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Authors(s)	Renou-Wilson, Florence, Moser, Gerald, Fallon, D., Farrell, Catherine A., Müller, Christoph, Wilson, David
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1 **Rewetting degraded peatlands for climate and biodiversity benefits: results from two**
2 **raised bogs**

3

4 F. Renou-Wilson¹, G. Moser², D. Fallon³, C. A. Farrell³, C. Müller^{1,2} and D. Wilson^{4*}

5 ¹ School of Biology and Environmental Science, University College Dublin, Ireland

6 ² Institute for Plant Ecology, Justus Liebig University Giessen, Germany

7 ³ Bord na Móna, Leabeg, Tullamore, Co. Offaly, Ireland

8 ⁴ Earthy Matters Environmental Consultants, Glenvar, Co. Donegal, Ireland

9

10 *Corresponding author. Email address david.wilson@earthymatters.ie

11

12 **Abstract**

13 Globally, peatlands are under threat from a range of land use related factors that have a
14 significant impact on the provision of ecosystem services, such as biodiversity and carbon (C)
15 sequestration/storage. In Ireland, approximately 84 % of raised bogs (a priority habitat listed
16 in Annex I of the EU Habitats Directive) have been affected by peat extraction. While
17 restoration implies the return of ecosystem services that were characteristic of the pre-
18 disturbed ecosystem, achieving this goal is often a challenge in degraded peatlands as post-
19 drainage conditions vary considerably between sites. Here, we present multi-year greenhouse
20 gas (GHG) and vegetation dynamics data from two former raised bogs in Ireland that were
21 drained and either industrially extracted (milled) or cut on the margins for domestic use and
22 subsequently rewetted (with no further management). When upscaled to the ecosystem level,
23 the rewetted nutrient poor domestic cutover peatland was a net sink of carbon dioxide (CO₂)
24 ($-49 \pm 66 \text{ g C m}^{-2} \text{ yr}^{-1}$) and a source of methane (CH₄) ($19.7 \pm 5 \text{ g C m}^{-2} \text{ yr}^{-1}$), while the
25 nutrient rich industrial cutaway was a net source of CO₂ ($0.66 \pm 168 \text{ g C m}^{-2} \text{ yr}^{-1}$) and CH₄
26 ($5.0 \pm 2.2 \text{ g C m}^{-2} \text{ yr}^{-1}$). The rewetted domestic cutover site exhibited the expected range of
27 micro-habitats and species composition found in natural (non-degraded) counterparts. In
28 contrast, despite successful rewetting, the industrially extracted peatland did not exhibit
29 typical raised bog flora. This study demonstrated that environmental and management
30 variables can influence species composition and, therefore, the regeneration of species typical
31 of natural sites, and has highlighted the climate benefits from rewetting degraded peatlands in
32 terms of reduced GHG emissions. However, rewetting of degraded peatlands is a major

33 challenge and in some cases reintroduction of bryophytes typical of natural raised bogs may
34 be more difficult than the achievement of proper GHG emission savings.

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36 **Keywords:** Climate, greenhouse gases, restoration, species diversity

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56 **1. Introduction**

57 Globally, peatlands and organic soils are under threat from a range of climatic and land use
58 related factors that have a significant impact on the provision of ecosystem services, such as
59 biodiversity (Lachance and Lavoie, 2004; Parish et al., 2008; Pasquet et al., 2015) and
60 carbon (C) sequestration and storage (Beetz et al., 2013; Evans et al., 2014; Renou-Wilson
61 et al., 2014; Tiemeyer et al., 2016).

62 In natural (i.e. not degraded) peatlands, permanently waterlogged conditions prevent
63 the complete decomposition of dead plant material, leading to the accumulation of C rich peat
64 (Page and Baird, 2016). Natural peatlands are generally net sinks for carbon dioxide (CO₂
65 uptake) and sources of methane (CH₄) emissions (Christensen et al., 2012; Roulet et al.,
66 2007) and this interplay in gas exchange over millennia has played a major role in the
67 regulation and maintenance of the global climate (Frolking et al., 2006). Furthermore,
68 natural peatlands are an important global biodiversity resource as they provide niche habitats
69 for a range of specialized fauna and flora (Chapman et al., 2003; Minayeva et al., 2017;
70 Parish et al., 2008).

71 Global emissions from land use change (e.g. deforestation, afforestation) are
72 estimated at 3.67 Gt CO₂ yr⁻¹ (Le Quéré et al., 2016), however this value does not include
73 emissions associated with the drainage of peatlands and organic soils, or from peat fires
74 (Houghton et al., 2012), which could potentially release an additional 0.9-3 Gt of CO₂ to the
75 atmosphere annually (Joosten et al., 2012; Smith et al., 2014). Furthermore, land use change
76 has also led to the loss of 15 % of global peatland habitats over the last century (Barthelmes,
77 2016).

78 In Ireland, loss of peatland habitat through conversion to grassland, forestry and from
79 peat extraction (industrial and domestic peat extraction) is estimated at 85 % of the national
80 resource (Malone and O'Connell, 2009) and the most recent national monitoring survey
81 showed that 84 % of raised bogs (a priority habitat listed in Annex I of the EU Habitats
82 Directive (EU Directive on the Conservation of Habitats, Flora and Fauna 92/43/EEC), have
83 been affected by peat extraction alone (NPWS, In press). Moreover, CO₂ emissions from
84 degraded Irish peatlands and from related peat activities (combustion of peat for energy,
85 horticulture) have been estimated at c. 11 to 12.4 Mt CO₂ each year (Barthelmes, 2016;
86 Wilson et al., 2013b), which is similar to annual emissions from the Irish transport sector
87 (Duffy et al., 2015). Critically, only 1,955 ha of raised bog in Ireland (~0.1 %) is considered

88 'active' i.e. capable of C sequestration and continued peat formation (Fernandez et al.,
89 2014).

90 Rewetting of peat soils (i.e. by ditch blocking, creation of bunds etc.) has been
91 suggested as an important climate change mitigation tool to reduce emissions, to create
92 suitable conditions for C sequestration and to stimulate biodiversity (Bonn et al., 2014;
93 Parish et al., 2008). However, a wide array of annual CO₂ (e.g. Knox et al., 2015; Renou-
94 Wilson et al., 2016; Wilson et al., 2016b; Wilson et al., 2007b; Yli-Petäys et al., 2007),
95 CH₄ (e.g. Günther et al., 2015; Strack and Zuback, 2013; Vanselow-Algan et al., 2015) and
96 N₂O (e.g. Dinsmore et al., 2009; Minke et al., 2016; Ye and Horwath, 2016) fluxes have
97 been reported for rewetted peatlands, largely driven by differences between climate zones
98 (e.g. boreal, temperate and tropical) and nutrient status (Wilson et al., 2016a). Other factors
99 such as time since rewetting, current vegetation composition and previous land use
100 management are also likely to have a significant influence on GHG dynamics post-rewetting,
101 although the extent of these effects may be highly site specific.

102 Rewetting and additional restoration measures have also the potential to re-instate the
103 original ecosystem services, such as water conservation, erosion and fire prevention; and co-
104 benefits, such as enhanced biodiversity value of peatlands (Parish et al., 2008; Tanneberger
105 and Wichtmann, 2011), which themselves would contribute to climate change adaptation. For
106 example, a specific Aichi Biodiversity Target (i.e. Target 15) to combat climate change and
107 biodiversity loss by the restoration of 15 % of degraded ecosystems by 2020 has been
108 proposed by the Convention for Biological Diversity (CBD, 2010) with rewetting and
109 restoration of peatlands put forward as a specific instance of such a win-win outcome. While
110 the biodiversity-climate change nexus is now well recognized (Parish et al., 2008), studies
111 pertaining to the restoration/rewetting of degraded peatlands still focus mostly on one single
112 objective (mainly due to the fact that these are in protected areas) and do not demonstrate this
113 potential synergy (Andersen et al., 2016).

114 In this study, we appraised the climatic benefits from the rewetting of degraded
115 peatlands in terms of reduced GHG emissions (from multi-year data sets) and the possible
116 return of the C sequestration function, as well as the potential for increased biodiversity
117 provision with the aim of identifying the most optimum land use category for future peatland
118 restoration policies. We investigated two rewetted peatlands (originally raised bogs) within
119 the same biogeographical region but under different land uses: industrial peat extraction
120 (which accounts for 7 % of the peatland resource) vs drained/domestic peat extraction (38 %
121 of the peatland resource). We assessed whether rewetting of these degraded peatlands can

122 bring back the biological structure and ecosystem functions typical of a natural raised bog.
123 We hypothesise that both sites are currently on dissimilar successional stages and will,
124 therefore, exhibit contrasting biodiversity and climate benefits.

125

126 **2. Material and methods**

127 *2.1 Study areas*

128 Blackwater, Co. Offaly is located in the Irish Midlands (Supplemental Material (S1)) and is a
129 former raised bog that was drained for milled peat extraction in the late 1950s. At the
130 cessation of extraction in 1999, the drainage pumps were deactivated. Since then, the
131 landscape has developed spontaneously as a composite of microsites; bare peat fields (some
132 with active drainage systems), and re-colonized drained and rewetted areas (Fallon et al.,
133 2012). The general hydrological and nutrient status of Blackwater is representative and
134 typical of the outcome of industrial peat extraction in Ireland (Farrell, 2008), and due to the
135 geographical location in the Irish midlands (a flat low-lying basin), rewetting of these sites
136 usually leads to the formation of shallow lakes. The average thickness of the residual peat
137 layer within the study site is 150 cm and the peat is composed mainly of highly humified
138 *Phragmites* peat overlying limestone parent material (Table 1).

139

140 Moyarwood, Co. Galway is located in the west of the country (S1) and was ditched (similar
141 drainage network to Blackwater) in 1983 but never developed for peat extraction. However,
142 significant degradation (subsidence) due to the drainage and from domestic peat extraction
143 around the margins of the bog and from burning events is evident. The topography is varied
144 but relatively flat on the eastern side where the GHG monitoring plots were established. The
145 drains were active until a rewetting programme commenced in 2012, which consisted of
146 blocking drains with peat dams at regular intervals (generally at any point where there was a
147 fall in a drain level of 10 cm). A drained vegetated area remains on the margins of the bog.
148 The average thickness of the peat within the site is 440 cm and the peat is composed mainly
149 of humified *Sphagnum* peat overlying limestone parent material (Table 1).

