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

3

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11

12 **Abstract**

13 Saltmarshes provide important services including flood control, climate regulation, and  
14 provisioning services when grazed by livestock for agriculture and conservation purposes.  
15 Grazing diminishes aboveground carbon, creating a trade-off between these two services.  
16 Furthermore, saltmarshes are threatened by overgrazing. To provide saltmarsh protection and  
17 ensure the continuing delivery of ecosystem services, there is a need to incentivise land  
18 managers to stock environmentally sensible densities. We therefore investigated the possibility  
19 of agri-environmental schemes and Payments for Ecosystem Services (PES) to compensate for  
20 lost livestock revenue under reduced grazing regimes and provide carbon sequestration and  
21 other benefits. This is the first study to consider the benefits arising from a potential carbon  
22 market to saltmarshes, although similar schemes exist for peatland and woodland. We  
23 calculated the net economic benefit (costs of livestock production are removed from revenue)  
24 to farmers obtained from a hectare of grazed saltmarsh under low (0.3 Livestock Units per  
25 hectare per year), moderate (0.6), high (1.0) and very high (2.0) stocking densities accounting  
26 for livestock revenue, carbon benefits, and agri-environmental subsidies. We repeated the

27 procedure considering additional benefits transferred from the literature in terms of  
28 provisioning, regulating and cultural ecosystem services provided by protected saltmarshes.  
29 The net benefits were assessed for a range of market carbon prices and social costs of carbon,  
30 e.g. the opportunity cost of carbon for society. Applying the model to Scottish saltmarshes we  
31 find that the current range of market prices could prompt transitions from high to moderate  
32 regimes in areas where livestock value is low, however break-even prices for transitions showed  
33 high spatial variability due to spatial variability in livestock values. In some areas of the West  
34 Highlands, the break-even carbon price is negative, indicating that the current agri-  
35 environmental schemes are able to more than compensate for the lost revenue accruing to  
36 farmers by a reduced grazing density. However, in other areas, such as the Outer Hebrides, the  
37 break-even carbon price is positive. Private PES schemes or increased public subsidies should  
38 then be provided to generate net benefits. It is reasonable to infer that a pure carbon market may  
39 have limited scope in incentivising consumers to buy carbon services, especially in areas with  
40 limited local number of buyers and corporates of small size. Under this circumstance, a  
41 premium carbon market offering bundled ecosystem services may help reduce grazing pressure  
42 across a larger number of Scottish saltmarshes, thereby providing globally important climate  
43 regulation services and at the same time protecting sensitive habitats.

#### 44 **Keywords**

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

#### 47 **Highlights**

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

## 53 **1. Introduction**

54           Saltmarshes are coastal habitats dominated by terrestrial plants that experience regular  
55 tidal inundation. They provide a wide range of ecosystem services including flood defence and  
56 wave attenuation (Pethick, 2002); water filtration and pollutant retention (Barbier et al., 2011);  
57 nutrient cycling (Burden et al., 2013); reservoirs for rare and specialist species (Jones et al.,  
58 2011); nursery for fish (Laffaille et al., 2000); habitat for grazing livestock (Olsen et al., 2011);  
59 and blue carbon sequestration (Beaumont et al., 2014). Despite their importance, considerable  
60 human activity has involved land reclamation for agriculture and urban development (Burden  
61 et al., 2013). As a result, saltmarsh cover in the UK has declined by 15% since 1945 (Beaumont  
62 et al., 2014) and quality has deteriorated, thereby compromising the capacity of saltmarshes to  
63 adapt to future sea level rise and climate change (Jones et al., 2011).

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

73           PES schemes currently existing in the UK fall under the category of carbon  
74 sequestration under the Woodland Carbon Code for newly planted woodland (Forestry  
75 Commission, 2017) and carbon sequestration with bundled co-benefits under the Peatland Code  
76 for restored peatland (IUCN, 2015; Reed et al., 2013). These Codes provide the verification and  
77 accreditation rules replacing voluntary standards such as the Verified Carbon Standard that are  
78 often characterised by high transaction costs (Bonn et al., 2014). Different forms of markets for

79 peatlands are reported by Bonn et al. (2014) who classify different schemes according to the  
80 services paid, whether they are publically or privately funded, and whether they are  
81 international or regional in scope. For example, the UK Peatland Code is a market based on  
82 corporate social responsibility (Bonn et al., 2014), covering restoration costs of peatland due to  
83 business potential investment. Other transactions are based on public money arising from agri-  
84 environmental schemes on carbon emission reduction (Reed et al., 2014) justified on the basis  
85 of paying for the fullest possible range of ecosystem service benefits, although they are poorly  
86 quantified (Reed et al., 2014). The above mentioned PES applied to peatlands fall in the  
87 category of voluntary transactions (private for carbon markets, and government mediated for  
88 agri-environmental schemes) between providers and users of resource-use proxies rather than  
89 ecosystem services (which are difficult to measure) and have the properties to show  
90 conditionality on specific agreed rules amongst parties (Wunder, 2015).

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

101         Compared with other vegetated habitats, saltmarshes have a great carbon capture and  
102 storage capacity as a result of high primary productivity and tidal trapping of organic matter  
103 (Chmura et al., 2003). Saltmarshes in Scotland sequester an estimated 2.35-8.04 t CO<sub>2</sub> ha<sup>-1</sup> yr<sup>-1</sup>  
104 and have a total carbon stock of 569.7 t ha<sup>-1</sup> (Beaumont et al., 2014). Unlike freshwater wetlands

105 which are often significant sources of methane, methane emissions from saltmarsh are  
106 negligible (Bridgham et al., 2006). Regular inundation promotes anoxic soils that delay  
107 decomposition, such that the carbon captured in saltmarsh soils is a long term sink (Yu and  
108 Chmura, 2010) well suited for targeting of a carbon emissions reduction market scheme.

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

118         The relationship between carbon sequestration and grazing on the other hand is quite  
119 complex, resulting from an interaction of stocking density, grazer type, saltmarsh zone,  
120 seasonality, and other abiotic parameters. Results from the few existing studies have been  
121 contradictory, perhaps exposing geographic differences or the overriding importance of abiotic  
122 factors. Grazed saltmarsh in Canada (Yu and Chmura, 2010) and the Netherlands (Elschot et  
123 al., 2015) had higher soil carbon contents than ungrazed sites while grazed saltmarsh in  
124 Denmark had lower soil organic matter content than ungrazed sites (Morris and Jensen, 1998).  
125 A large-scale study in England and Wales found grazing to reduce aboveground vegetation, but  
126 to provide relatively small effects on belowground carbon stores (Kingham, 2013). In addition,  
127 Kingham (2013) found that effects of grazing on above and belowground carbon stores were  
128 generally quite small compared to effects of abiotic factors. This is likely because changes in  
129 aboveground carbon stores take time to translate to changes in belowground carbon stores, and

130 mechanisms such as plant compensatory growth do not occur instantly in response to a stressor  
131 (Holland et al., 1996).

132         In this study we investigate the use of livestock grazing as a management tool for  
133 saltmarsh climate regulation services, a concept familiar to terrestrial systems (Bhogal et al.,  
134 2011; Ma et al., 2006), and the extent to which the monetary valuation of this service can be  
135 translated into a financial tool (PES). In contrast to traditional PES schemes implemented in  
136 woodland and peatland ecosystems which rely on habitat restoration and creation, the study  
137 scenario only involves changing land use. A PES scheme for carbon benefits would operate by  
138 compensating farmers for the revenue lost due to reducing stocking density. Providing an  
139 economic incentive to farmers who manage saltmarshes to reduce grazing pressure would not  
140 only help combat climate change, but ensure that the managers themselves have a vested  
141 interest in the protection of the saltmarsh, thus limiting the need for extensive regulation.  
142 Nonetheless, a relevant effort in monitoring would be required to show that a saltmarsh  
143 managed under a PES project is providing additional benefits compared to the baseline, and  
144 fast, reliable and economic tools (such as simple spreadsheet) to do it must be proposed.  
145 Grazing management is already widely used as a conservation tool (Bouchard et al., 2003) and  
146 is easily adapted to address saltmarsh carbon sequestration, as light levels of grazing have the  
147 dual benefit of enhancing biodiversity and minimising the loss of aboveground carbon (Adnitt  
148 et al., 2007; Yu and Chmura, 2010).

