Table 5 Opportunity cost of livestock│ The value lost every year in agricultural revenue under a transition from high to moderate stocking density.

Discount rate (%)	£ lost per year	
	Sheep	Cattle
3	57.22	46.33
5	45.48	36.82
10	28.96	23.45
15	21.02	17.01

Table 6 Break-even carbon prices to offset opportunity costs│ Required prices of carbon (£ t^{-1} C) to offset the opportunity cost of having reduced livestock. At these carbon prices farmers would not lose any money changing from one grazing regime to the next. At any higher prices farmers would gain money by reducing grazing pressure.

	Change in grazing regime					
	moderate to low	high to low	very high to low	high to moderate	very high to moderate	very high to high
Sheep	143.46	54.90	107.00	-11.53	99.18	143.46
Cattle	102.40	24.31	70.24	-34.26	63.35	102.40

Figure 1. Scottish saltmarsh distribution| Map of 249 saltmarsh sites and 174 parishes containing saltmarsh across Scotland. West coast sites are smaller but more numerous than east coast sites. Sites marked red indicate overgrazed saltmarshes (Haynes, 2016).

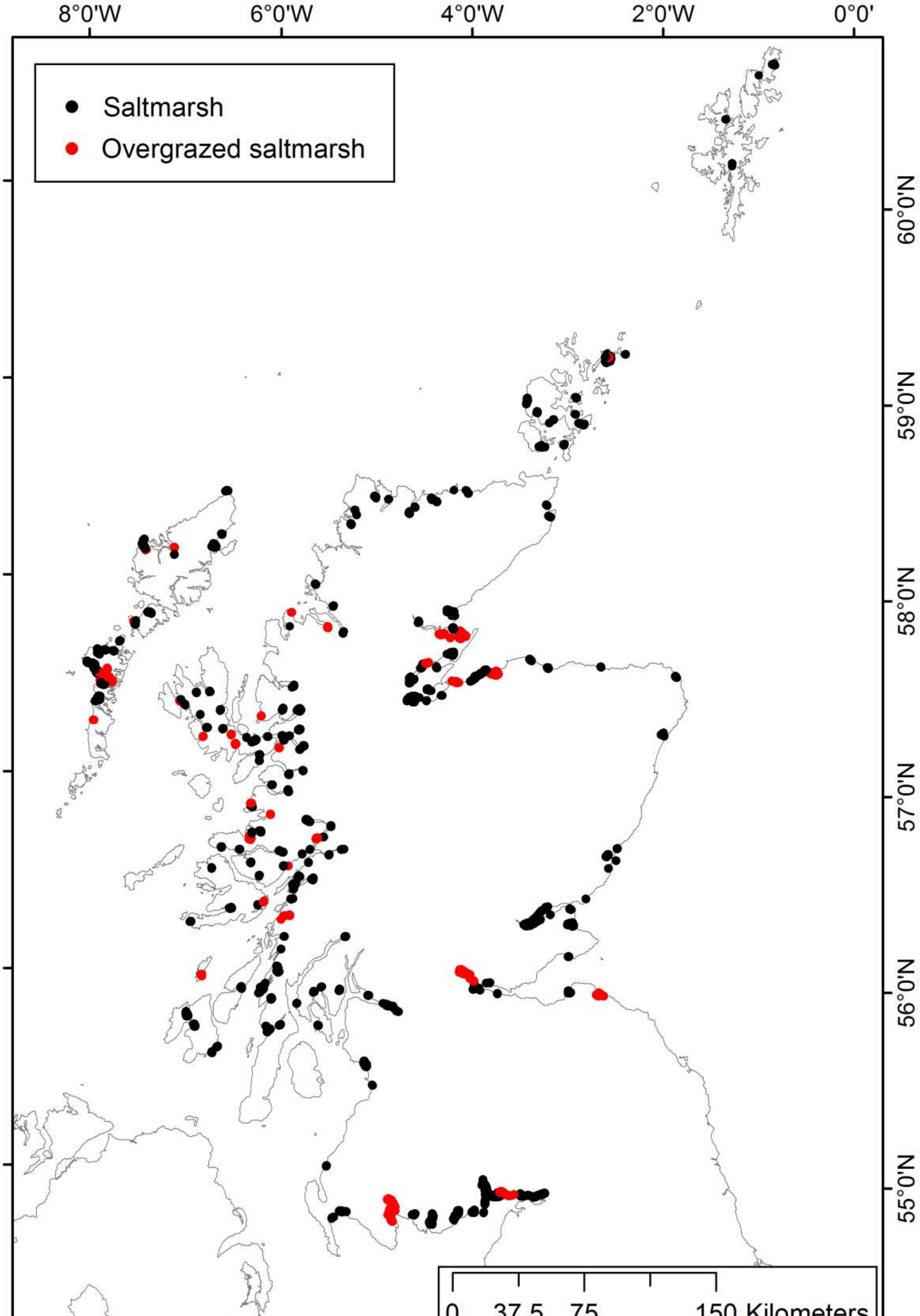


Figure 2. Change in benefits from high to low regime transitions | Map indicating the change in net benefit over 30 years arising from a transition from high to low grazing regime with a carbon price of £1 per t C at 5% discount rate.