150

151 *2.2 Climate*

152 The sites are located within the temperate zone as defined by IPCC (2006) and are
153 characterized by an oceanic climate with prevailing south-west winds, mild mean annual air

154 temperatures and moderately high annual rainfall. Although there is only 30 km distance
 155 between the sites (S1), the 30-year mean annual air temperature (1980-2010) is slightly lower
 156 at Blackwater (9.6 °C) strongly affected by lower winter temperatures in the area, while mean
 157 annual precipitation is 245 mm higher in Moyarwood (Table 1). During the study, potential
 158 evapotranspiration (PE; mm) ranged from 482 to 551 mm at Blackwater and from 502 to 517
 159 mm at Moyarwood (Note: 30-year mean annual PE values are not available for the Met
 160 stations at Gurteen (Blackwater) and Athenry (Moyarwood)).

161

162 **Table 1** Location, climate and physico-chemical characteristics* of the two rewetted sites.
 163 Air temperature and precipitation data (1981-2010) from Met Éireann, Gurteen station
 164 (Blackwater) and Athenry station (Moyarwood).

Site name	Blackwater	Moyarwood
Previous land use	Industrial peat extraction (30 years)	Drained/domestic peat extraction (30 years)
Time of rewetting	1999	2012
Latitude	53°17'48.9"N	53° 20'49.2"N
Longitude	7°57'56.3"W	8°30'55.4"W
Sub-region	Midlands	West
Mean annual air temperature (°C)	9.6	9.9
Mean precipitation (mm yr⁻¹)	948	1193
Current peat type von Post scale	<i>Phragmites</i> H7	<i>Sphagnum</i> H6
Parent material	Limestone	Limestone
Acrotelm	Absent	Present but subsided
Peat depth (m)	1.5	4.4
pH	4.9	4.4
Electrical conductivity (µs cm⁻¹)	350	102
Bulk density (g cm⁻³):		
- drained	0.19	0.13
- rewetted	0.14	0.08
C (%)	52.4	51.5
N (%)	2.14	1.32
C:N	24.5	39
Study period	1/5/2011: 30/4/2015	1/4/2013: 31/3/2015
Number of GHG plots:		
-drained	3	3
-rewetted	11	12

165 *See Renou-Wilson et al. (2018) and related technical report for details regarding soil sampling.

166

167 2.3 Field measurements

168 2.3.1 Biodiversity assessment

169 The hydro-ecological state of each site was assessed on the basis of a habitat assessment,
170 survey of plant communities and indicator species (positive and negative). Four key abiotic
171 and biotic components of biodiversity assessment were identified for the appraisal of the
172 restored/rewetted status of each site, namely 1) hydrological integrity (see details in Renou-
173 Wilson et al., 2018); 2) physico-chemical parameters of peat and drain water (pH, C:N, bulk
174 density, electrical conductivity); 3); micro-habitat assessment (heterogeneity and condition);
175 and 4) vegetation composition (species and abundance).

176 The relative presence of different micro-habitat types (hummocks, pools, hollows,
177 lawns and flats) provide a measure for habitat heterogeneity in typical raised bogs (Schouten,
178 2002). In order to carry out the assessment of the biotic components, five habitat quadrats
179 “HQ” (4 x 4 m plot) were established along a “W” shape transect through representative
180 areas of each site, running perpendicular to the main drainage systems. In each HQ, the
181 vegetation was described by identifying the percentage cover of the main plant functional
182 types (PFT): woody vegetation; ericoid dwarf shrubs, total graminoids (grasses, sedges,
183 forbs), bryophytes (Sphagnum mosses, other mosses, liverworts), lichens (demonstrating
184 absence of burning events), litter and bare peat. Four peat samples and the nearest drain water
185 were also sampled for analysis. Furthermore, the four corners of each HQ formed a
186 vegetation quadrat “VQ” (1 x 1 m) and, therefore, 20 vegetation quadrats were also identified
187 at each site in order to assess the fourth component of our survey method. All taxa of vascular
188 plants, mosses and lichens and their cover values were recorded at each VQ. Algae were
189 recorded but not identified. Cover of each species within a VQ was estimated using a revised
190 Domin scale (Kent and Coker, 1992). The scale includes cover values from 0 to 4, whereby
191 0=absent, 1=rare (<5 %), 2=occasionally (5-20 %), 3=frequently (21-50 %) and 4=dominant
192 (>50 %).

193 Species richness and the Shannon-Wiener Index were determined as indicators of
194 vegetation diversity for each site and were both determined as mean values of the 20 VQs.
195 Species richness was further determined in total species number, total vascular plant species
196 number and total bryophyte species number per site. Dominant PFTs and species for each
197 study site were determined on the basis of the respective cover value medians per VQ. Cover-
198 weighted means of Ellenberg Indicator Values (EIV) were calculated from the VQ data for

199 soil moisture, acidity and nitrogen (Ellenberg and Leuschner, 2010). Indicator values that
200 were unknown or not considered in Ellenberg's publication were based on Hill et al. (1999)
201 or Hill et al. (2007). Nomenclature for vascular plants follows Parnell and Curtis (2012),
202 Atherton et al. (2010) for bryophytes, and Whelan (2011) for lichens.

203

204 2.3.2 Environmental monitoring

205 Weather stations (WatchDog Model 2400; Spectrum Technologies Inc., Aurora, IL, USA)
206 were established at each site and recorded photosynthetic photon flux density (PPFD; μmol
207 $\text{m}^{-2} \text{s}^{-1}$) and soil temperature ($^{\circ}\text{C}$) at 5, 10 and 20 cm depths at 10-min intervals. Additional
208 soil loggers (Hobo External Data Loggers; Onset Computer Corporation, Bourne, MA, USA)
209 were installed at each site to capture potential variation in soil temperatures between drained
210 and rewetted microsites and recorded hourly soil temperatures ($^{\circ}\text{C}$) at 5, 10 and 20 cm
211 depths. Water table levels (WT) were manually measured from perforated dipwells (internal
212 diameter 2 cm) that were inserted adjacent to each GHG measurement collar at fortnightly or
213 monthly intervals and linearly interpolated between measurement days to provide continuous
214 data (Alm et al., 2007; Wilson et al., 2015). Wooden boardwalks were established at each
215 site to facilitate GHG measurements and to prevent damage to the vegetation and
216 compression of the peat. At Blackwater, the boardwalks on the northern side of the collars
217 were elevated (~ 1 m from the ground) to facilitate the installation of extension chambers over
218 the *Phragmites* plants.

219

220 2.3.3 GHG monitoring

221 Blackwater: In 2011, 15 square stainless steel collars (60 x 60 cm) with a channel at the top
222 were inserted to a depth of 30 cm into the peat in the following microsites: reeds (*Phragmites*
223 *australis*; n=8), sedges (*Carex rostrata*, *Eriophorum angustifolium*; n=3) and drained bare
224 peat (n=3). Although open water accounts for approximately one-third of the rewetted area,
225 gas monitoring plots were not established there for logistical reasons.

226 Moyarwood: A total of 15 collars were established in early 2013 with 12 located in the
227 rewetted area along a transect perpendicular to the ditches. One collar in the rewetted transect
228 was suspended within a ditch, supported on wooden batons that extended across the ditch.
229 The collar was placed in a similar manner to the other collars so that the channel at the top of
230 the collar was above the water level. As per IPCC guidance (IPCC, 2014), former ditches

231 were not considered as being separate from the remainder of a rewetted site, and in this
232 instance the collar located in the ditch was considered part of the transect. The vegetation in
233 the collars was representative of the habitats found in the wider site, i.e. flats with some
234 Sphagnum lawn and a few hummocks. Three monitoring plots were located in the drained
235 area on the eastern margin, which was dominated by poorly growing heather (*Calluna*
236 *vulgaris*) and lichens (*Cladonia* spp.). Collars were not established in drainage ditches in the
237 drained areas at either site for logistical reasons; instead we use the IPCC default emission
238 factors (IPCC, 2014) for ditches in drained peatlands where appropriate.
239

240 Greenhouse gas fluxes were measured at fortnightly or monthly intervals from May 2011 to
241 April 2015 at Blackwater and from April 2013 to March 2015 at Moyarwood. Each
242 measurement campaign consisted of 2-4 days. Net ecosystem exchange (NEE) was measured
243 with a static transparent polycarbonate chamber (60 x 60 x 33 cm) equipped with two internal
244 fans to ensure mixing of the headspace air. Vent holes on the chambers ensured that pressure
245 artefacts and ebullition (for methane) were minimised during chamber placement. At
246 Blackwater, extension chambers (Günther et al., 2015; Wilson et al., 2007b) equipped with
247 fans were used for collars that contained *Phragmites australis*. A radiator cooling system
248 (with submerged ice packs) was used to maintain the temperature within the chamber close to
249 the ambient air temperature (Alm et al., 2007), and PPFD was recorded from a sensor (PAR-
250 1; PP Systems, King's Lynn, Norfolk, UK) located in the chamber. NEE was measured under
251 a range of light levels (PPFD; $\mu\text{mol m}^{-2} \text{s}^{-1}$) as the position of the sun changed throughout the
252 day. In early mornings, an artificial shroud that blocked approximately 50 % of incoming
253 PPFD was placed over the chamber to permit the measurement of NEE at low PPFD levels
254 ($<100 \mu\text{mol m}^{-2} \text{s}^{-1}$). CO_2 measurements were carried out between 8 am and 6 pm in the
255 summer and between 9 am and 3 pm in the winter to ensure that the maximum PPFD was
256 reached at each measurement date. Ecosystem respiration (R_{eco}) was then measured by
257 covering the chamber in an opaque cover and CO_2 exchange was measured as outlined above.

258 Methane (CH_4) and nitrous oxide (N_2O) fluxes were measured at monthly intervals
259 (multiple measurements were carried out during the 4 day measurement campaigns in
260 summer) using an opaque, polycarbonate chamber (60 x 60 x 25 cm) equipped with a battery-
261 operated fan that mixed the air within the chamber headspace. As for CO_2 sampling,
262 extension chambers were used in the reed microsite at Blackwater. For a more detailed
263 description of GHG sampling approach and laboratory analysis see Wilson et al. (2013a).