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



## 181 **2.2 General methodological approach**

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

## 199 **2.3 Methods for calculation of benefits from saltmarshes**

### 200 **2.3.1 Livestock benefits**

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

206 statistical purposes and for the payment of farming grants and subsidies (“Agricultural Parishes,”  
207 2017).

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

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

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<sup>1</sup> 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.

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

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

### 243 **2.3.2 Carbon benefits**

244 To calculate carbon benefits, the quantification of how changing grazing pressure  
245 affects saltmarsh carbon stocks is needed. Kingham (2013) found that aboveground plant  
246 biomass decreased by 2.1 t ha<sup>-1</sup> LU<sup>-1</sup> across high, mid, low and pioneer saltmarsh in north Wales  
247 and north-west England. Plant biomass is converted to carbon given that plants almost always  
248 contain between 45-50% carbon (Magnussen and Reed, 2004). Multiplying by an average of  
249 47.5%, this gives a reduction of aboveground carbon content due to grazing of 0.998 t C ha<sup>-1</sup>  
250 LU<sup>-1</sup>. We use data from England and Wales (not having more specific data from Scotland) to  
251 estimate these effects due to a lack of equivalent studies in Scotland, even though we expect  
252 spatial variability.

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

260 Market prices of carbon in the EU Emissions Trading System have been in the range of  
261 £3-£25 t<sup>-1</sup> C since 2007 and below £10 for the past five years (Marcu et al., 2016). Rather than  
262 testing a single market price, we perform a sensitivity analysis using four prices (£1 t<sup>-1</sup> C, £2 t<sup>-1</sup>  
263 C, £5 t<sup>-1</sup> C, and £10 t<sup>-1</sup> C) each at four discount rates (3%, 5%, 10%, and 15%). While market  
264 prices are paid to ecosystem service providers in a carbon emissions trading scheme, an  
265 alternative way of valuing carbon benefits is to consider the social cost of carbon (SCC). SCC  
266 refers to the economic damages incurred by emitting one tonne of carbon and is a cost taken on  
267 by society. If caps ensured efficient carbon removal levels, then market price and SCC would  
268 be the same (Nordhaus, 2017). A range of social costs at respective discount rates indicative of  
269 the current literature are considered (£8.04 t<sup>-1</sup> C at 5%, £23.38 t<sup>-1</sup> C at 3%, £37.26 t<sup>-1</sup> C at 2.5%,  
270 and £65.76 t<sup>-1</sup> C at 3%) (Nordhaus, 2017; Tol, 2009; van den Bergh and Botzen, 2014).

### 271 **2.3.3 Agri-environmental subsidy**

272 The UK Agri-Environment Climate Scheme compensates farmers providing  
273 environmental public goods under Pillar II of the EU Common Agricultural Policy (Reed et al.,  
274 2014). In particular, the Wetland Management Option rewards farmers who manage  
275 saltmarshes to benefit faunal and floral biodiversity with £90.03 ha<sup>-1</sup> year<sup>-1</sup> (“Wetland  
276 Management,” 2017). Although the scheme does not currently consider the carbon

277 sequestration service of saltmarshes, reducing grazing pressure to benefit biodiversity  
278 incidentally ensures that more aboveground carbon is maintained (Kingham, 2013).

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

#### 287 **2.3.4 Valuation of other saltmarsh ecosystem services**

288 Saltmarshes provide additional services on top of the provisioning service of livestock  
289 grazing and the regulating service of carbon sequestration. Using a series of choice experiments,  
290 Christie and Rayment (2012) determined that the public (sample from England and Wales) is  
291 willing to pay £450 ha<sup>-1</sup> year<sup>-1</sup> to secure services currently delivered by SSSI conservation  
292 activities in coastal grazing marsh in England and Wales. These services include nature's gifts<sup>2</sup>,  
293 research and education, climate regulation, water regulation, sense of experience, charismatic  
294 species, and non-charismatic species. Excluding climate regulation (to avoid double counting)  
295 and water regulation (the degree of which depends on the size of the site) this corresponds to  
296 £375.13 ha<sup>-1</sup> to secure the remaining services. This value is corrected for Scotland applying a  
297 willingness to pay transfer with adjustment for income elasticity as given in Pearce et al. (2006):

$$298 \quad WTP_{Scotland} = WTP_{England+Wales} (Y_{Scotland}/Y_{England+Wales})^e \quad (1)$$

---

<sup>2</sup> 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.

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

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

### 312 **2.3.5 Calculating benefit and break-even carbon prices**

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

$$314 \quad NB(t) = B_L(t) - B_C(t) + B_S(t) + B_V(t) \quad (2)$$

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

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

324 
$$NPV = \sum_{t=0}^T \frac{NB(t)}{(1+i)^t} \quad (3)$$

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

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

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

### 342 **3. Results**

#### 343 **3.1 Carbon benefits**

344 Carbon benefits over 30 years for various market carbon prices and discount rate for 1  
345 LU ha<sup>-1</sup> yr<sup>-1</sup> are shown in Table 1. These represent the monetary value of (i) carbon emission  
346 avoided and (ii) the methane equivalent carbon emission avoided by livestock if grazing  
347 intensity is reduced by 1 LU ha<sup>-1</sup> yr<sup>-1</sup>. It is possible to estimate relative values as arising from  
348 changes in grazing regimes from the difference in benefits when transitioning from one regime

349 to another. For example, the carbon benefits lost under a very high regime ( $2 \text{ LU ha}^{-1} \text{ yr}^{-1}$ ) are  
350 the benefits in Table 1 multiplied by two, as the grazing intensity is twice as high. Thus, a  
351 transition from a very high to high grazing regime will result in carbon benefits equivalent to  
352 the values presented in Table 1. Insofar as these values represent the amount of money  
353 generated if a carbon market provided payments for reduced emissions due to reduction from  
354 very high to high grazing regime, these carbon benefits are quite low.

#### 355 TABLE 1

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

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

#### 367 TABLE 2

### 368 **3.2 Change in net benefits when altering grazing regimes**

369 Table 3 illustrates how reductions in grazing regimes result in either loss or gain of net  
370 economic benefit from a hectare of saltmarsh over a period of 30 years. For a majority of cases  
371 where sheep and cattle grazing intensity is reduced from a higher regime to a lower regime,  
372 land managers can expect to lose money under the current range of market carbon prices (Table  
373 3). This is because the value of carbon benefits lost at higher stocking density is much lower

374 than the revenue gained from livestock. The greatest losses are incurred under a transition from  
375 very high to low regimes, as this is when the greatest change in stocking density occurs at a  
376 reduction of 1.7 LU ha<sup>-1</sup> yr<sup>-1</sup>. The reduction in net benefit is lower with higher carbon prices, as  
377 these equate to higher carbon benefits to offset revenue loss.

378 TABLE 3

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

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

395 TABLE 4

396 **3.3 Livestock opportunity costs**

397 Examining the transition from high to moderate stocking density further, as this is the  
398 most favourable transition having the lowest opportunity cost and greatest increase in net

399 benefit (Tables 3 and 4), the loss in livestock revenue that is incurred each year is given in Table  
400 5. The figures represent the amount of money farmers would lose for reducing stocking density  
401 by 0.4 LU ha<sup>-1</sup> yr<sup>-1</sup>. Conversely, this is also the amount of compensation needed to incentivise  
402 a reduction in grazing pressure, whether provided through ecosystem service benefits or  
403 subsidiary aid.