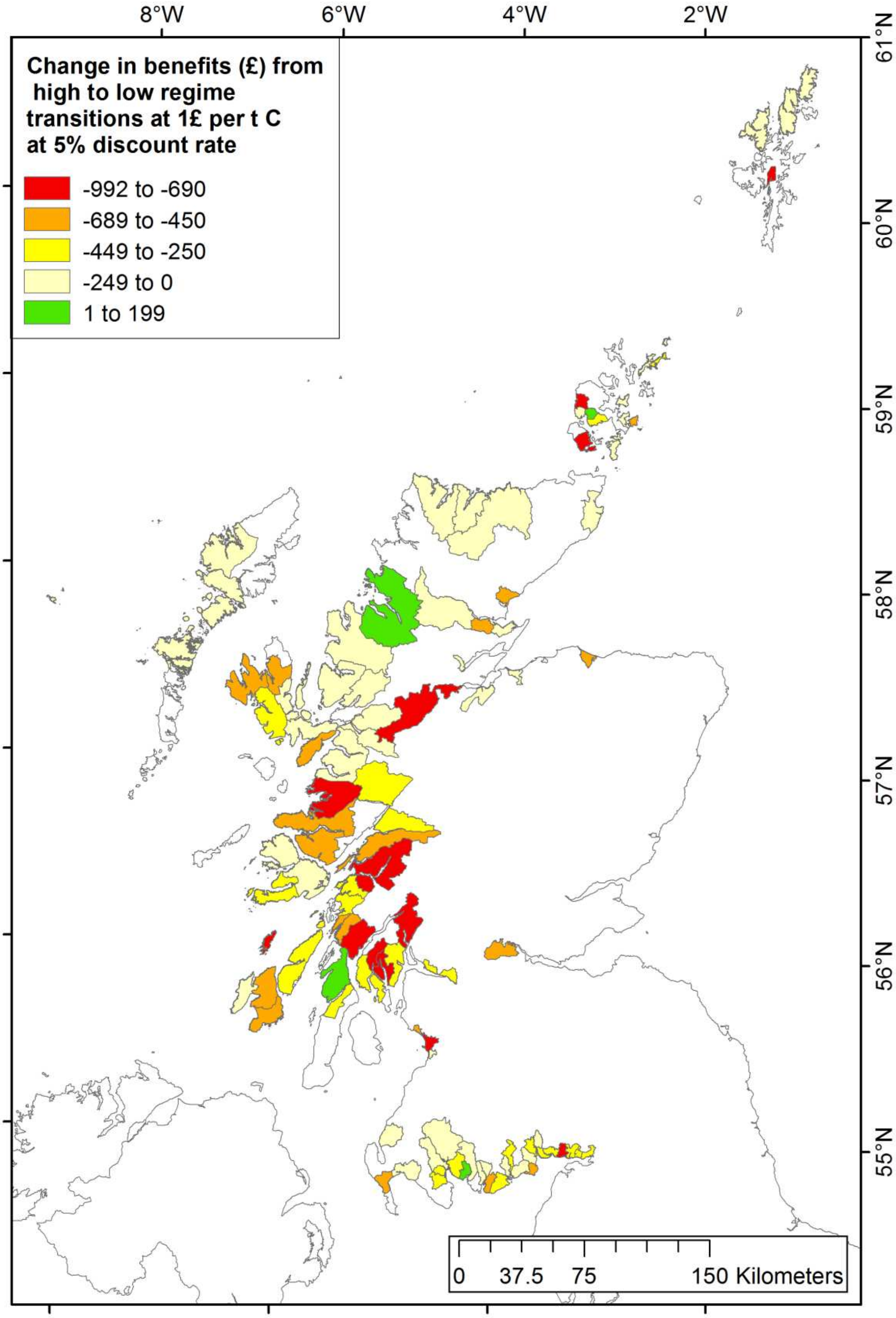


Figure 3. Change in benefits from high to moderate regime transitions| Map indicating the change in net benefit over 30 years arising from a transition from high to low grazing regime with a carbon price of £1 per t C at 5% discount rate.

