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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>

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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 CO₂ ha⁻¹ yr⁻¹
104 and have a total carbon stock of 569.7 t ha⁻¹ (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⁻¹) 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 used¹. 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⁻¹ 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⁻¹. Although parish SO LU⁻¹ should be in the same range as
226 national SO LU⁻¹ (Table S1), preliminary analysis revealed for several parishes unrealistically
227 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.

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⁻¹, split into exclusive cattle or sheep (Table S1) that do not provide
235 any geographic information; and (ii) the SO LU⁻¹ 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⁻¹ yr⁻¹
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⁻¹ yr⁻¹), moderate (0.6 LU ha⁻¹ yr⁻¹), high (1.0
242 LU ha⁻¹ yr⁻¹), and very high (2.0 LU ha⁻¹ yr⁻¹).

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⁻¹ LU⁻¹ 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⁻¹
250 LU⁻¹. 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⁻¹ 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⁻¹ C, £2 t⁻¹
263 C, £5 t⁻¹ C, and £10 t⁻¹ 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⁻¹ C at 5%, £23.38 t⁻¹ C at 3%, £37.26 t⁻¹ C at 2.5%,
270 and £65.76 t⁻¹ 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⁻¹ year⁻¹ (“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⁻¹ 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⁻¹ year⁻¹ to secure services currently delivered by SSSI conservation
292 activities in coastal grazing marsh in England and Wales. These services include nature's gifts²,
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⁻¹ 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):

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

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.

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⁻¹
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⁻¹) 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⁻¹ yr⁻¹ 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⁻¹ yr⁻¹. 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⁻¹ yr⁻¹. 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⁻¹ year⁻¹ 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⁻¹ yr⁻¹, 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⁻¹) 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⁻¹ yr⁻¹. 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⁻¹ 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⁻¹ in Stenness to £1,527.70 LU⁻¹ in
426 Monkton & Prestwick³ (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⁻¹ 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⁻¹ 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⁻¹ 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⁻¹ 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⁻¹ C to £963.53 t⁻¹ 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⁻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

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⁻¹ C and £102.40 t⁻¹ 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⁻¹ 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⁻¹ 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⁻¹ (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 subsidises 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 habitats⁴. 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 CO₂ 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

⁴ 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⁻¹ 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⁻¹ 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**

803 The study proposed is described in Dominic Muenzel's MSc thesis submitted as a partial
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⁻¹ (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 ⁻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

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⁻¹ (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 ⁻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

825

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 ⁻¹ 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	150
	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
<i>Cattle</i>							
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

830

831

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 ⁻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
<i>Sheep</i>							
<i>without SSSI valuation</i>							
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
<i>with SSSI valuation</i>							
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
<i>Cattle</i>							
<i>without SSSI valuation</i>							
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
<i>with SSSI valuation</i>							
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