264 To incorporate the seasonal dynamics of the vegetation into CO₂ exchange models, a
265 green area index (GAI) was estimated for each of the vegetated collars. In brief, this involved
266 measuring the green photosynthetic area of all vascular plants (leaves and stems) within five
267 sub-sample plots (8 cm x 8 cm) in the GHG collar at monthly intervals. Moss % cover was
268 estimated at the same time. Species-specific model curves were applied to describe the
269 phenological dynamics of the vegetation of each collar, and the models (vascular plants and
270 moss) were summed to produce a plot-specific GAI (see Wilson et al., 2007a)

271

272 *2.3.4 Flux calculations*

273 Flux rates (mg CO₂ m⁻² h⁻¹, mg CH₄ m⁻² h⁻¹, µg N₂O m⁻² h⁻¹) were calculated as the linear
274 slope of the CO₂, CH₄ and N₂O concentrations in the chamber headspace over time, with
275 respect to the chamber volume, collar area and air temperature. A flux was accepted if the
276 coefficient of determination (r^2) was at least 0.90. An exception was made in cases where the
277 flux was close to zero for e.g. in early morning/late evening when there are light constraints
278 on photosynthetic activity or in winter time when soil processes are typically slower and the
279 r^2 is always low (Alm et al., 2007). In these cases, the flux data were examined graphically
280 and fluxes with obvious nonlinearity (due to chamber leakage, fan malfunction, ebullition,
281 etc.) were discarded. The remainder were evaluated using Akaike's Information
282 Criterion for small sample sizes (AICc) and fluxes that exhibited low AICc values
283 (representing lower variance and better model fitting) were accepted. In this study, we
284 followed the sign convention whereby positive values indicate a flux from the peatland to the
285 atmosphere (source) and negative values indicate a flux from the atmosphere to the peatland
286 (sink). Gross primary production (GPP) was calculated as NEE minus R_{eco} (Alm et al., 2007)
287 and the closest R_{eco} flux value in time to a NEE flux value was used with care taken to ensure
288 that air (within the chamber) and soil temperatures were similar at the time of measurement.

289

290 *2.3.5 Modelling*

291 Statistical and physiological response models were constructed and parameterized for each
292 microsite within the study sites (see S2). Model evaluation was based on the following
293 criteria: (1) statistically significant model parameters ($p < 0.05$), (2) lowest possible standard
294 error of the model parameters and (3) highest possible coefficient of determination (adjusted
295 r^2). During model construction, the relationship between R_{eco}, GPP, CH₄ or N₂O and a range
296 of independent environmental variables (recorded in conjunction with flux measurements)

297 was tested. Only variables that increased the explanatory power of the model were included.
298 The models were accepted if the residuals were evenly scattered around zero. GPP was
299 related to PPF_D using the Michaelis–Menten-type relationship that describes the saturating
300 response of photosynthesis to light (Tuittila et al., 1999), and to GAI and/or water table
301 (S2a). GPP model coefficients and associated standard errors were estimated using the
302 Levenberg–Marquardt multiple nonlinear regression technique (IBM SPSS Statistics for
303 Windows, version 21.0, Armonk, NY, USA). The R_{eco} models are based upon the Arrhenius
304 equation (Lloyd and Taylor, 1994) and are nonlinear models related to soil temperature and
305 water table or volumetric moisture content (S2b). The CH₄ models are nonlinear models
306 related to soil temperature and water table (S2c).

307

308 *2.3.6 Annual GHG balances*

309 The response functions estimated for R_{eco}, GPP and CH₄ were used for the reconstruction of
310 the annual GHG balances for the drained and rewetted areas. Fluxes were reconstructed for
311 each sample plot in combination with an hourly time series of (1) T_{5cm}, (2) WT levels linearly
312 interpolated from weekly measurements, (3) PPF_D values recorded by the weather station
313 and (4) plot-specific modelled GAI that described the phenological development of the
314 vegetation. Annual NEE was calculated as the sum of annual GPP (negative values) and
315 annual R_{eco} (positive values). Annual balances (g C m⁻² yr⁻¹) were calculated for each sample
316 plot by integrating the hourly values over each 12-month period. Annual CO₂ balances from
317 the drained areas in both sites were previously reported in Wilson et al. (2015) and here we
318 provide an additional year of CO₂ data from both sites.

319

320 *2.3.7 Upscaling*

321 High resolution habitat maps were created for each site. A vegetation and habitat mapping
322 survey was carried out and aerial colour photographs at a scale of 1:6000 were used as a base
323 map. The minimum habitat unit size for the survey was set at 0.5 ha. Habitat classifications
324 (Fossitt, 2000) were applied on site using the aerial photos as a guide. Features on the ground,
325 such as vegetation, watercourses and roads were recorded. Once the field work was
326 completed, the maps were digitised using Arc Map10 GIS package, with every unit (in this
327 case vegetation communities, open water, roads etc.) mapped digitally. Guidelines as outlined
328 by Smith et al. (2011) were followed. At Blackwater, the proportion of reeds, sedges and
329 open water in the rewetted site were 13, 57, and 30 % respectively. At Moyarwood, the

330 vegetation was relatively homogenous across the site and so the mean annual CO₂ and CH₄
331 balances from the collars were scaled up to an area of one hectare.

332

333 *2.3.8 Statistical analysis*

334 Statistical analyses were performed using SPSS version 21.0 for Windows (IBM SPSS
335 Statistics for Windows, Armonk, NY, USA). P values smaller than 0.05 were considered
336 statistically significant. All data were tested for normality using the Kolmogorov–Smirnov
337 test. Where the data were not normally distributed, the repeated-measures Friedman and
338 Wilcoxon signed-rank nonparametric tests were used. Uncertainty in reconstructed annual
339 R_{eco} and NEE was calculated by summing up the maximum and minimum standard errors
340 (Renou-Wilson et al., 2014). The species abundance data of rewetted plots in Blackwater and
341 Moyarwood and intact raised bogs were analyzed with multivariate statistics using Canoco
342 4.5. The initial detrended correspondence (DCA) analysis revealed a gradient on the first axis
343 that was longer than 5 SD units. Therefore, the unimodal DCA model of species distribution
344 along gradients was applied.

345

346 *2.3.9 Synergies*

347 The outcome of rewetting / restoration activities are very site specific, which makes valuation
348 of the ecosystem services provided by these sites a challenge (Glenk et al., 2014).
349 Nevertheless, a robust valuation is needed if current land use management and alternative
350 uses are to be compared for local, regional or national policy development, where it is critical
351 to simplify the links between functional processes, such as the condition of the vegetation
352 (composition and cover) and habitats, and the related ecosystem services. We assigned a
353 ranking (1-5) to six metrics (measured in this study) under the categories of climate change
354 mitigation and biodiversity provision. The criteria used to determine the rankings are defined
355 in Table 2. Scaling within the rankings was assigned based on the range in values generally
356 reported in the scientific literature for each metric, although we are cognisant that there is a
357 level of subjectivity to the way that they are used here. For comparative purposes, a ranking
358 of 5 is commensurate to a natural raised bog in most of the metrics. For CH₄, a ranking of 3 is
359 more applicable for a temperate natural site (e.g. Goodrich et al., 2015; Green and Baird,
360 2017).

361

362

363

364 **Table 2.** Criteria used to determine rankings (1-5) for the decision matrix. Positive values for
 365 carbon dioxide (CO₂), methane (CH₄) and nitrous oxide (N₂O) indicate net losses from the
 366 peatland to the atmosphere and negative values (< 0) indicate net uptake by the peatland.

Metric	Units	Ranking				
		1	2	3	4	5
Climate change						
CO ₂	g C m ⁻² yr ⁻¹	> 90	90- 61	60-31	30-1	≤ 0
CH ₄	g C m ⁻² yr ⁻¹	> 30	30-21	20-11	10-1	≤ 0
N ₂ O	g N m ⁻² yr ⁻¹	> 1	0.9 to 0.61	0.6-0.31	0.3-0.1	≤ 0
Biodiversity						
Micro-habitat diversity	-	0 to 0.5	0.6 to 1	1.1 to 2	2.1 to 3	> 3
Bryophyte no.	-	<3	3 to 4	5 to 6	7 to 8	> 9
Shannon Wiener Index	-	0 to 0.99	1 to 1.99	2 to 2.99	3 to 3.99	>4

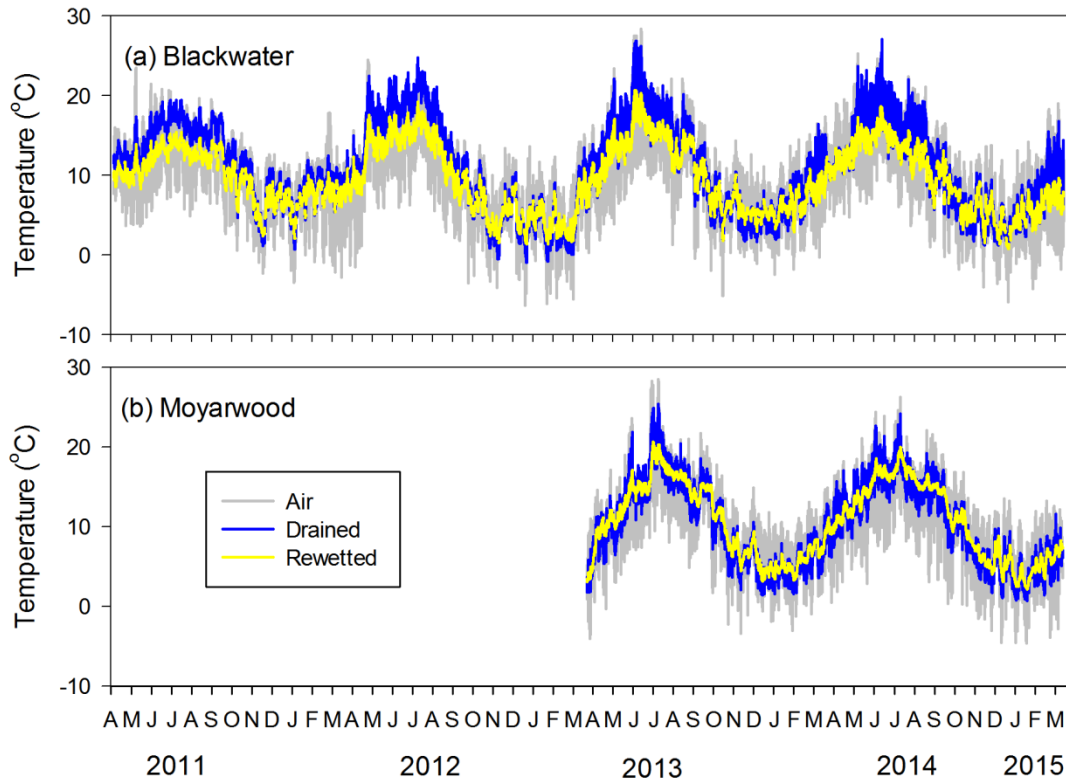
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368