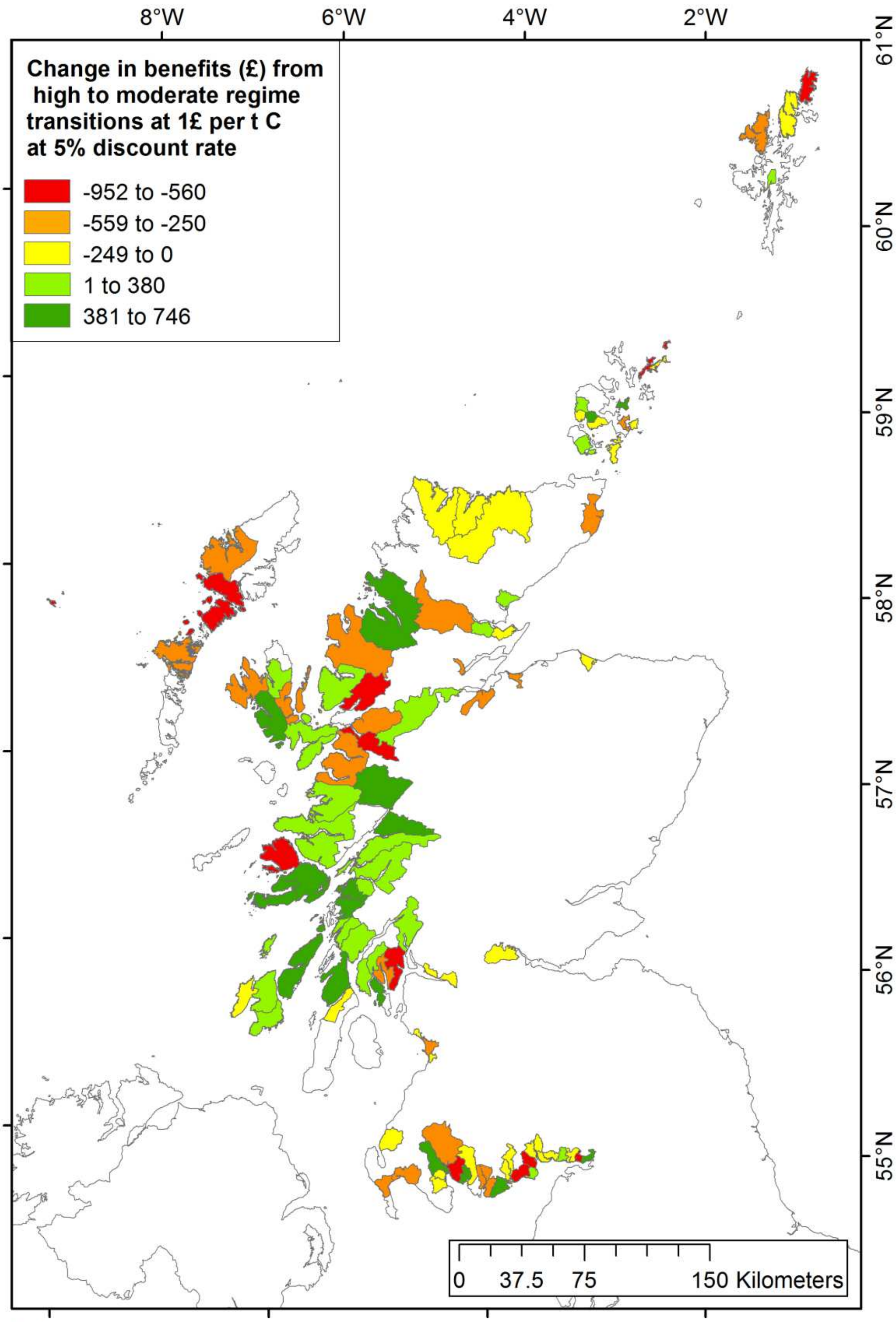


Figure 4. Break-even carbon prices for high to low regime transitions | Map indicating the required price of carbon to compensate for loss in agricultural revenue for a transition from high to low grazing regimes.