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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⁻¹ 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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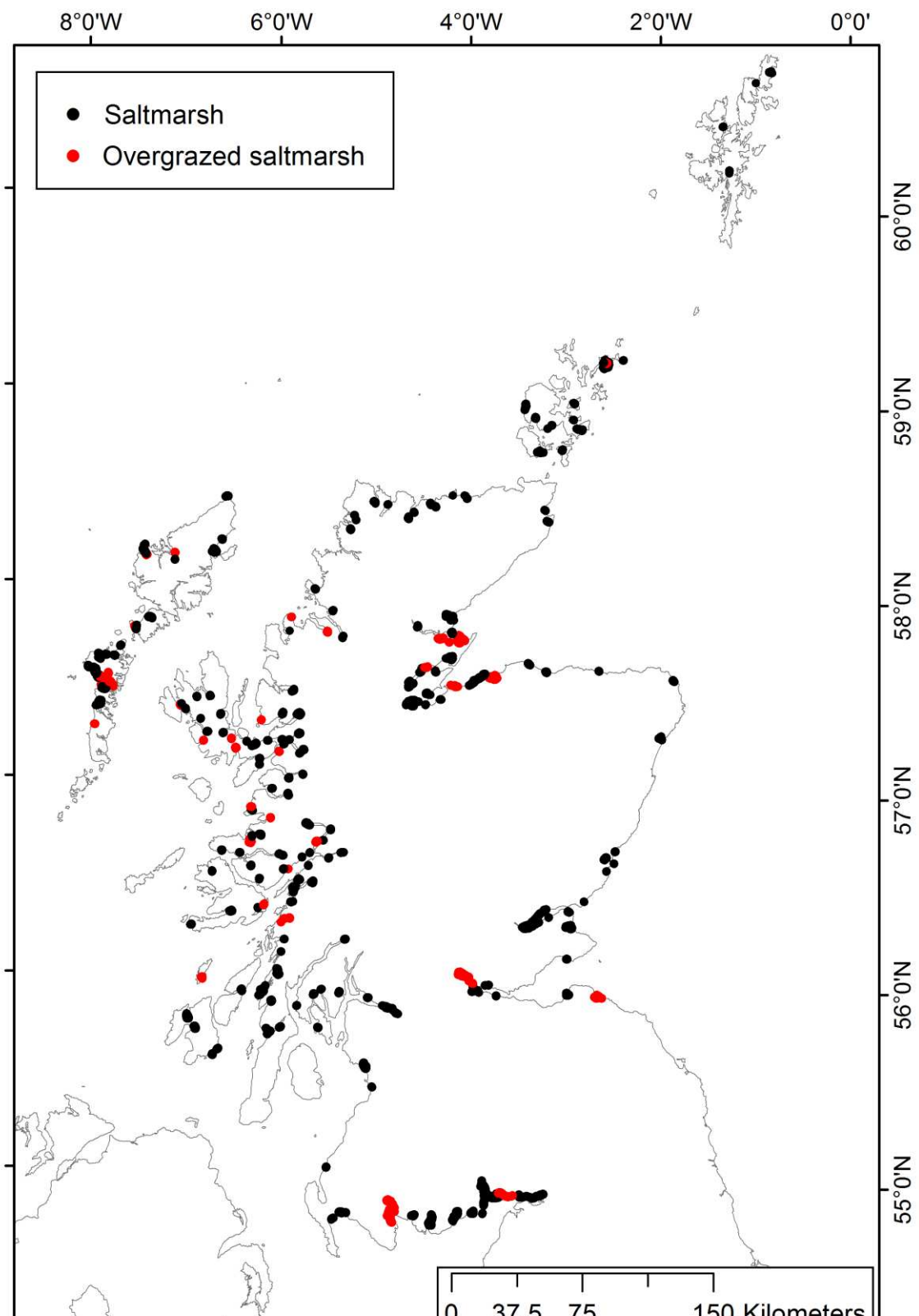
864 **Figure 1. Scottish saltmarsh distribution** | Map of 249 saltmarsh sites and 174 parishes

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).