369 **3. Results**

370 *3.1 Weather*

371 A distinct seasonal pattern in air and soil temperatures was evident in both sites throughout
 372 the study (Fig. 1). At Blackwater, the lowest air temperature values (-7 °C) occurred during
 373 the winter of 2012/2013 (Fig. 1a), with highest air temperatures (29 °C) recorded in mid-
 374 summer 2013. Average annual air temperatures for each 12-month period were 9.8, 8.7, 10
 375 and 9.7 with year 2 and 3 varying considerably from the long-term average (Table 1). At
 376 Moyarwood, the lowest air temperature (-4 °C) was recorded in April 2013 and the highest
 377 (29 °C) in mid-summer 2013 (Fig. 1b). Soil temperatures at both sites displayed the same
 378 seasonal patterns as air temperatures but were subject to less hourly variation in both the
 379 drained and rewetted areas.



380

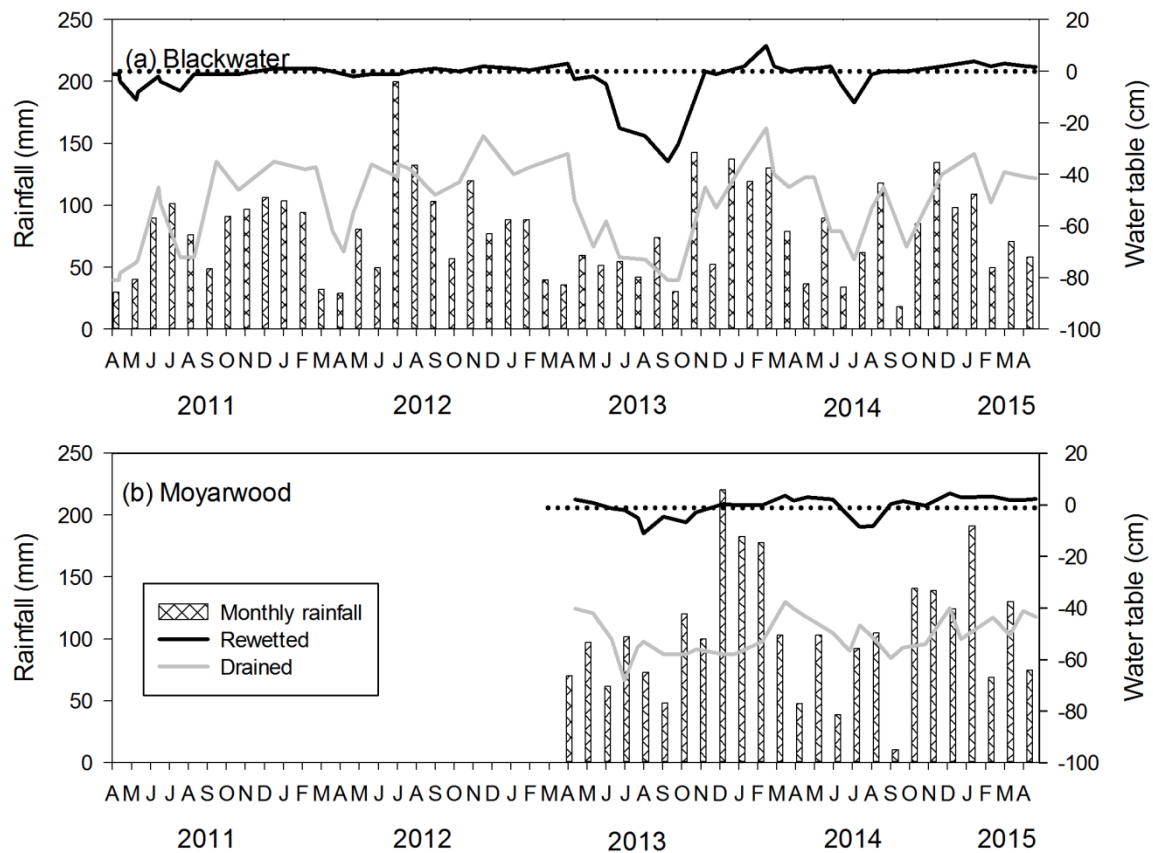
381 **Fig. 1** Hourly air (grey line) and soil temperatures (°C) at (a) Blackwater and (b) Moyarwood
 382 for the duration of the study. Soil temperatures are from 5cm depth in the drained (blue line)
 383 and rewetted (yellow line) areas of the study sites.

384 At Blackwater, rainfall for each 12-month period; 949 mm, 1049 mm, 949 mm and 926 mm
 385 was similar to the long term average (Table 1), although rainfall distribution varied
 386 considerably between years with the summer of 2013 much drier than the others (Fig. 2). At
 387 Moyarwood, the rainfall in the first year of the study (1425 mm) was 19 % higher than the
 388 long-term average, driven by wetter summer and winter conditions, while the second year
 389 (1193 mm) was similar to the long-term average.

390 *3.2 Hydrological and physico-chemical parameters*

391 Both rewetted sites displayed a similar moisture profile with the water table at or just below
 392 the ground surface (Fig. 2). Water table levels displayed spatial and temporal variability over
 393 the monitoring periods (Fig. 2). All the rewetted plots displayed annual mean WT levels
 394 above -20 cm throughout the study and are confirmed as hydrologically restored. In 2013, the
 395 WT in the rewetted areas in Blackwater dropped below -20 cm for 3 months (Fig. 2a),
 396 although a similar response was not observed in Moyarwood at the same time (Fig. 2b). The
 397 WT in the drained areas of both sites were subject to considerable fluctuations throughout the

398 study period with the deepest WT levels observed in Blackwater (-80 cm) in autumn 2013
 399 (Fig. 2a).



400
 401 **Fig. 2** Monthly rainfall (mm; hatched bars) and water table levels (cm) in the rewetted (black
 402 line) and drained (grey line) areas in (a) Blackwater (April 2011 to April 2015) and (b)
 403 Moyarwood (April 2013 to April 2015).

404 Blackwater had a pH of approximately 5 and a high von Post value associated with
 405 *Phragmites* peat (Table 1). This mesotrophic status is due to its basal fen peat now exposed to
 406 the surface after peat extraction and which is also more nutrient rich (C:N = 25). This was
 407 corroborated by the analysis of surface water that showed high pH (7.4) and electrical
 408 conductivity ($316 \mu\text{s cm}^{-1}$) values. In comparison, the surface layer of Moyarwood was more
 409 acidic (pH 4.4) and nutrient poor (C:N = 39) with oligotrophic surface water of low pH (4.3)
 410 and low electrical conductivity ($59.8 \mu\text{s cm}^{-1}$).

411

412 3.3 Habitat and vegetation profile

413 Blackwater lacked micro-habitat heterogeneity as no typical raised bog micro-habitats were
 414 recorded in the surveyed part of the site. Moyarwood displayed a range of typical micro-

415 habitats (mean of 2.4 types of habitats, such as hummocks, hollows, lawns, and flats per
 416 habitat quadrat) but had no pools (Table 3). The proportion of vascular and bryophyte species
 417 differed between sites and was not necessarily reflected in the total species number (Table 3).
 418 Moyarwood had three times more bryophytes than Blackwater but the latter had the highest
 419 number of total species (27 versus 21). The species spread also varied as shown by the ratio
 420 of ‘species number per plot: total species number’: Blackwater displayed a low ratio (18 %)
 421 indicating that some species occurred only together. At Moyarwood, 50 % or more of all
 422 species occurred at all plots. The Shannon Wiener Index was higher at Moyarwood (2.26)
 423 than at Blackwater (1.34).

424 **Table 3** Biodiversity variables, dominant Plant Functional Types (PFT), and dominant
 425 species recorded at both rewetted sites. Micro-habitat diversity: mean of different types
 426 present per quadrat; Ellenberg Index Values (EIV) for moisture, acidity and nitrogen are unit
 427 less.

Site name	Blackwater	Moyarwood
Micro-habitat diversity	0	2.4
Total species	27	21
Bryophyte species	3	9
Bryophyte/total species ratio	0.11	0.43
Shannon Wiener Index	1.34	2.26
Dominant PFTs in habitat quadrats	Sedges Litter	Sedges Ericoids/Sphagnum
Dominant species in vegetation quadrats	<i>Carex rostrata</i> <i>Phragmites australis</i>	<i>Calluna vulgaris</i> <i>Sphagnum capillifolium</i>
EIV moisture	8.4	8.0
EIV acidity	4.5	2.0
EIV nitrogen	3.5	1.5