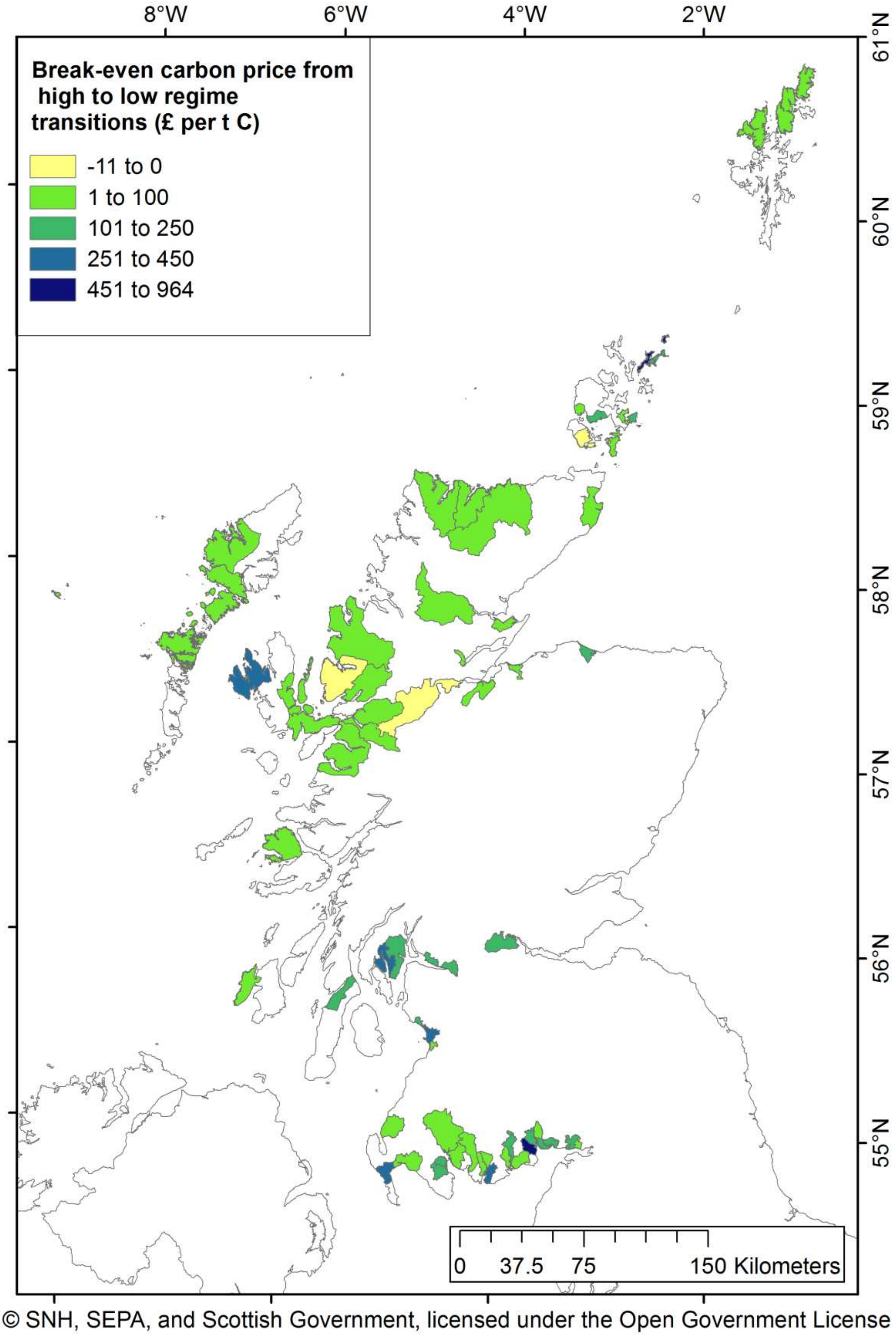


Figure 5. Break-even carbon prices for high to moderate regime transitions| Map indicating the required price of carbon to compensate for loss in agricultural revenue for a transition from high to moderate grazing regimes.