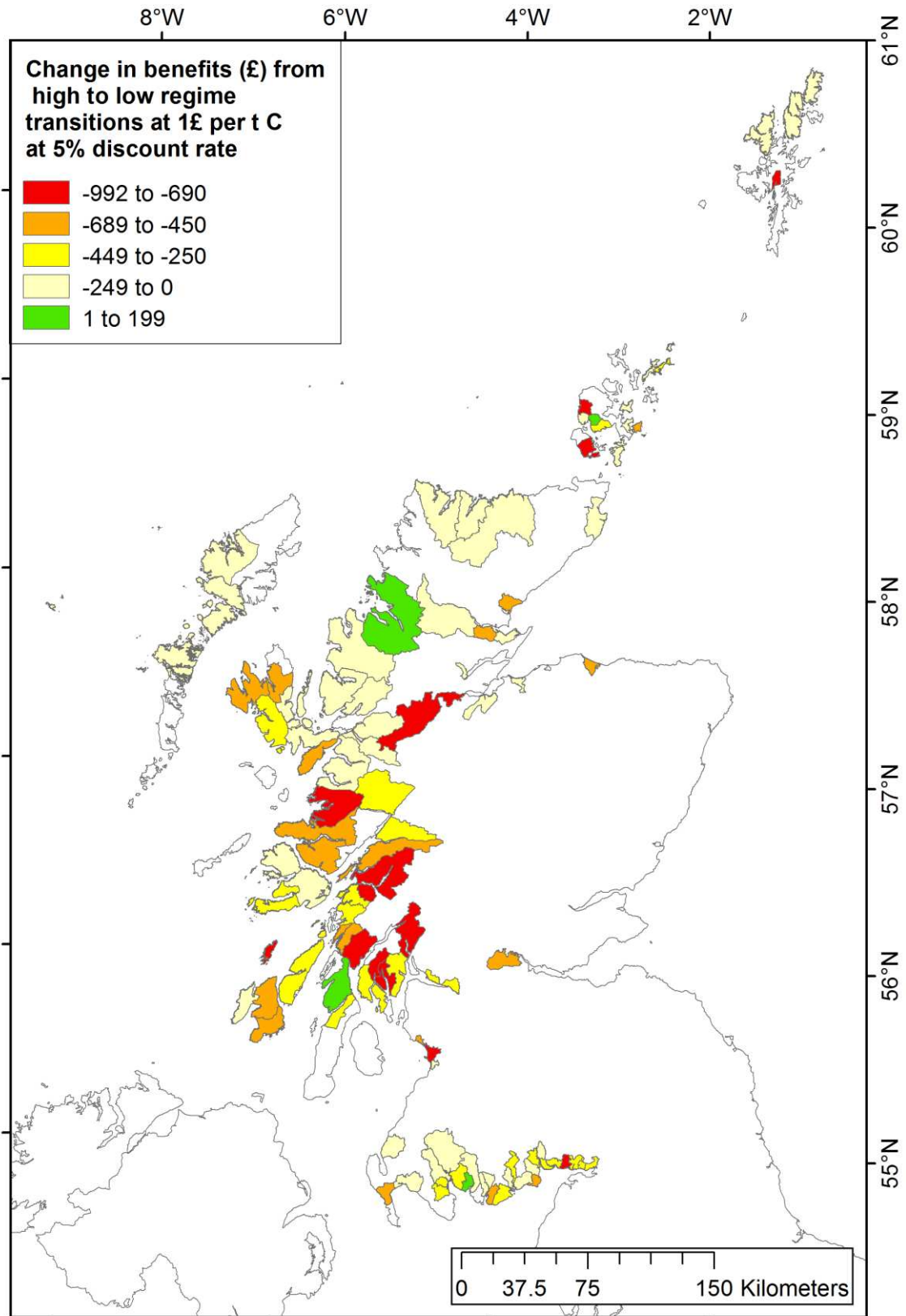
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