#### 404 TABLE 5

### 405 3.4 Break-even carbon prices

406 Break-even carbon prices were overall higher for sheep than for cattle (Table 6), as the  
407 former are more valuable when considered per livestock unit. Negative break-even price  
408 occurred under a transition from high to moderate sheep and cattle stocking densities, as sites  
409 then qualified for the additional £90.03 ha<sup>-1</sup> compensation scheme which drove down the  
410 required carbon benefit to offset loss of agricultural revenue. The fact that these values are  
411 negative indicates that land managers are better off under the less intense regimes. In other  
412 cases, the positive break-even prices indicate the market price that carbon would have to be at  
413 to incentivise managers to reduce stocking density. The range of positive carbon prices in Table  
414 6 are higher than the range of market prices we explored, but comparable to the range of social  
415 costs of carbon indicative of the literature. Therefore, if carbon removal levels were efficient  
416 and offset occurred at the full social cost, these would be sufficient to prompt a decrease in  
417 grazing pressure. The highest break-even prices occurred under a transition from moderate to  
418 low and very high to high stocking densities (Table 6). The prices of these two transitions are  
419 identical because both involve halving of stocking density and no addition or removal of  
420 subsidy.

#### 421 TABLE 6

### 422 **3.5 Geographic differences in opportunity costs**

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

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

#### 433 FIGURE 2

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

#### 439 FIGURE 3

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

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<sup>3</sup> Differences in values are given by the different composition of livestock across the regions

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

449 FIGURE 4

450 FIGURE 5

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

#### 459 **4. Discussion**

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

#### 463 **4.1 Viability of findings for a pure carbon market in mitigating saltmarsh grazing** 464 **pressure**

465 We set out to investigate whether a pure carbon market could prompt a reduction of  
466 grazing pressure on Scottish saltmarshes by calculating net benefits of land use and break-even  
467 carbon prices to compensate revenue loss. The break-even prices were directly related to  
468 opportunity costs, such that prices were lower for cattle grazing regimes than for more valuable  
469 sheep grazing regimes, and had both negative and positive values. Negative break-even prices  
470 were calculated under certain scenarios, particularly the most favourable high to moderate

471 regime change, due to gains from subsidies being greater than the relatively small reduction in  
472 stocking density. This illustrates the role that agri-environment subsidies have in promoting less  
473 intense grazing management practice, and that there is no need for further compensation  
474 schemes when livestock value is low.

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

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

490         In our analysis of livestock values based on real parish compositions (Figures 4-5), the  
491 lowest break-even carbon price was around £-77 t<sup>-1</sup> C. However, parishes displayed a huge  
492 variability in break-even prices, such that even in the most favourable transition from high to  
493 moderate some parishes had break-even prices around £960 t<sup>-1</sup> C. These results illustrate that a  
494 subset of parishes could well have pure carbon markets with price of carbon in international  
495 market ranges while still managing to incentivize reductions in grazing pressure, while in other  
496 parishes this is largely unfeasible due to high value of livestock. There is some promise in that

497 lower break-even prices are found on the west coast and offshore islands (Figure 2), areas that  
498 are identified by Haynes (2016) as containing saltmarsh sites which lack designation and are  
499 threatened by overgrazing. These parishes then, having lower opportunity costs, will more  
500 easily elicit decreases in grazing intensity through the introduction of a PES scheme selling  
501 carbon emission reduction and other services. In the majority of parishes however, it appears  
502 rather unfeasible that a pure carbon market can provide carbon benefits at a high enough value  
503 to incentivise saltmarsh agriculturists to decrease grazing intensity. Moreover, the number of  
504 potential carbon credits that can be exchanged in this local market would be very limited and  
505 insufficient to compensate corporates emissions.

#### 506 **4.2 Towards a voluntary carbon PES scheme**

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

518 Notably, perceptions on the importance of saltmarsh ecosystem services are highly influenced  
519 by people's perspective and level of education, and whether services are directly experienced  
520 and affect local livelihoods (Soy-massoni et al., 2016). This requires a specific analysis of the  
521 demand of the services more appreciated by stakeholders under different management regimes,

522 as occurred for peatlands (Bonn et al, 2014). The valuation of services provided by a saltmarsh  
523 site is therefore highly influenced by local perceptions and conditions. When we considered the  
524 benefit transfer of services provided by SSSI sites, we simplified this by adapting the perceived  
525 social benefit from English and Welsh saltmarshes from Christie and Rayment (2012). To have  
526 a more accurate picture, however, the next step would be to assess social perception and  
527 willingness to pay for services in our study sites from different potential buyers, focusing  
528 possibly on broad domains with similar typology of farming rather than attempting to assess  
529 values for each parish individually.

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

537 Once beneficiaries and the main services demanded are identified, it is possible to figure  
538 out a specific market for saltmarshes services. Here we focus on voluntary carbon market.  
539 Compared to the compliance carbon markets developed under the Kyoto protocol, voluntary  
540 markets have more opportunities to apply to local land use projects and higher flexibility to  
541 bundle benefits. Moreover, schemes under the UN framework such as Joint Implementation,  
542 Clean Development Mechanisms and Reduced Emissions from Deforestation and Degradation  
543 work directly with and through national government processes and usually require a threshold  
544 of thousands of carbon credits, difficult to reach for coastal project (Wylie et al., 2016).  
545 Conversely, international voluntary standards for blue carbon projects are usually based on  
546 tailored methodologies (usually validated by Plan Vivo and Verified Carbon Standard -VCS)  
547 and prove to be easier to implement due to the higher flexibility and lower costs of the required

548 carbon accounting, verification and certification. VCS has so far provided general standards for  
549 land-based climate change mitigation projects. Moreover, VCS VM0024 provides GHGs  
550 assessment for coastal wetland creation, while the VM 0033 (2015) for tidal wetland restoration.  
551 Recently, adaptations to VCS standard have been proposed by Reed et al. (2013) to verify  
552 changes in GHGs fluxes resulting from peatland restoration. However, verification and  
553 accreditation remain financially feasible only for large-scale projects and big investors (Wylie  
554 et al., 2016). To overcome these difficulties, regional voluntary agreements are emerging; they  
555 may be national or more localised markets targeting local/national investors under new or  
556 adapted existing standards to tailor specifically to local restoration schemes (Kosoy and Guigon,  
557 2012). Examples for the UK are the peatland and woodland restoration projects (Forestry  
558 Commission, 2017; IUCN, 2015). An advantage of these markets is that the standards proposed  
559 are often based on VCS guidance, but use their own empirical or modelled measurement to  
560 verify carbon emission reduction: they are rigorous for local investors, but more cost effective  
561 than the VCS standard. Finally, they often target more services, not having the possibility to  
562 reach a critical threshold for only one (for example high volume of carbon credits sold).

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

573 Bundling carbon sequestration with other services requires identifying whether low  
574 levels of grazing conflict with other supporting, regulating, and cultural services. As already  
575 mentioned, there is good evidence that low levels of saltmarsh grazing benefits biodiversity  
576 (Adnitt et al., 2007; Jones et al., 2011). There is some evidence that nutrient cycling in coastal  
577 grasslands is maximised under ungrazed regimes (Ford et al., 2012) and that grazing slows  
578 down microbial turnover (Olsen et al., 2011), although classic theory dictates that intensive  
579 grazing promotes faster nutrient cycling than light or ungrazed regimes (Bardgett et al., 1998).  
580 Flood control is diminished with high levels of grazing as large herbivores such as cattle cause  
581 soil compaction and decrease soil infiltration rates (Carroll et al., 2004). This suggests that  
582 perhaps low levels of small herbivore grazing such as sheep may have smaller or negligible  
583 negative effects. Grazing effects on cultural services depend somewhat on the cultural aspect  
584 being valued. For example, the cultural service of environmental appreciation of forb and  
585 flower abundance have been found to be maximised under extensively grazed regimes (Ford,  
586 2012), but extensive grazing is known to decrease abundance of wading birds (Chapman, 2007).  
587 There is the need, therefore, before valuing bundled ecosystem services, for further research to  
588 identify the services that trade-off and those that complement when aiming for a lessening of  
589 grazing intensity.