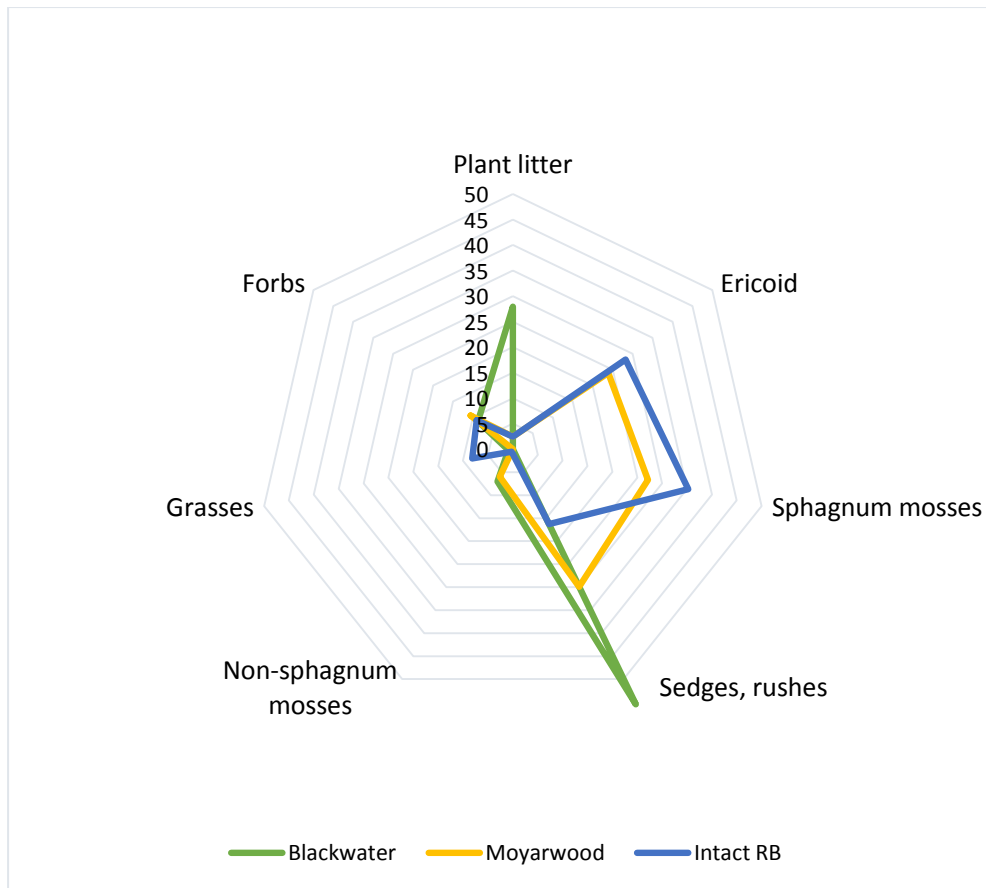
428

429 Woody species, ferns and algae were absent in Moyarwood and only detected in very low
 430 numbers in Blackwater. The most dominant plant functional types (PFTs) at both sites were
 431 ‘sedges’ but with different combination of species occurring, namely *Eriophorum vaginatum*
 432 and *E. angustifolium* with *Carex panicea* in Moyarwood compared to the sedge combination
 433 of *Carex rostrata* and *E. angustifolium* in Blackwater. The second most dominant PFTs in
 434 Moyarwood were ericoids and *Sphagnum* mosses (in particular *Sphagnum capillifolium*)
 435 while plant litter was detected in high amounts at Blackwater. While very common moss
 436 species, such as *Polytrichum commune* and *Campylopus introflexus* regularly occurred in the
 437 more degraded Blackwater site, the fen-specialized bryophyte *Campylium stellatum* was also

438 present. Although not recorded in the quadrats, *Sphagnum subnitens* was observed on very
439 small patches and is known to colonize wet cutaway peatlands after sedges and rushes. At
440 Moyarwood, hummocks were widespread but not very large and were associated with species
441 such as *S. capillifolium*, *S. magellanicum*, *S. papillosum* and *Hypnum cupressiforme*. Another
442 indicator species of healthy natural raised bogs is *S. fuscum*, which was found on one
443 hummock. The average EIVs across all vegetation plots for Moyarwood indicated that the
444 site was wet, quite acidic and infertile. In contrast, Blackwater had higher moisture, acidity
445 and nitrogen EIVs demonstrating that while being very wet, the peat was more nutrient-rich,
446 akin to a ‘fen’.

447 At Moyarwood, species indicative of drier past conditions, for example an abundant
448 cover of *Narthecium ossifragum*, *Trichophorum caespitosum* and *E. vaginatum*, while *C.*
449 *panicea*, a negative indicator particularly in true "Midlands raised bogs" occurred on all
450 vegetation quadrats. Of great significance was the absence of common alien (invasive)
451 species at both sites. Invasive species are considered to be the main direct drivers of
452 biodiversity loss in Europe and a threat to natural habitats in Ireland in particular (Caffrey et
453 al., 2014). Lichens (mostly *Cladonia portentosa*), an indicator of the absence of fire events,
454 were present at Moyarwood but not Blackwater. When comparing our results with typical
455 natural raised bogs that were located in the same region of the studied sites (data extracted
456 from Renou-Wilson et al., 2011), we can assess the success or failure of each site vis-à-vis
457 ultimate objectives of rewetting/restoration. The radar graphics in Fig. 3 display the mean
458 cover of PFTs for both sites compared with a typical natural site. Sphagnum mosses, ericoid
459 and sedges formed a typical assemblage of PFTs in Moyarwood in similar proportion to a
460 natural site, except for the higher cover of sedges and the presence of non-Sphagnum mosses.
461 The PFT profile at Moyarwood had the closest resemblance with that of a natural raised bog.
462 In contrast, Blackwater vegetation demonstrated a clear divergent pattern with
463 ‘sedges/rushes’ as dominant PFT.

464



465

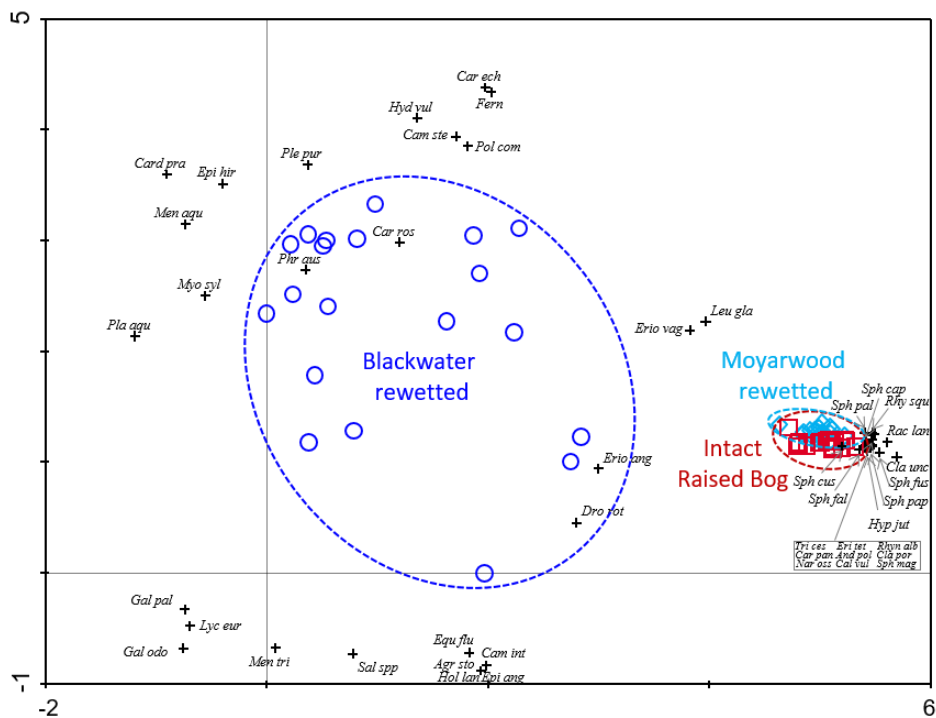
466 **Fig. 3** Spider plot of Plant Functional Types percentage cover recorded at an intact Raised
 467 Bog (RB) and at Blackwater and Moyarwood.

468 The multivariate analysis of the species abundance data via DCA resulted in a sum of all
 469 eigenvalues of 4.752 (Axis 1: 0.903, Axis 2: 0.432). The gradient length of axis 1 was 5.315
 470 and 3.327 for axis 2. The cumulative percentage variance of species data for the first four axis
 471 was: 19.0, 28.1, 33.3 and 36.9. In the DCA biplot of species and samples (Fig. 4), the high
 472 similarity and homogeneity of species composition between rewetted Moyarwood plots and
 473 intact raised bogs was clearly observable from the overlapping sample cloud and the
 474 dominance of *Sphagnum species*. The rewetted Blackwater plots showed a higher
 475 heterogeneity in species composition, which was in parts dominated by *P. australis*. The
 476 distance of the Blackwater samples to the next intact raised bog sample was greater than 2-5
 477 SD units, showing medium to high dissimilarity of the species composition between them.

478

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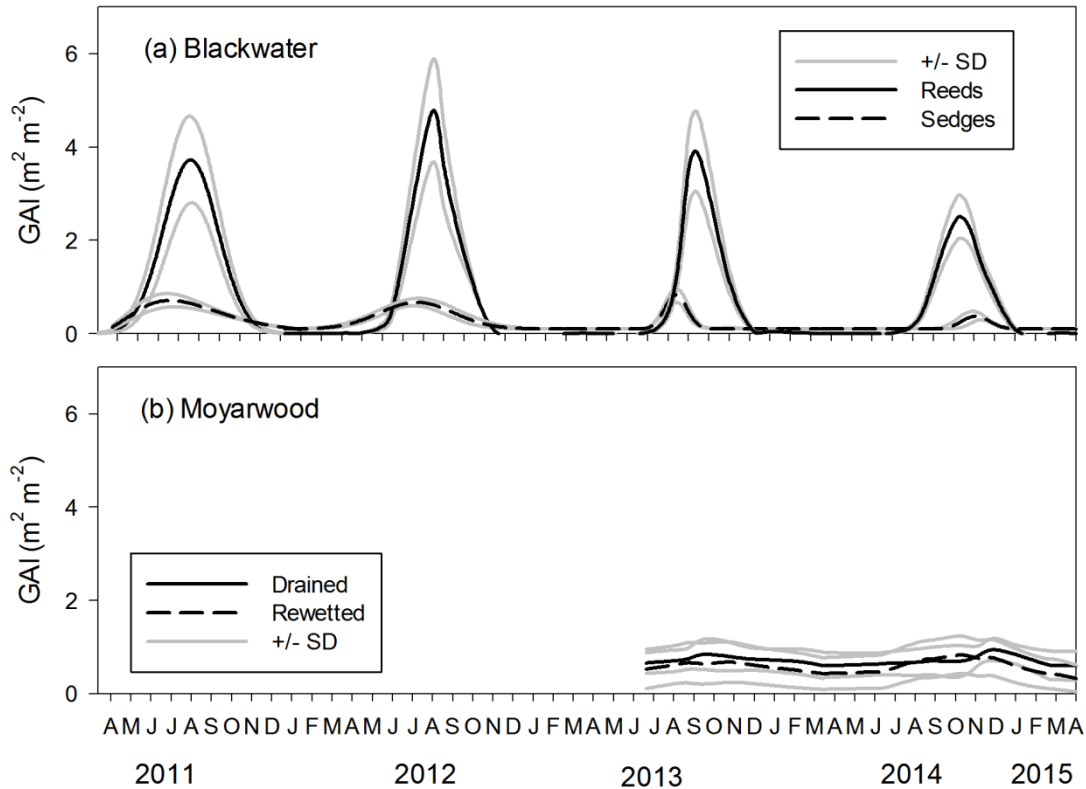
483 **Fig. 4** Detrended correspondence analysis (DCA) biplot of species and sample plots at both
 484 rewetted sites and an intact raised bog. Sum of all eigenvalues of 4.752 (Axis 1: 0.903, Axis
 485 2: 0.432).