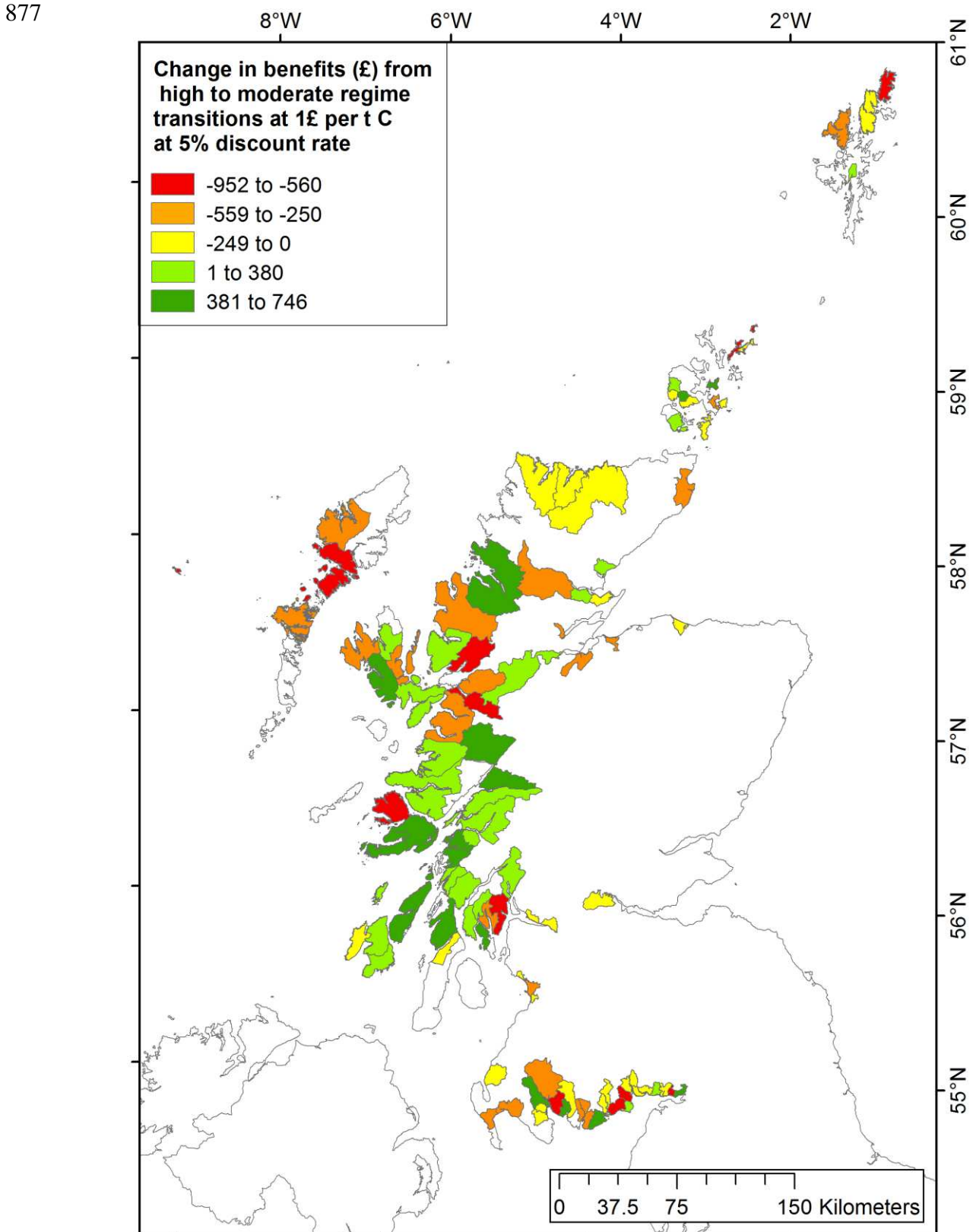