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

598 finally incentives might induce the argument that polluters must be paid for conservation  
599 reducing intrinsic or altruistic motivations (Chan et al., 2017).

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

621 PES must account for risks to permanence (point ii), and in particular natural risks  
622 caused by climate change and different patterns of erosion that might reduce the area dedicate

623 to grazing. A series of strategies must be adopted to minimise these risks, from the evaluation  
624 of exposure and vulnerability of saltmarshes to external stressors, to their mitigations provided  
625 by a larger scale of participants; from the formulation of landscape-scale PES schemes to the  
626 promotion of financial instruments such as insurance or PES credit buffers to replace or make  
627 alternative payments for a failed project (Friess et al., 2015). Other forms of risks are internal  
628 to the project management such as lack of sufficient funds to meet the opportunity costs of  
629 reduced grazing or reversal to higher intensive use in habitats outside the area of intervention.  
630 This is one of the seven concerns raised by Chan et al (2017) upon a PES scheme, where any  
631 new markets or system of incentives creates new externalities. However, there is building  
632 evidence that the latter can be addressed by promoting the roles of farmers as stewards of the  
633 countryside, and rewarding patterns of behaviour (Newtown, 2011) rather than environmental  
634 outcomes. This could facilitate the attestation of long-term commitments to reduced grazing  
635 effort that subsidies paying the full economic costs cannot address, contrarily to what is assumed  
636 by mainstream economics (Goldman-Benner et al., 2012; Wunder, 2013; Ferraro and Kiss,  
637 2002). An alternative externality to a PES applied to saltmarsh grazing is the relocation of  
638 grazing from natural pasture to stable production, removing the benefits of methane emissions  
639 reduction from reducing ruminant. The additionality of this service, at least internally to the  
640 country, can be guaranteed under a national cap system (quotas) that reduces the total number  
641 of herds. However, it seems impossible to guarantee additionality if the reduction of grazing in  
642 the UK is compensated by an equivalent increase in another country.

643 Finally, to guarantee cost-effective monitoring of GHGs emissions reduction (point iii),  
644 it is essential to develop a protocol that facilitates indirect valuation of GHGs to reduce the  
645 costs of direct measurements or process-based computational models (Bonn et al., 2014),  
646 expected to be relevant in those contexts (such as saltmarsh habitats) where carbon emissions  
647 show high variability over space and time. Proxy variables can be specifically developed for  
648 GHGs emissions and tailored to the regional context as proposed for the valuation of carbon

649 emissions change of aquatic restored habitats<sup>4</sup>. For our specific case, it is possible to propose  
650 an empirical model fitting data from UK studies on management of saltmarshes,  
651 biogeochemical parameters (water table level, soil moisture, soil temperature, etc.), and  
652 ecological gradients (species composition).

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

658         A final consideration is about the importance of valuation to establish a PES. Although  
659 valuation is not always needed to create a market (Vatn, 2010), a potential cause of PES failure  
660 could be due to the difficulties associated with the agreement of prices of the services to be  
661 exchanged. A sound valuation of ecosystem services, under deliberated approach (Spash, 2008;  
662 Kenter et al., 2017) or reverse auction (Stoneham et al., 2003), could facilitate agreement  
663 between the parties, reducing transaction costs, and providing fair prices (usually lower than  
664 those paid in a fixed-price system) that reflect cost of management in a context of intrinsic  
665 motivation (Chan et al., 2017). Valuation is not relevant also when trading is not the mechanism  
666 chosen to carry out a transaction of ecosystem services (Wunder, 2015). This is true for the  
667 Peatland Code (Reed et al., 2013) and the PES scheme proposed in this paper, where private  
668 financial transactions are supposed to be made by corporates for enhancing social responsibility  
669 rather than trading carbon credits for offsetting CO<sub>2</sub> emissions. Moreover, in the case of  
670 saltmarshes, where the ecosystem provides a wide range of services that can vary across space  
671 and time, it is often difficult and time- and resource-consuming to provide accurate valuations

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

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

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

### 689 **4.3 Assumptions and limitations**

690         It is important to highlight that the results given here are highly dependent on a series  
691 of assumptions introduced to simplify complex real-world grazing practices into a format that  
692 could apply more generally to typical saltmarshes. The reason why a transition from high to  
693 moderate and low stocking density was feasible is because we assumed sites stocked at a density  
694  $< 0.6 \text{ LU ha}^{-1} \text{ yr}^{-1}$  can obtain financial support from the Wetland Management agri-environment  
695 option. It may be argued that such an assumption is precarious, especially since grazer type,  
696 local conditions and seasonal variation determine whether a particular grazing intensity is

697 harmful for biodiversity at a particular site. We could not obtain the average stocking density  
698 of sites currently managed under this option due to an absence of data. However, even with this  
699 assumption the underlying logic for the trends we observe and discuss holds true. Essentially,  
700 if a decrease in grazing pressure is sufficient to allow farmers to claim subsidies, this drives  
701 down the required carbon price to a value that might guarantee private voluntary payments able  
702 to compensate forgone income and guarantee a bundle of benefits. The specific value of this  
703 carbon price will depend on local conditions (opportunity cost in each parish) and on the degree  
704 to which grazing intensity is reduced.

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

708 We assume the full ecosystem services of SSSI protected sites are provided by sites  
709 grazed under low grazing regimes, even if these sites are not currently under SSSI designation.  
710 We did this to obtain an approximation of the social benefit that saltmarsh sites provide, but  
711 future work will have to investigate how the value of services differs between protected and  
712 non-protected sites. We also assume that the value of livestock grazed on saltmarsh is  
713 equivalent to national average livestock values. Moreover, it is important to keep in mind that  
714 the enforcement of any stocking density on a saltmarsh site can be more effective in either  
715 privately owned land or saltmarsh with clear fenced designations. PES is not intended as a silver  
716 bullet that can address any environmental problem. Where it is missing the authority to manage  
717 ecosystems, a prior condition to the success of PES is to enforce property rights (Engel et al.,  
718 2008). Sites where any agriculturist can freely graze their livestock cannot receive PES unless  
719 there is a record of total grazing pressure over the year.

720 There are some limitations to the grazing effect on carbon sequestration we adapted  
721 from a study of English and Welsh saltmarshes (Kingham, 2013). The reduction in aboveground  
722 vegetation due to grazing (for further detail see section 2.2) was a single effect without a

723 confidence interval or probability distribution, preventing us from using a statistical approach.  
724 Saltmarshes in the UK are classified into four bio-geographical regions with distinct vegetation  
725 communities: western, south-eastern, western Scottish and eastern Scottish (Adam, 1990). The  
726 effect of grazing on Scottish saltmarsh may therefore differ slightly from the effects observed  
727 in the English and Welsh region. As there is currently no study which has explored this for  
728 Scotland specifically, there is a need for further research in this regard. Such a study must  
729 consider effects on both aboveground and belowground stores, considering soil carbon in  
730 coastal ecosystems is a very important component of blue carbon accounting (Pendleton et al.,  
731 2012). The subsequent model of grazing effects should then be validated with on the ground  
732 collection of primary data. Further socio-economic research, for example via use of  
733 questionnaires, could be used to validate the assumptions proposed by the model in the regions  
734 that show promise in incorporating alternative financing schemes.

735

## 736 **5. Conclusions**

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

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

749 Post and Lundin, 1996). As a part of this toolset, Payments for Ecosystem Services provide an  
750 incentive-based mechanism promoting nature conservation (Lau, 2013). Blue carbon initiatives  
751 are becoming common under the Kyoto Protocol Clean Development Mechanism, whereby  
752 climate mitigation projects are carried out in developing countries (Palmer and Silber, 2012).  
753 Blue carbon restoration projects of mangrove, seagrass and saltmarsh habitats in countries such  
754 as Kenya, Vietnam, Madagascar and India help alleviate poverty, finance conservation, and  
755 ensure the continuing provision of climate mitigation services (Hejnowicz et al., 2015; Lau,  
756 2013; Wylie et al., 2016). While our study focused on saltmarsh grazing in Scotland, the  
757 management of saltmarsh grazers for carbon sequestration purposes could be investigated in  
758 other countries using the same approach, both developing and developed, for blue carbon  
759 initiatives.