486

487 3.4 Green area index (GAI)

488 At Blackwater, average (\pm standard deviation) GAI values in the GHG collars ranged from 0
 489 (non-growing season) to $4.6 \pm 1.1 \text{ m}^2 \text{ m}^{-2}$ in the reeds, and from 0 to $0.8 \pm 0.2 \text{ m}^2 \text{ m}^{-2}$ in the
 490 sedge microsite (Fig. 5a). The drained area at Blackwater was unvegetated. There were
 491 noticeable differences between years in regard to the start and finish of the growing season,
 492 and the maximum GAI value observed in mid-summer (Fig. 5a). The amount of green plant
 493 surface available for photosynthesis over the growing season, as determined by the area under
 494 the GAI curves, varied considerably between years and followed the trend:
 495 2011>2012>2013>2014. This trend was also evident in the sedge microsite. At Moyarwood,
 496 the shape of the GAI curves and average GAI values were similar in both years in the drained
 497 and rewetted areas (Fig. 5b), and ranged from 0.6 to $0.9 \pm 0.3 \text{ m}^2 \text{ m}^{-2}$ in the former and from
 498 0.3 to $0.8 \pm 0.4 \text{ m}^2 \text{ m}^{-2}$ in the latter.

499



500
 501 **Fig. 5** Average modelled green area index (GAI; $\text{m}^2 \text{m}^{-2}$) in (a) Blackwater and (b)
 502 Moyarwood. Grey lines indicate \pm standard deviation (SD) of the mean.

503

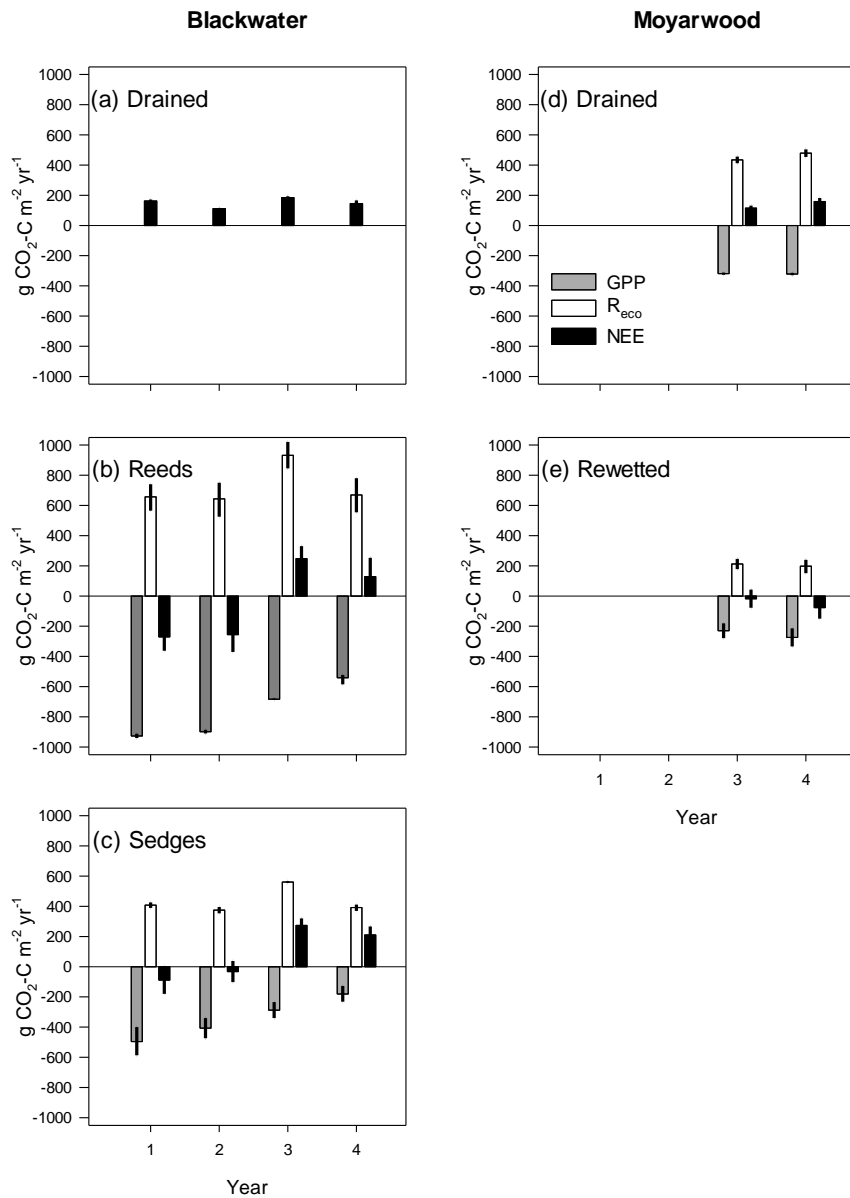
504 3.5 Annual GHG balances

505 3.5.1 Carbon dioxide fluxes

506 The drained areas in both sites were a net annual CO_2 source in all years of the study (Fig. 6a
 507 and 6d). At Blackwater, CO_2 emissions (in the absence of a vegetation component R_{eco} and
 508 NEE are analogous) were highest in year 3 ($185 \pm 8 \text{ g C m}^{-2} \text{ yr}^{-1}$) and lowest in year 2 ($111 \pm$
 509 $5 \text{ g C m}^{-2} \text{ yr}^{-1}$). The 4-year average (\pm standard error) was $151 \pm 11 \text{ g C m}^{-2} \text{ yr}^{-1}$. At
 510 Moyarwood, R_{eco} was higher than GPP in both years in the drained area and resulted in net
 511 CO_2 losses of $115 \pm 26 \text{ g}$ and $159 \pm 23 \text{ g C m}^{-2} \text{ yr}^{-1}$ in years 1 and 2 respectively. The 2-year
 512 average was $137 \pm 24 \text{ g C m}^{-2} \text{ yr}^{-1}$.

513 The reed microsite at Blackwater was a very strong CO_2 sink in the first two years of
 514 the study; -271 ± 92 and $-255 \pm 118 \text{ g C m}^{-2} \text{ yr}^{-1}$ (Fig. 6b). In year 3, R_{eco} increased by 45 %
 515 and GPP dropped by 24 % from the values estimated in the previous year. This produced a
 516 very strong CO_2 source of $248 \pm 86 \text{ g C m}^{-2} \text{ yr}^{-1}$. In year 4, both R_{eco} and GPP decreased and

517 the microsite was a smaller CO₂ source of $129 \pm 124 \text{ g C m}^{-2} \text{ yr}^{-1}$. Average NEE (4-year) for
 518 the reed microsite was $-37 \pm 229 \text{ g C m}^{-2} \text{ yr}^{-1}$. The sedge microsite followed the same trend as
 519 the reed microsite; a small CO₂ sink in years 1 and 2, and a strong CO₂ source in years 3 and
 520 4 (Fig. 6c). Average NEE (4-year) for the sedge microsite was $90 \pm 68 \text{ g C m}^{-2} \text{ yr}^{-1}$. Average
 521 NEE for the rewetted area (excluding open water) was estimated at $66 \pm 168 \text{ g C m}^{-2} \text{ yr}^{-1}$.



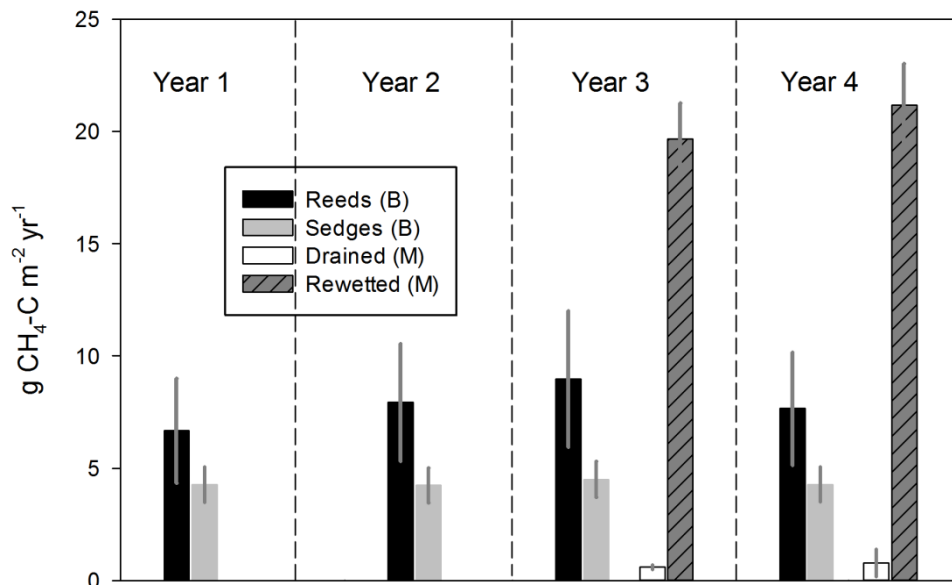
522

523 **Fig. 6** Average annual modelled net ecosystem exchange (NEE), gross photosynthesis (GPP)
 524 and ecosystem respiration (R_{eco}) and (g C m⁻² yr⁻¹) ± standard errors in Blackwater (left side
 525 panels) and Moyarwood (right side panels). Positive values indicate net loss of carbon
 526 dioxide (CO₂) from the peatland to the atmosphere and negative values indicate net CO₂
 527 uptake by the peatland.