869 **Figure 2. Change in benefits from high to low regime transitions** | Map indicating the
870 change in net benefit over 30 years arising from a transition from high to low grazing regime
871 with a carbon price of £1 per t C at 5% discount rate.

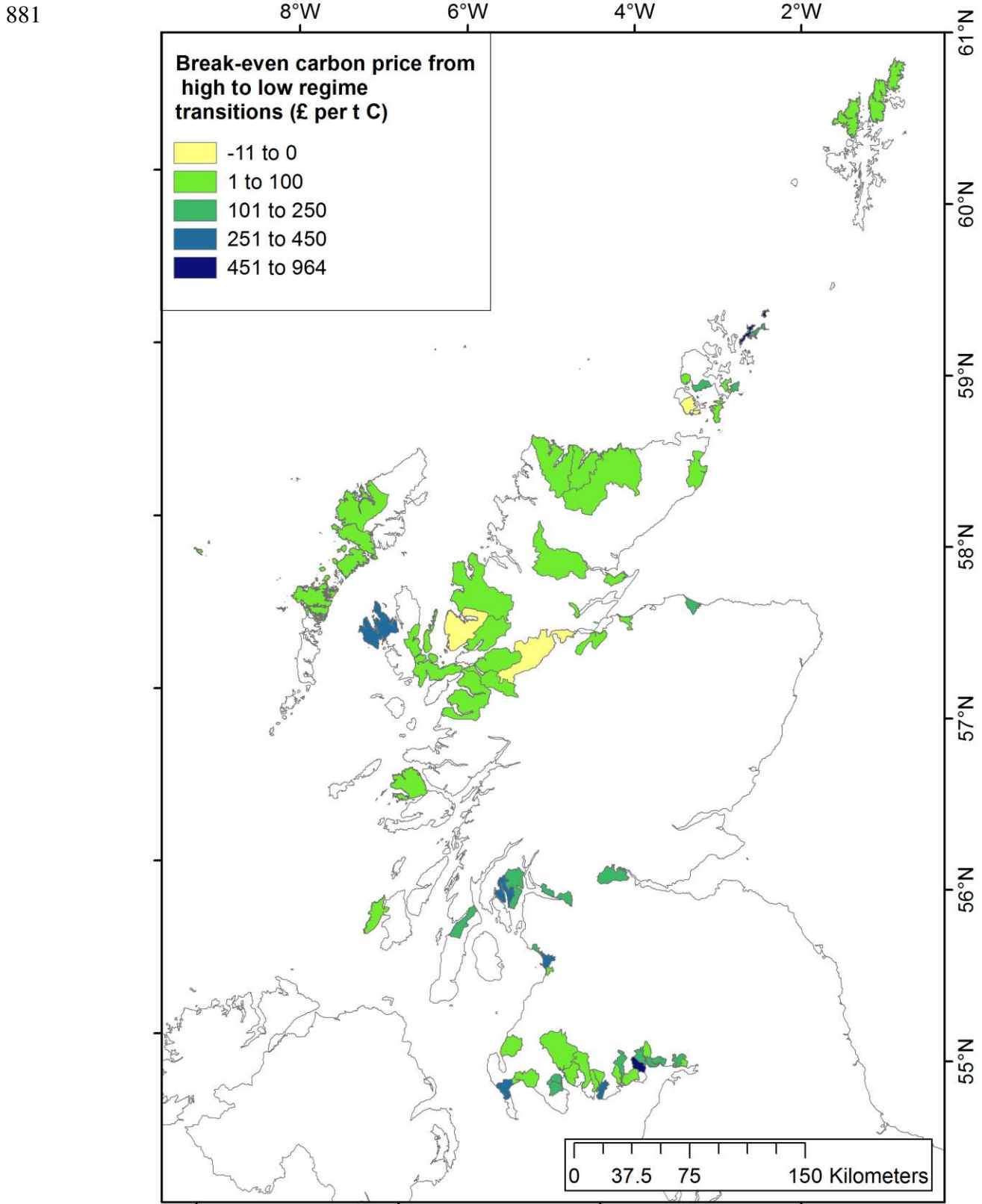
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874 **Figure 3. Change in benefits from high to moderate regime transitions** | Map indicating
875 the change in net benefit over 30 years arising from a transition from high to low grazing regime
876 with a carbon price of £1 per t C at 5% discount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