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

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

777 Under the scenario of grazing reduction from 1 to 0.6 LU ha<sup>-1</sup> we expect to have a  
778 reduction of 9,260 heads of which 51% cattle (4,759 units) and 49% sheep (4,501 units). These  
779 reduced cattle units correspond only to 1.3% of the total production in the 174 parishes where  
780 saltmarshes are present, and to 0.26% of the entire Scottish production. Moreover, the reduction  
781 in sheep is 0.35% of the production in the 174 parishes where saltmarshes are present and  
782 0.067% of the global Scottish production. Although it is not aim of this research to quantify the  
783 economic impact on the agricultural sector, considering the very small reduction in livestock  
784 units, it is unlikely that these figures will have a relevant impact on trade balance, negative  
785 effect on food security and distortion on prices. Conversely, the rewards by public subsidies  
786 and private incentives can help sustain income and reduce pressure on a sensitive habitat  
787 favouring the production of environmental benefits and contributing to provide a new role to  
788 farmers as local stewards of nature rather than mere producers of market commodities.

789 In its early stages, the PES saltmarsh market is unlikely to function as a formal carbon  
790 offsetting tool. Instead, it allows third parties to demonstrate corporate and public social  
791 responsibility and to benefit from services which could lessen costs of, for example, water  
792 filtration or sea defences. As saltmarsh carbon projects become more common, they may more  
793 easily gain verification and accreditation, to name one example with the Verified Carbon  
794 Standard (VCS, 2017), allowing the transfer of carbon emission reduction credits in an  
795 emissions trading scheme. The Woodland Code started in a similar fashion and now allows  
796 projects to generate tradable carbon, while the Peatland Code is on its way to this stage (IUCN,  
797 2015). In this way, although a pure carbon market appears unfeasible, a premium carbon market  
798 offering bundled ecosystem services may help reduce grazing pressure on Scottish saltmarshes,  
799 thereby providing globally important climate regulation services and at the same time protecting  
800 sensitive habitats.

801

802 **7. Acknowledgements**

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804 fulfilment for the degree of MSc Ecosystem-Based Management of Marine Systems,  
805 University of St Andrews (August 2017).

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

809

810 **Table 1 Carbon benefits lost under a range of market prices** | Value of carbon emission  
 811 avoided from 2015-2045 from 1 ha of saltmarsh when grazing is reduced by 1 LU ha<sup>-1</sup> (high  
 812 grazing regime) using a range of market carbon prices. Benefits for low, moderate, and very  
 813 high regimes are calculated by multiplying values reported in Table 1 by respective stocking  
 814 densities.  
 815

Discount rate (%)	Carbon price (£ t <sup>-1</sup> 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

816

817

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

824

Discount rate (%)	Carbon price (£ t <sup>-1</sup> 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

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826

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

828 Change in net benefits (in GBP) over 30 years caused by a change in sheep and cattle grazing

829 regimes. Positive values in bold indicate gains in net benefit.

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

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

830

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832

833 **Table 4 Change in net social benefit of grazing regimes** | Change in net benefits (in GBP)

834 over 30 years when sheep and cattle grazing regimes are changed using social costs of carbon.

835 For each livestock type, scenarios without additional service valuation and with SSSI valuation

836 from Christie and Rayment (2012) are presented.

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

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

843

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

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849 **Table 6 Break-even carbon prices to offset opportunity costs** | Required prices of carbon  
 850 (£ t<sup>-1</sup> C) to offset the opportunity cost of having reduced livestock. At these carbon prices  
 851 farmers would not lose any money changing from one grazing regime to the next. At any  
 852 higher prices farmers would gain money by reducing grazing pressure.

853

	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

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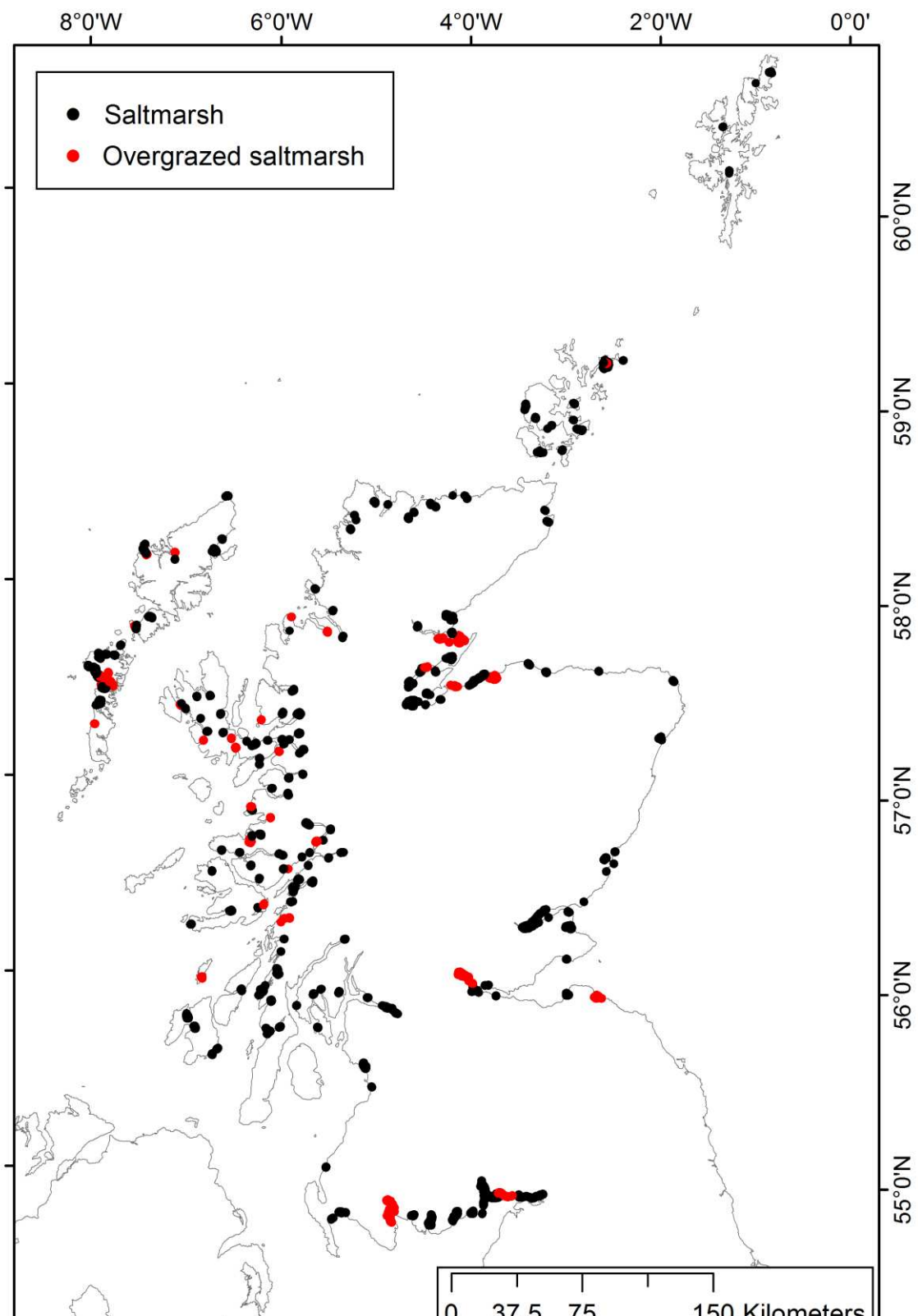
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864 **Figure 1. Scottish saltmarsh distribution** | Map of 249 saltmarsh sites and 174 parishes  
865 containing saltmarsh across Scotland. West coast sites are smaller but more numerous than east  
866 coast sites. Sites marked red indicate overgrazed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