528 At Moyarwood, the rewetted area was a CO₂ sink in both years of the study (Fig. 6e).
 529 R_{eco} was similar between years, although GPP was 20 % higher in the second year. Average
 530 NEE (2-year) for the rewetted area was estimated at $-49 \pm 68 \text{ g C m}^{-2} \text{ yr}^{-1}$.

531 *3.5.2 Methane and nitrous oxide fluxes*

532 Methane fluxes at the drained area in Blackwater were below the detection level of our
 533 equipment, so a value of zero was assigned (Fig. 7). Both rewetted microsites in Blackwater
 534 were annual CH₄ sources in all years of the study (Fig. 7). Emissions ranged from 6.7 to 9
 535 C m⁻² yr⁻¹ in the reed microsite and 4.2 to 4.5 g C m⁻² yr⁻¹ in the sedge microsite. Average
 536 CH₄ emissions for the upscaled rewetted area at Blackwater (excluding open water) were
 537 estimated at $5.0 \pm 2.2 \text{ g C m}^{-2} \text{ yr}^{-1}$. At Moyarwood, average annual CH₄ emissions (linearly
 538 interpolated between measurements) in the drained area were $0.77 \text{ g} \pm 0.49 \text{ g C m}^{-2} \text{ yr}^{-1}$ (Fig.
 539 7). Average CH₄ emissions for the rewetted area were $19.7 \pm 5 \text{ g C m}^{-2} \text{ yr}^{-1}$. Nitrous oxide
 540 (N₂O) fluxes were below the detection level of the equipment throughout the study period
 541 and were therefore assigned a value of zero.



542
 543 **Fig. 7** Average annual modelled CH₄ fluxes ($\text{g C m}^{-2} \text{ yr}^{-1}$) \pm standard error in Blackwater (B)
 544 and Moyarwood (M). Methane fluxes at the drained area in Blackwater were below the
 545 detection level of our equipment, so a value of zero was assigned.

546

547

548 3.5.2 Synergies

549 In the decision matrix, climate change benefits from rewetting were slightly higher at
 550 Moyarwood where the site was ranked at 5 (maximum) for both CO₂ and N₂O (Table 4). For
 551 CH₄, it ranked lower than Blackwater reflecting the disparity in annual CH₄ emissions
 552 between the two sites. However, in term of biodiversity provision, Moyarwood was over
 553 twice as high as Blackwater, in particular for micro-habitat diversity and bryophyte numbers.
 554 Overall, Moyarwood ranked 1.5 times higher than Blackwater.

555 **Table 4** Decision matrix for ecosystem services; climate change mitigation and biodiversity
 556 provision for the Blackwater and Moyarwood sites. See Table 2 for criteria used to determine
 557 rankings.

Site/Service	Climate change mitigation				Biodiversity				Overall total
	CO ₂	CH ₄	N ₂ O	Sub-total	Micro-habitat diversity	Bryophytes species	Shannon Wiener Index	Sub-total	
Blackwater	2	4	5	11	1	2	2	5	16
Moyarwood	5	3	5	13	4	5	3	12	25

558

559

560 **4. Discussion**

561 Restoration has been defined as “assisting the recovery of an ecosystem that has been
 562 degraded, damaged or destroyed” and can be viewed as a *process* and/or an *outcome*
 563 (McDonald et al., 2016); both with well-defined goals (Rocheftort, 2000). In terms of the
 564 latter, the widest possible range of ecosystem services is re-established and the “new”
 565 ecosystem looks and functions similar to a historical pre-disturbed/degraded ecosystem (i.e.
 566 the target reference state), although a future driven focus has also been proposed (e.g. Martin-
 567 Ortega et al., 2017). However, in peatlands where much of the peat has been extracted (e.g.
 568 the raised bogs used for industrial peat extraction), this goal is impossible to achieve (at least
 569 in the short- and medium term) and the development of a fen ecosystem is a more feasible
 570 option (e.g. Cobbaert et al., 2004; Rocheftort et al., 2016). Clearly, it is important to establish

571 at the outset the criteria by which success will be evaluated and to thereby determine the
572 position of a particular site on a restoration trajectory (Andersen et al., 2016). While
573 cognisant that other key components of peatland functioning, such as microbial community
574 structure (e.g. Urbanová et al., 2011), faunal composition (e.g. Hannigan et al., 2011), water
575 quality (Martin-Ortega et al., 2017) and water borne carbon exports (e.g. Waddington et al.,
576 2008), for example, were not quantified in this study, the two sites in this study, nonetheless,
577 provide a valuable case study in regard to peatland restoration given that they were both
578 raised bogs prior to disturbance (of varying intensity) but have plainly developed along
579 fundamentally different restoration trajectories with dissimilar hydrological regimes,
580 vegetation composition and GHG dynamics.

581

582 *4.1 Hydrological regime*

583 From a hydrological standpoint both sites have been successfully rewetted, as ascertained by
584 the water table remaining at, just below or above the surface for the majority of the study
585 period (Fig. 2), as well as by the high EIV moisture values associated with the current
586 vegetation (Table 3). Species assemblages that form active raised bog require a mean water
587 level near or above the surface of the bog lawns for most of the year (Robroek et al., 2007).
588 In addition, seasonal fluctuations should not exceed 20 cm in amplitude and should only be
589 10 cm below the surface except for very short periods of time (NPWS, In press). Overall, the
590 two sites have been successfully ‘re-plumbed’ and both the “passive” rewetting at Blackwater
591 (i.e. cessation of drain pumping) and the more active approach at Moyarwood (i.e. blocking
592 of drains) have created a wet environment similar to the pre-disturbed ecosystem.

593 In terms of water chemistry, the electrical conductivity values varied between sites
594 with Moyarwood displaying values similar to a nutrient poor raised bog (Kettridge et al.,
595 2011; Van der Schaaf et al., 2002), while the electrical conductivity values at Blackwater are
596 more consistent with nutrient rich peatlands (Van der Schaaf et al., 2002). The pH of both
597 sites are comparable to natural sites, although the soil pH at Blackwater is at the lower range
598 of the mesotrophic category (Jeglum, 1971), similar to natural fens (Doyle and O' Críodáin,
599 2003). This is to be expected as (a) Blackwater is fed by relatively alkaline water influenced
600 by the limestone parent material beneath the peat and (b) the presence of residual fen peat
601 following extensive extraction of the more acidic upper peat layers.

602

603 *4.2 Species diversity*

604 Species diversity is typical of the natural counterparts: typically low in the less
605 degraded site at Moyarwood and with a higher number of species recorded in Blackwater due
606 to the water influx. The presence of monodominant stands such as *P. australis* and the
607 absence of a well-developed bryophyte layer at Blackwater signify a transition mire (from
608 open water to fen). If the site remains wet, the development of a new ‘moderately rich’
609 ecosystem is the most likely trajectory with an increased species of mosses. This may not
610 happen naturally, however, given the lack of seed banks in the locality and the difficulties in
611 reintroducing fen bryophytes in particular (Rocheffort et al., 2016). Following the Canadian
612 experience on similar sites, intervention may be required with a combination of mechanized
613 diaspore transfer and fertilization with phosphorus (the limiting nutrient in such sites and
614 present in very low amounts in Blackwater (0.033 %) (F. Renou-Wilson, unpublished data).
615 It may also be possible to transfer specific *Sphagnum* spp. since they are known to expand at
616 higher pH levels amongst fen communities if precipitation remains high (Vicherová et al.,
617 2017).

618 On the other hand, increased frequency of dry events, whereby the water table drops
619 below -20 cm may be a further obstacle to bryophyte establishment as grasses and forbs will
620 compete with the existing Cyperaceae and Juncaceae species; typical species of degraded
621 bogs that are present in the surrounding areas. This continued increase in species is not
622 desirable from a biodiversity perspective (diversity of biological information) since
623 homogenization is at play (i.e. where most of the new species found in a habitat are also
624 found in other ecosystems in the locality).

625 The presence of *Sphagnum capillifolium* at Moyarwood is of high significance as it
626 indicates the presence of water table levels close to the surface and above. The presence of
627 other *Sphagnum* species such as *S. papillosum* and *S. magellanicum* strongly indicates that
628 vegetation succession at this site is on the correct trajectory. The type of *Sphagnum* species
629 present is also critical for certain ecosystem processes, such as peat formation. For example,
630 studies have shown that *Sphagnum cuspidatum* litter readily decomposes (Belyea, 1996;
631 Hogg, 1993), particularly the young stems (Limpens and Berendse, 2003). The Moyarwood
632 site has recovered a palette of *Sphagnum* species that has permitted the return of the C
633 sequestration function as well as increased biodiversity. The latter was also supported by the
634 presence of other raised-bog specialized species (e.g. the carnivorous *Drosera* spp.)
635 confirming the return of wet and acidic environmental conditions but also demonstrates the
636 increased conservation value of the site, as these species do not occur in any other ecosystems
637 on the island.

638

639 4.3 GHG dynamics

640 The drained areas of both sites were annual CO₂ and CH₄ sources driven largely by soil
641 temperatures (Figure 1, S3 in Supplemental Information). The annual values observed here
642 are within the range reported by Wilson et al. (2015) for a large number of peat extraction
643 sites in Ireland and the UK (that also used data from both the sites in this study), but lower
644 than the derived emission factors for drained peat extraction sites in the temperate zone
645 reported by Drösler et al. (2014). In the latter study, the CO₂ emission factor in particular was
646 strongly affected by data from Canadian peatland sites, where the CO₂ emissions were much
647 higher than extraction sites in Europe (see Wilson et al. 2015, for a more detailed discussion).
648 While the CO₂ emissions reported for our sites are lower than other drained peatland land
649 uses (e.g. cropland, grassland) they are still substantial, especially when compared to mineral
650 soils (Abdalla et al., 2013). Therefore, if these emissions were replicated across the relatively
651 large areas affected by peat extraction in Ireland (Malone and O'Connell, 2009), then these
652 degraded ecosystems represent a substantial hotspot of GHG emissions at the national scale
653 (Wilson et al., 2013b).