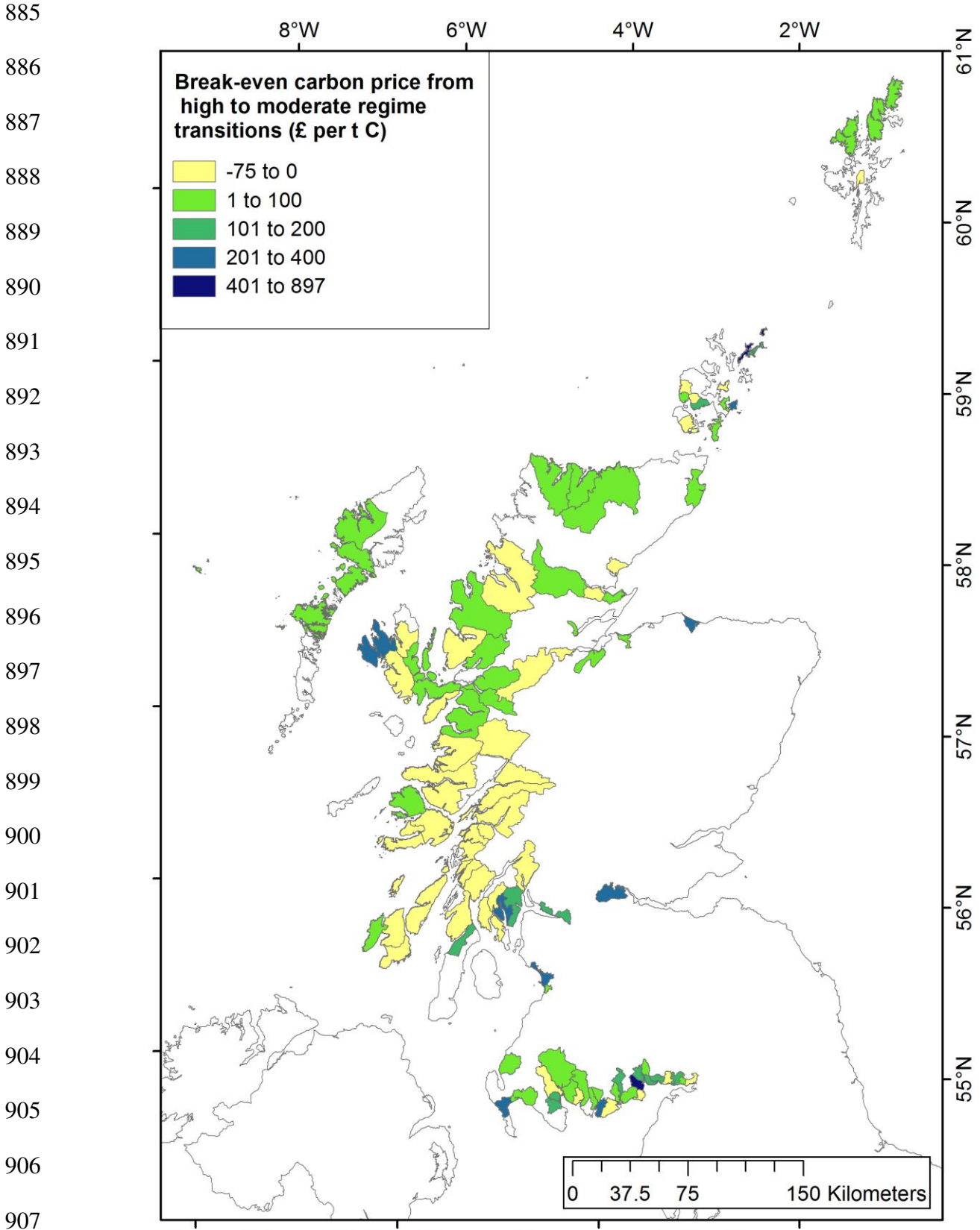


878 **Figure 4. Break-even carbon prices for high to low regime transitions** | Map indicating the
879 required price of carbon to compensate for loss in agricultural revenue for a transition from
880 high to low grazing regimes.



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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.