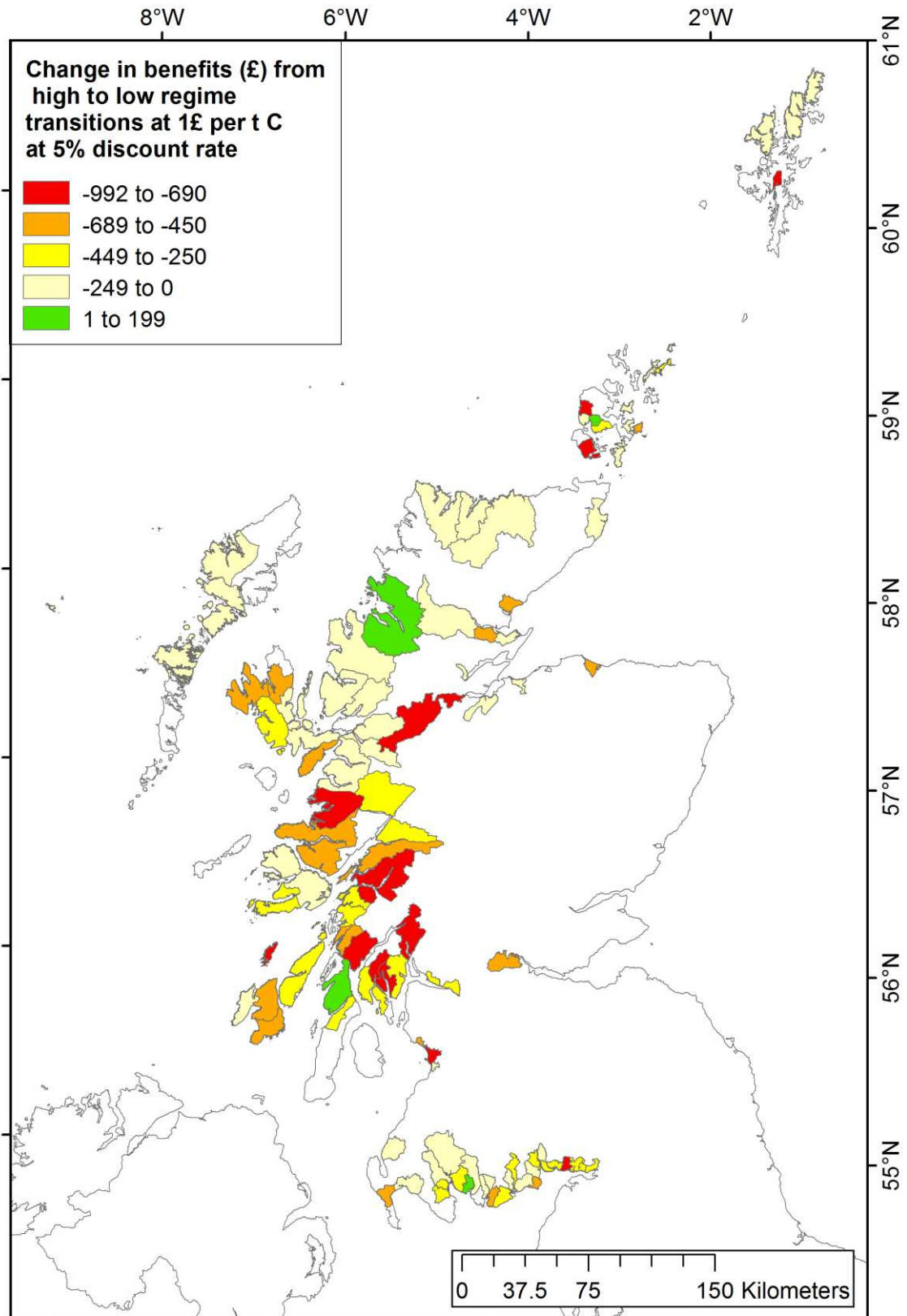
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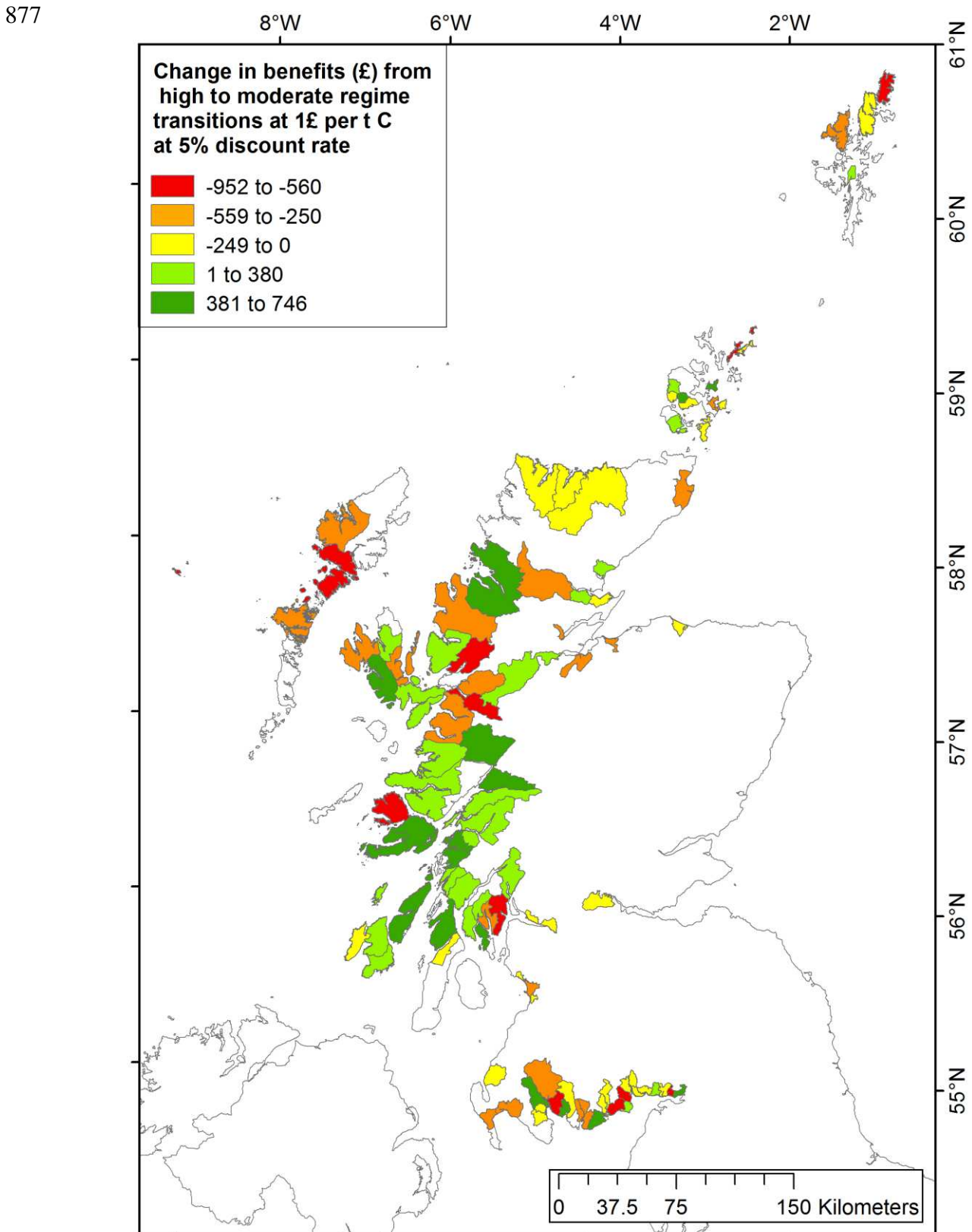


869 **Figure 2. Change in benefits from high to low regime transitions** | Map indicating the  
870 change in net benefit over 30 years arising from a transition from high to low grazing regime  
871 with a carbon price of £1 per t C at 5% discount rate.