654 Rewetting has had a significant impact on GHG dynamics at both sites in this study.
655 Although, the rewetted area at Blackwater was found to be a net CO₂ source, rewetting still
656 represents a significant saving in terms of avoided emissions (and thereby has a direct climate
657 benefit) in comparison to the drained area (Fig. 8). The 4-year average value reported here for
658 the rewetted area at Blackwater ($66 \pm 168 \text{ g CO}_2\text{-C m}^{-2} \text{ yr}^{-1}$) is slightly higher than the Tier 1
659 value (0.50) for nutrient rich temperate peatlands derived by Blain et al. (2014) for the IPCC
660 Wetlands Supplement, although the very wide uncertainty range associated with our value is
661 indicative of a site that fluctuates considerably from sink to source over a short time scale
662 (i.e. 4 years). Nevertheless, it would suggest that the site has moved significantly along a
663 restoration/rehabilitation trajectory. However, GHG emissions/removals from the large body
664 of open water at the site were not quantified in this study and could potentially represent
665 significant hotspots of CO₂ and CH₄ emissions. In the absence of truly comparable studies,
666 values reported by Franz et al. (2016) for a eutrophic rewetted fen would suggest that this
667 could be the case. Notwithstanding, there is potential for further improvements at this site,
668 particularly if *Phragmites* reeds were to colonize the open water area. Our results indicate
669 that *Phragmites* is an effective C sequestration species when water levels remain close to the
670 surface, as has been reported in other post-rewetting peatland studies (Günther et al., 2015;

671 Minke et al., 2016), but can quickly become a large CO₂ source during drier years (Günther
672 et al., 2015). Annual CH₄ emissions from *Phragmites* in this study were similar to those
673 reported by Huth et al. (2013) for a rewetted fen for a year with high rainfall, but lower than
674 values recorded in other comparable rewetted sites (Minke et al., 2016; van den Berg et al.,
675 2016). However, some studies have shown that the measurement of CH₄ fluxes in *Phragmites*
676 stands with opaque chambers (as in this study) could be underestimated as emissions in these
677 plants are strongly controlled by light levels (Minke et al., 2014), although this effect may
678 only be an issue during mid-summer (Günther et al., 2014). Methane emissions from the
679 sedge microsite were low and comparable to natural peatland ecosystems (e.g. Laine et al.,
680 2007).

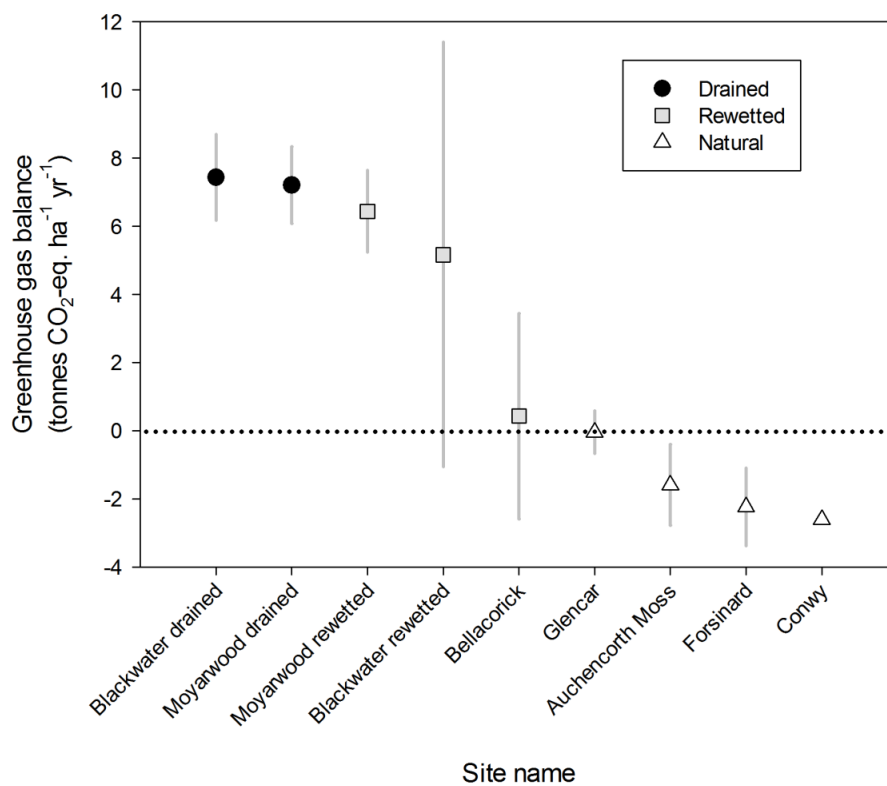
681 Methane emissions at Moyarwood in the first two years post-rewetting were higher,
682 particularly in comparison to natural sites (e.g. Laine et al., 2007; Levy and Gray, 2015). It
683 may be possible that the high initial CH₄ emissions will decline over time as the litter
684 inundated during rewetting activities is decomposed (Limpens et al., 2008); GHG monitoring
685 is ongoing at the site and an extended dataset will provide valuable information. Over the 2-
686 year study period, Moyarwood was a small net annual CO₂ sink (with high uncertainty
687 associated with the estimate) and exhibited values similar to natural sites (e.g. McVeigh et
688 al., 2014). Although rainfall was similar between the two sites for most of the summer
689 months in 2013, the monthly value for July in Moyarwood (101.5 mm), however, was over
690 twice as high as recorded for the Blackwater area (42 mm) and undoubtedly ensured that the
691 former site acted as an annual CO₂ sink, even in that relatively “dry” year. Moreover, the
692 abundance of *Sphagnum* species at Moyarwood may have (a) acted as a buffer to the drought-
693 like conditions (particularly as the rainfall in July would have rehydrated the mosses) and
694 may have helped prevent excessive water drawdown during that period similar to natural
695 bogs (Waddington et al., 2011) and (b) enhanced CO₂ uptake simply because they
696 decompose much more slowly than other species (Clymo, 1965). Nitrous oxide fluxes were
697 not detected at either site, as also reported for some rewetted peatlands (e.g. Günther et al.,
698 2015; Minke et al., 2016; Wilson et al., 2016b), although low N₂O fluxes have been reported
699 for other rewetted studies (see Wilson et al., 2016a).

700

701 4.4 Synergies

702 Successful rewetting of degraded peatlands is a major challenge and, in some cases, may be a
703 balancing act between biodiversity or climate benefits. In this study, the climate benefits from
704 rewetting were slightly higher at Moyarwood where the strong CO₂ sink function offset the

705 substantial CH₄ emissions (Table 4). Nonetheless, both sites have a warming impact on the
 706 climate and are clearly some distance away from functioning like natural sites (Fig. 8).
 707 Moreover, emissions from the open water at Blackwater (if quantified) could produce a
 708 situation where rewetting may actually be more deleterious for the climate than drainage,
 709 particularly if CO₂ emissions from drained areas were relatively low compared to other
 710 drained peatlands (Drösler et al. 2014). However, in regard to biodiversity provision,
 711 Moyarwood was clearly superior to Blackwater in all aspects (Table 4) and would indicate
 712 that rewetting of this land use category could provide the desired synergies between climate
 713 regulation and biodiversity provision.



714
 715 **Fig. 8** Greenhouse gas balance (tonnes CO₂-eq. ha⁻¹ yr⁻¹) at the drained and rewetted sites at
 716 Blackwater and Moyarwood. Dissolved organic carbon (DOC) and CH₄ emissions from
 717 drainage ditches were calculated using guidance provided by IPCC (IPCC, 2014). Methane
 718 and N₂O fluxes are converted to CO₂-equivalents (t CO₂-eq ha⁻¹ yr⁻¹) according to their
 719 global warming potential (GWP) on a 100-year timescale: CH₄ = 28 and N₂O = 265 (Myhre
 720 et al., 2013). Error bars indicate standard deviation (not available for Conwy). Positive values
 721 indicate a net warming impact on the climate and negative values indicate a net cooling
 722 impact. Ombrotrophic sites within the British Isles with long-term GHG datasets (>4 years)
 723 at Bellacorick, Ireland (Wilson et al., 2016b), Glencar, Ireland (Koehler et al., 2011;

724 McVeigh et al., 2014), Auchencorth Moss, Scotland (Dinsmore et al., 2010; Dinsmore et al.,
725 2013; Helfter et al., 2015), Forsinard, Scotland (Levy and Gray, 2015) and Conwy, Wales
726 (CEH, unpublished data) are shown for comparative purposes.

727

728 **5. Conclusions**

729 In Ireland, the current state of peatlands and the consequences of widespread degradation in
730 terms of loss of various ecosystem services have been highlighted (e.g. Renou-Wilson et al.,
731 2011; Wilson et al., 2013b), thereby establishing a framework for the development of the first
732 National Peatlands Strategy (NPWS, 2015). Rewetting as an initial step towards restoration
733 or towards new functioning ecosystems is called upon to meet not only the biodiversity
734 targets but also climate change mitigation targets. However, from an economic point of view
735 it is clearly impossible to rewet all the degraded peatland sites, assuming that they could be
736 rewetted in the first instance (Andersen et al., 2016). Instead, it is more realistic to target
737 specific sites that maximize the synergistic benefits of biodiversity provision and climate
738 change mitigation and, therefore, provide effective cost benefits (Bonn et al., 2014).

739 The results from this study would indicate that raised bogs, which have been degraded
740 through drainage and domestic extraction on the margins, could be prime candidates for
741 future peatland restoration efforts in Ireland in that they provide suitable synergies between
742 climate change mitigation and biodiversity provision. However, we accept that this
743 prioritizing approach could mean that some industrial extraction sites will remain drained
744 even after extraction has ceased. Nonetheless, given the large areas potentially associated
745 with this land use category (38% of the national peatland resource versus 7% for industrial
746 extraction) (Malone and O'Connell, 2009), actions (both financial and practical) should be
747 directed at suitable sites that can be (a) successfully rewetted and (b) provide the
748 biodiversity/climate benefits outlined above. Site-specific evaluations (micro-habitat and
749 vegetation assessment), regardless of land use category, prior to the initiation of restoration
750 actions, would also be valuable for the decision making process. In this regard, hydrological
751 modelling (Mackin et al., 2017) and remote sensing techniques (e.g. Connolly and Holden,
752 2017) would also provide critical information to allow for targeted conservation, rewetting or
753 restoration efforts but these approaches must be backed up by appropriate fiscal instruments
754 either through direct funding from government or from carbon/biodiversity offset schemes,
755 such as in operation in other jurisdictions (Tanneberger and Wichtmann, 2011).

756

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764

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