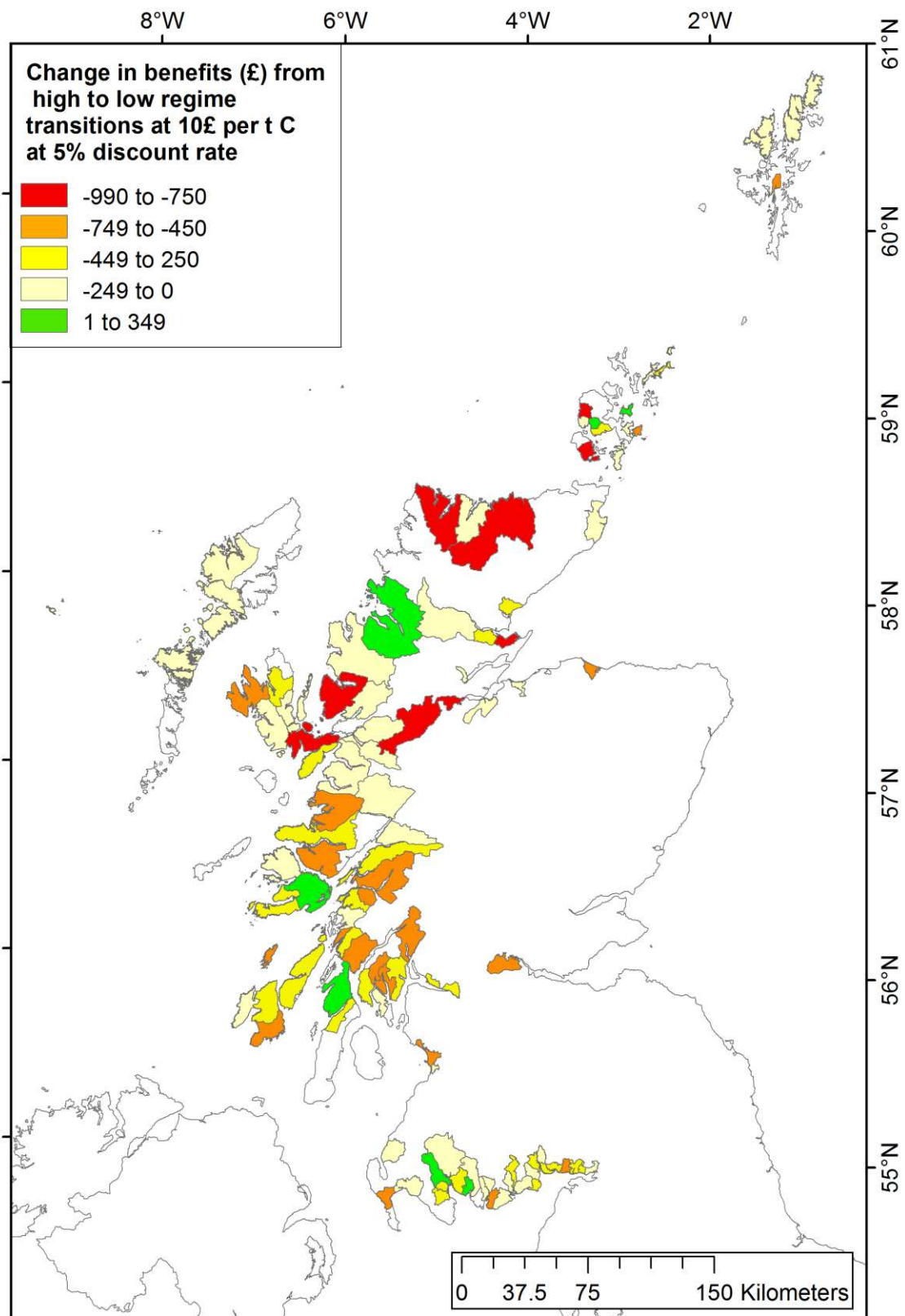
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909 **Figure S1. Change in benefits from high to low regime transitions** | Map indicating the