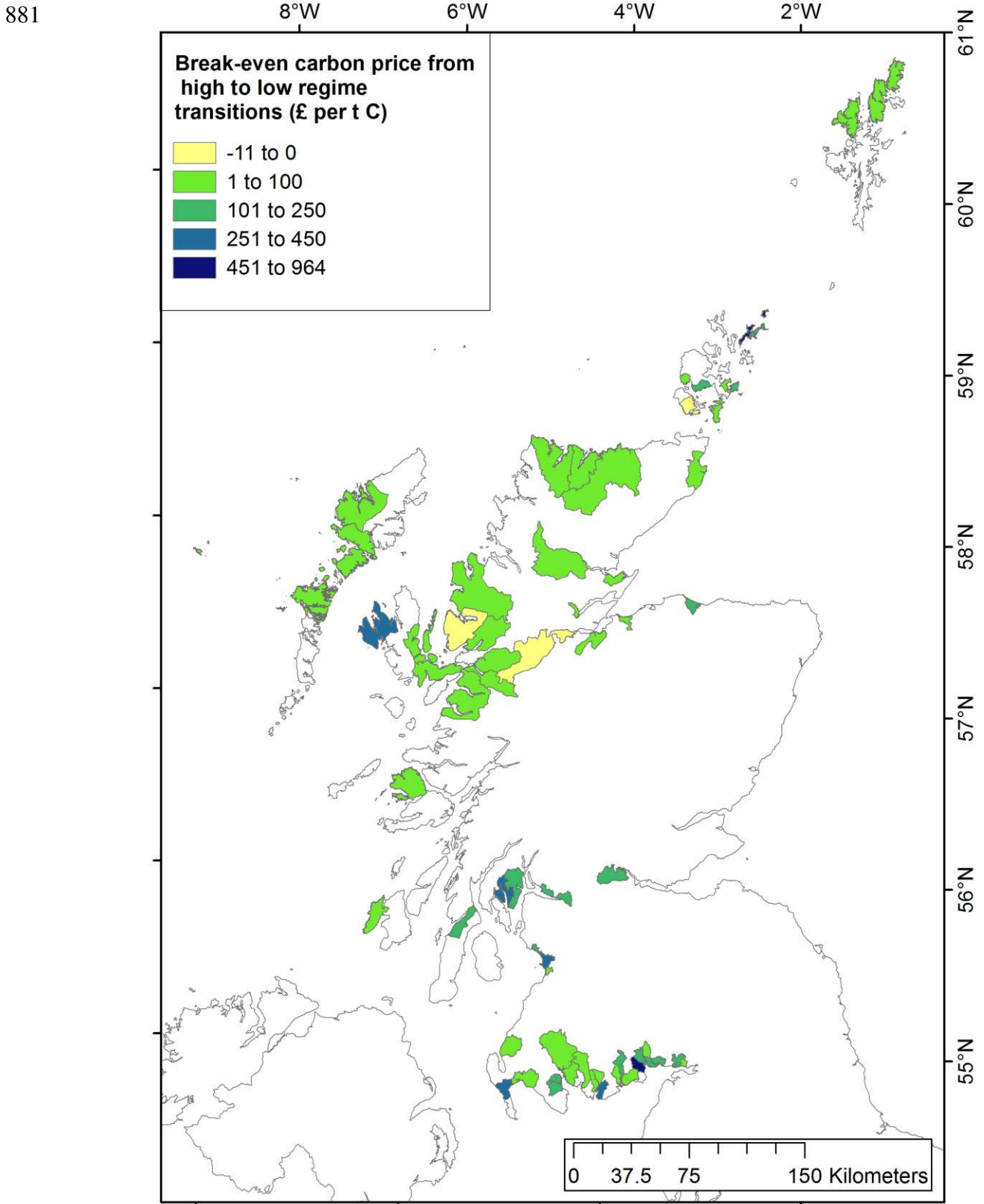
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874 **Figure 3. Change in benefits from high to moderate regime transitions** | Map indicating  
875 the change in net benefit over 30 years arising from a transition from high to low grazing regime  
876 with a carbon price of £1 per t C at 5% discount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

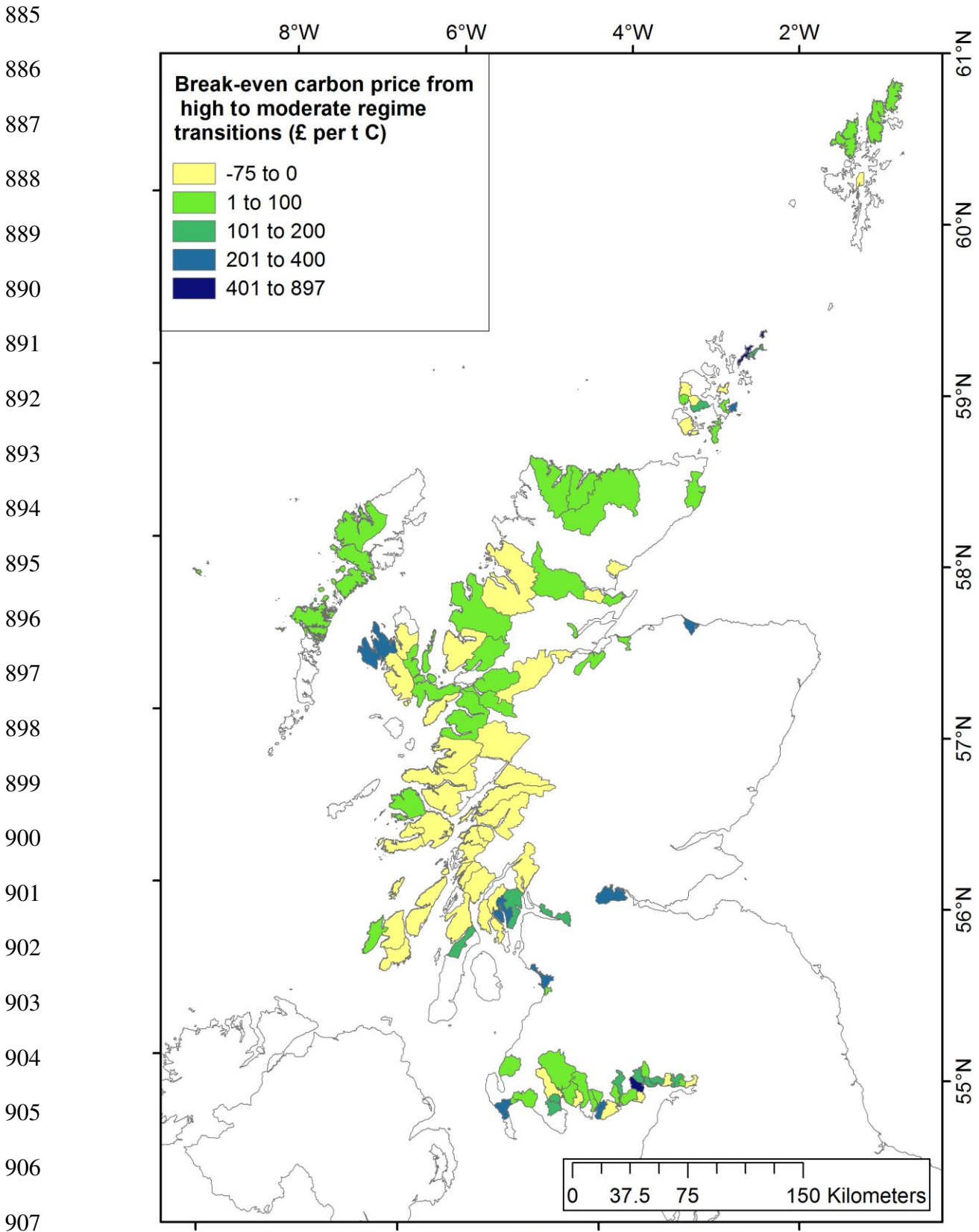


878 **Figure 4. Break-even carbon prices for high to low regime transitions** | Map indicating the  
879 required price of carbon to compensate for loss in agricultural revenue for a transition from  
880 high to low grazing regimes.