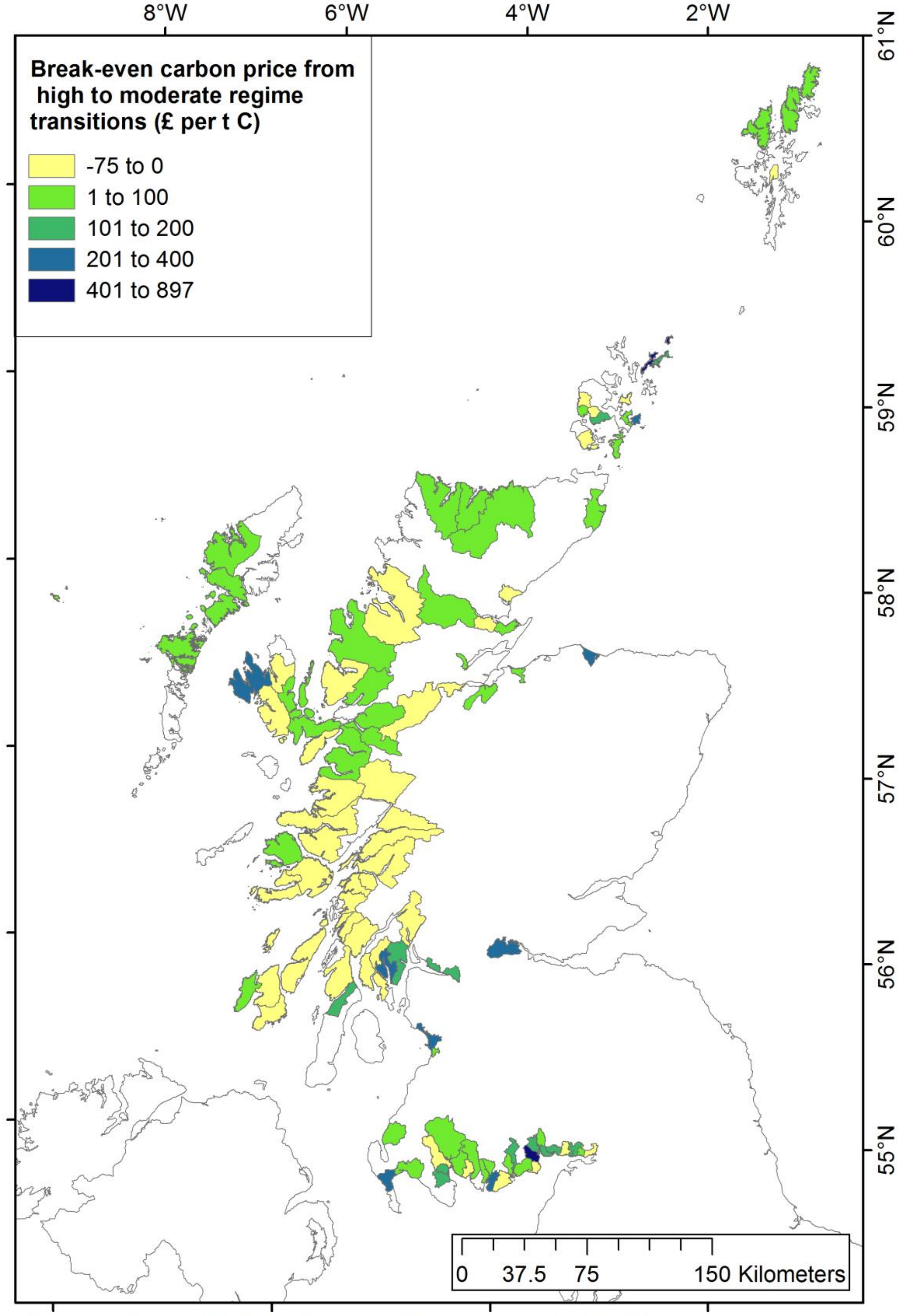


Figure S1. Change in benefits from high to low regime transitions| Map indicating the change in net benefit over 30 years arising from a transition from high to low grazing regime with a carbon price of £10 per t C at 5% discount rate.