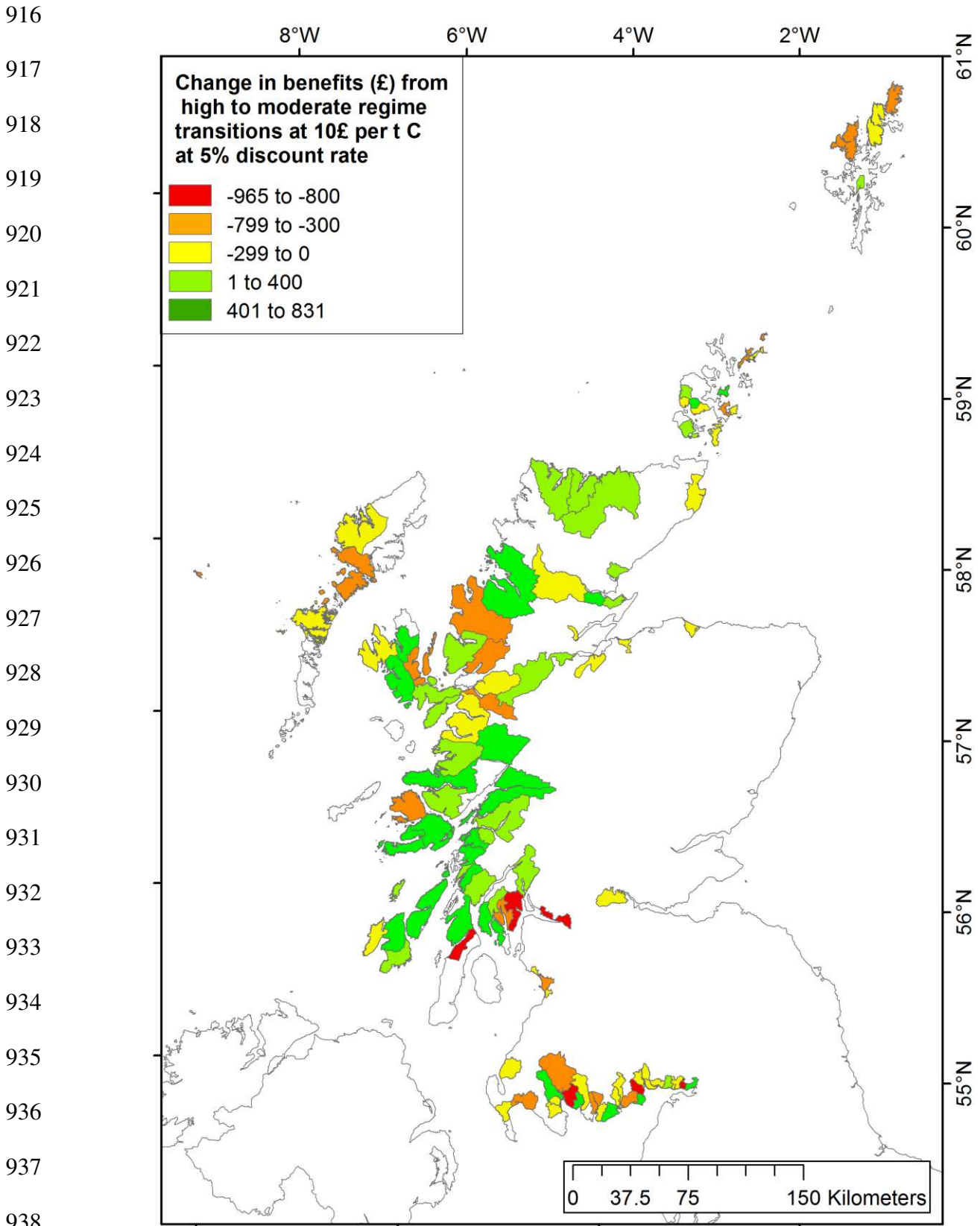
910 change in net benefit over 30 years arising from a transition from high to low grazing regime

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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