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



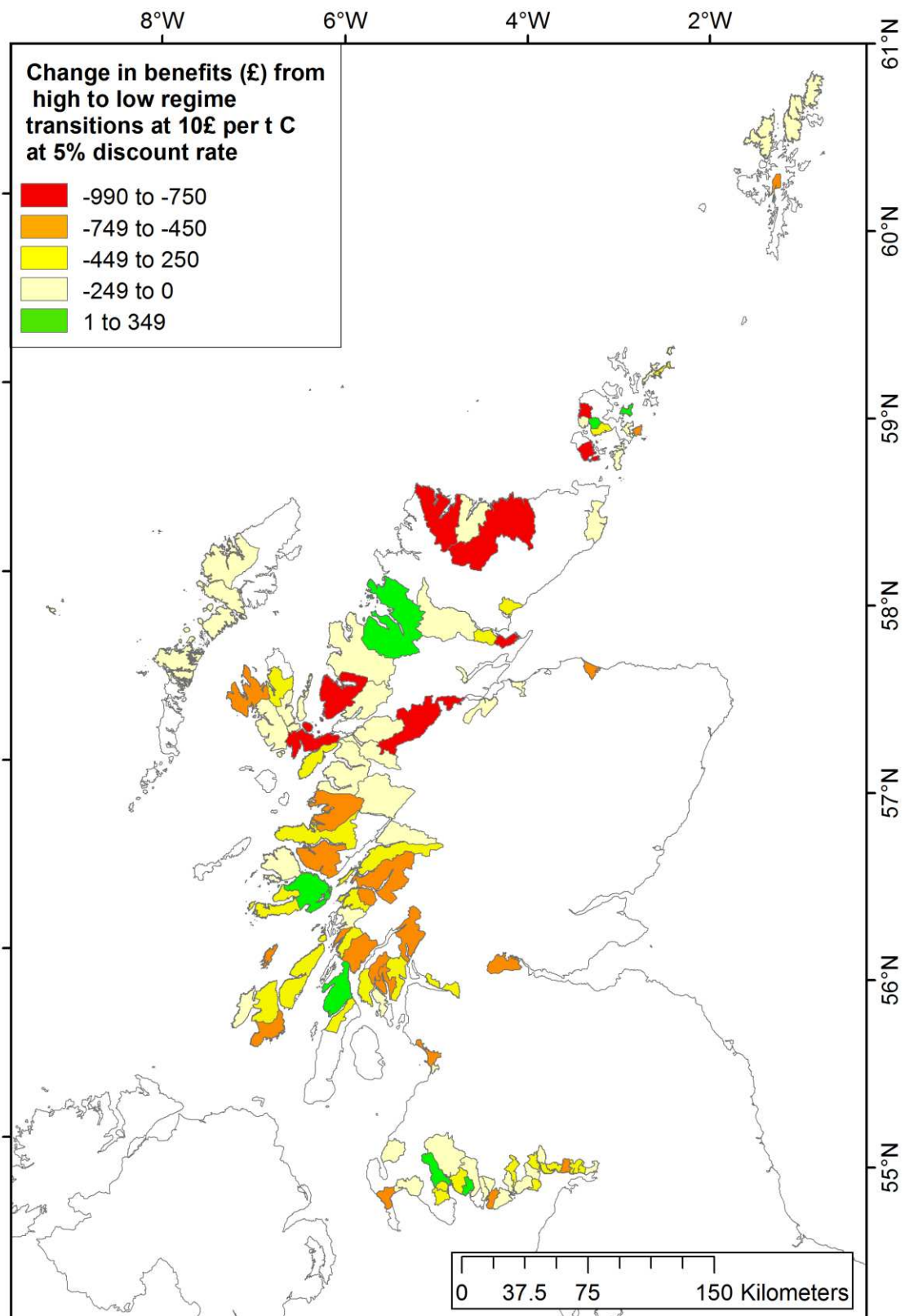
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909 **Figure S1. Change in benefits from high to low regime transitions** | Map indicating the

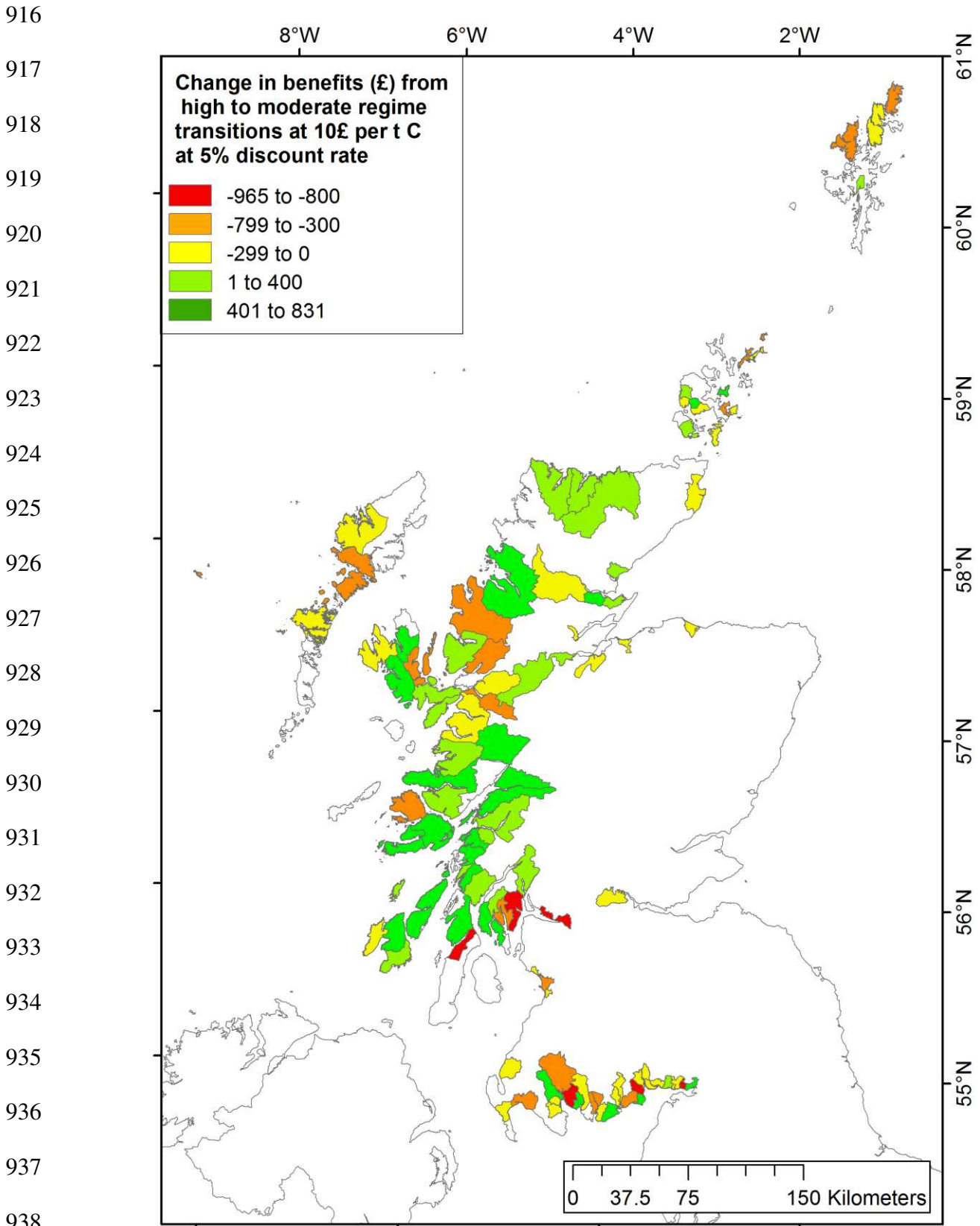
910 change in net benefit over 30 years arising from a transition from high to low grazing regime

911 with a carbon price of £10 per t C at 5% discount rate.

912



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



939

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