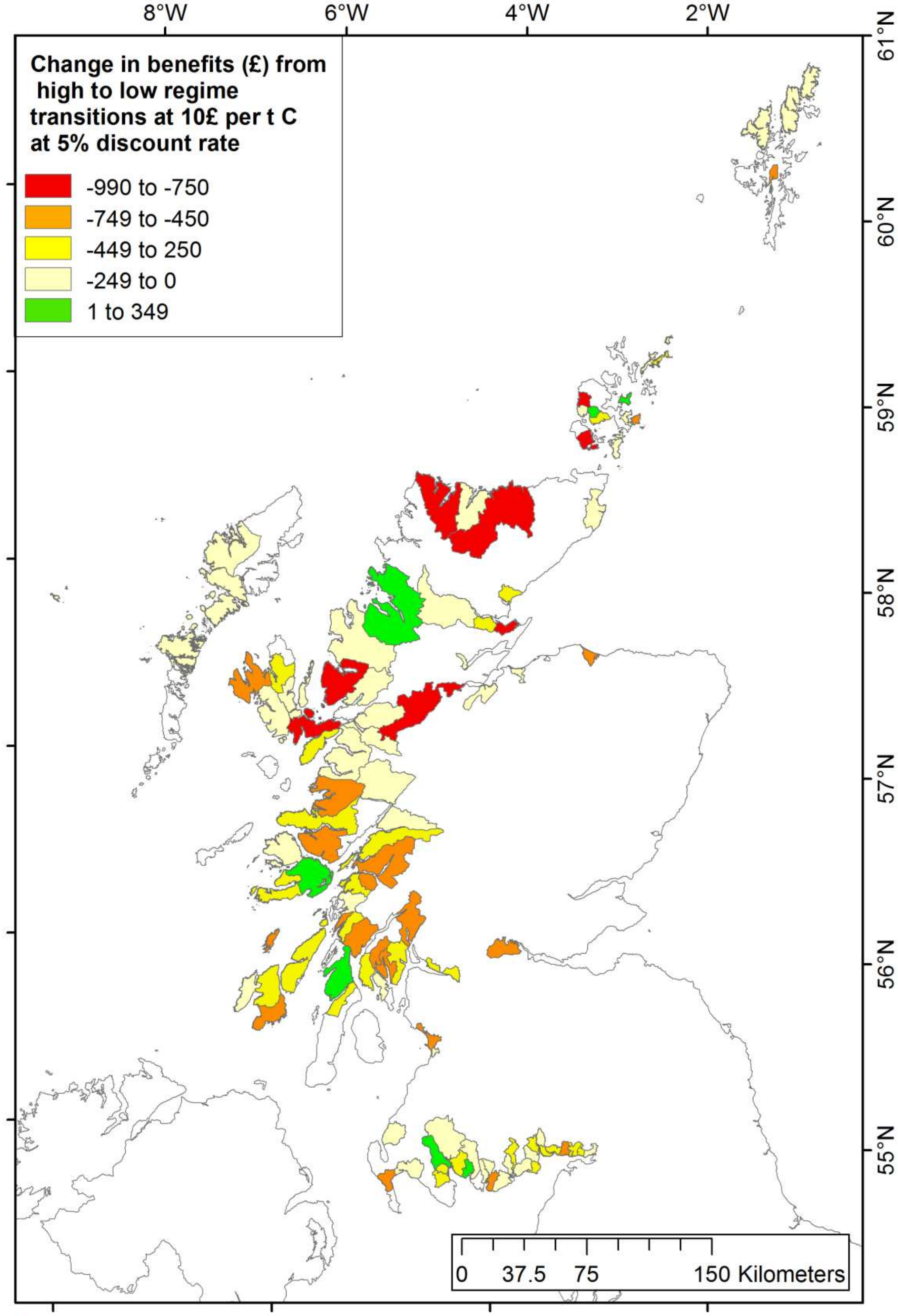
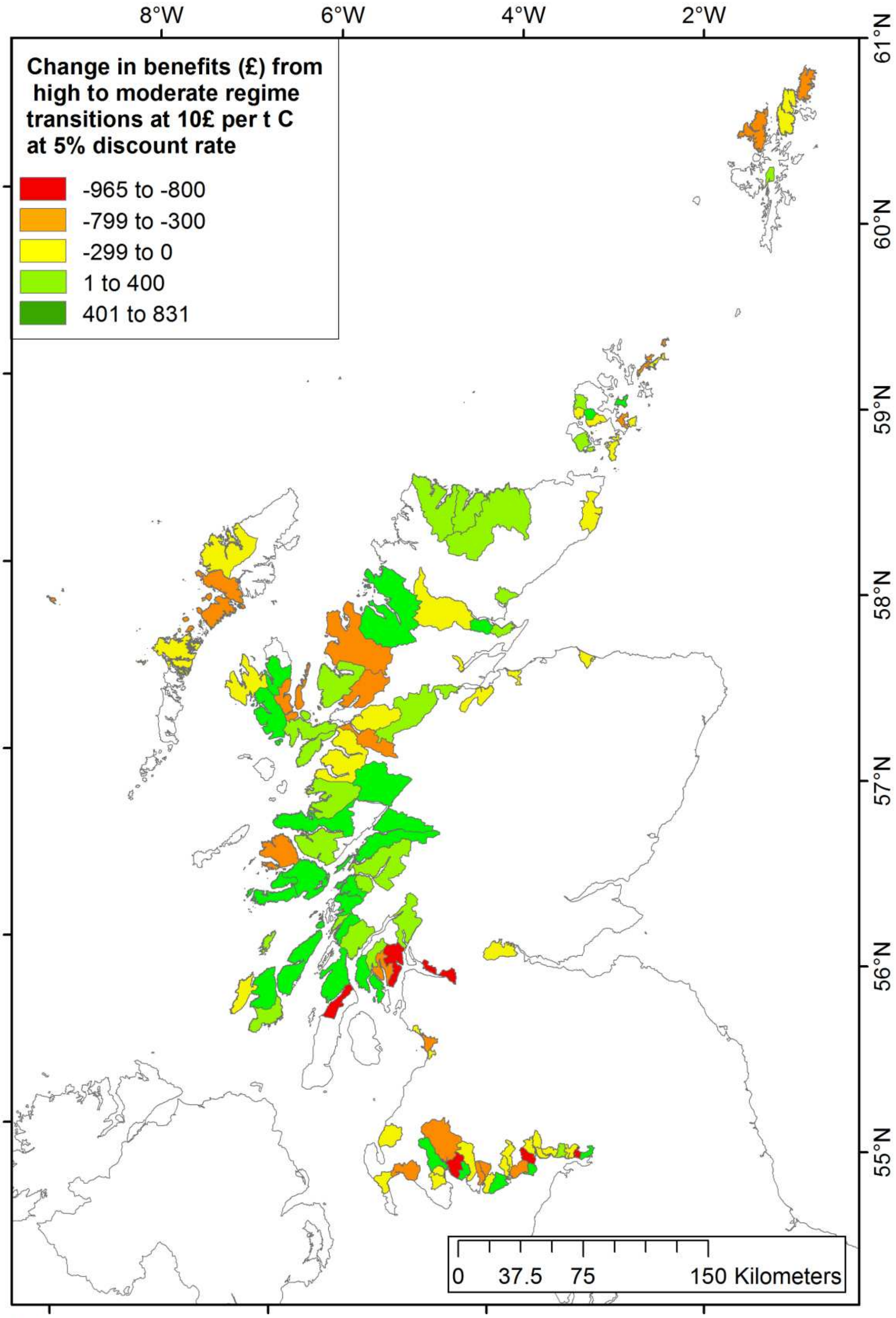


Figure S2. Change in benefits from high to moderate regime transitions | Map indicating the change in net benefit over 30 years arising from a transition from high to moderate grazing regime with a carbon price of £10 per t C at 5% discount rate.



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