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1 **Title:** Evaluation of the ECOSSE model for simulating soil carbon under short rotation
2 forestry energy crops in Britain

3 **Running title:** Modelling soil carbon under SRF in Britain

4

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17 Key words: ECOSSE model, soil carbon, short rotation forestry, energy crops, process-based
18 model, land-use change.

19 **Type of paper:** Original Research Article

20

21 **Abstract**

22 Understanding and predicting the effects of land-use change to short rotation forestry (SRF)
23 on soil C is an important requirement for fully assessing the C mitigation potential of SRF as
24 a bioenergy crop. There is little current knowledge of SRF in the UK and in particular a lack
25 of consistent measured datasets on the direct impacts of land use change on soil C stocks.

26 The ECOSSE model was developed to simulate soil C dynamics and greenhouse gas (GHG)
27 emissions in mineral and organic soils. The ECOSSE model has already been applied
28 spatially to simulate land-use change impacts on soil C and GHG emissions. However, it has
29 not been extensively evaluated under SRF.

30 Eleven sites comprising 29 transitions in Britain, representing land-use change from non-
31 woodland land uses to SRF, were selected to evaluate the performance of ECOSSE in
32 predicting soil C and soil C change in SRF plantations.

33 The modelled C under SRF showed a strong correlation with the soil C measurements at both
34 0-30 cm ($R = 0.93$) and 0-100 cm soil depth ($R = 0.82$). As for the SRF plots, the soil C at the
35 reference sites have been accurately simulated by the model. The extremely high correlation
36 for the reference fields ($R \geq 0.99$) shows a good performance of the model spin-up. The
37 statistical analysis of the model performance to simulate soil C and soil C changes after land-
38 use change to SRF highlighted the absence of significant error between modelled and
39 measured values as well as the absence of significant bias in the model.

40 Overall, this evaluation reinforces previous studies on the ability of ECOSSE to simulate soil
41 C and emphasize its accuracy to simulate soil C under SRF plantations.

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46 **Introduction**

47 At the ecosystem scale the average total carbon (C) stock (including soil) of temperate forest
48 biomes is approximately 280 tC ha^{-1} which is equivalent to $1030 \text{ tCO}_2 \text{ ha}^{-1}$. (Saugier et al.,
49 2001; Grace, 2005). In order to quantify the Great Britain (GB) woodfuel resource McKay et
50 al. (2003) carried out a thorough assessment of the standing biomass in GB forests. Based on
51 the results presented by McKay et al. (2003), Morison et al. (2012) reported an average figure
52 for UK woodland C stock in trees of approximately $209 \text{ tCO}_2 \text{ ha}^{-1}$

53 Average soil C for woodland in the UK varies greatly with soil type, but a GB average value
54 is approximately $859 \text{ tCO}_2 \text{ ha}^{-1}$ (down to 1 m soil depth; Morison et al., 2012). Morison et al.
55 (2012) also reported that the C in the litter adds an additional $60 \text{ tCO}_2 \text{ ha}^{-1}$, and that to this
56 should be added the deadwood or coarse woody debris component, estimated at $3 \text{ tCO}_2 \text{ ha}^{-1}$
57 (Gilbert, 2007). Therefore, Morison et al. (2012) suggest that the average UK woodland C
58 stock is $1131 \text{ tCO}_2 \text{ ha}^{-1}$, about 10% more than the reported temperate biome value. This
59 figure may be surprising, as much of the woodland area in the UK is relatively young, but it
60 is largely because of the large soil C stock in peatland areas (Morison et al., 2012). Morison
61 et al. (2012) therefore concluded that the average soil C for GB is $778 \text{ tCO}_2 \text{ ha}^{-1}$, and the
62 average woodland C stock is then estimated at $1051 \text{ tCO}_2 \text{ ha}^{-1}$, excluding the deep peat C
63 stock and areas.

64 Forest soils usually contain more C than equivalent soils under cropland, due to repeated
65 mechanical disturbance during cropping, fallow periods, reduced plant inputs under cropland
66 compared to trees and the removal of a large fraction of C sequestered by crop production in
67 grain (e.g. Mann, 1986; Grigal and Berguson, 1998). Forest soils also usually contain more C
68 than soils under grassland (Guo and Gifford 2002). Furthermore, forest C sinks play an
69 important role in the Kyoto Protocol, both under article 3.3 for afforestation/reforestation/

70 deforestation (ARD) activities, and article 3.4 for forest management activities (Smith et al.,
71 2005). Therefore, increasing forest areas could help sequester C in the soil and providing
72 accurate estimates of changes in forest soil C are of critical importance.

73 There has been long-standing interest in biomass fuel in the UK since the 1970s oil crisis.
74 Willow grown as short rotation coppice (SRC) is the most common woody perennial crop
75 (Hardcastle 2006), but other species such as poplar and sycamore have also been
76 investigated. The concept of short rotation forestry (SRF) is distinct from SRC. The
77 underlying principle is to grow a plantation at close spacing (up to 5000 plants/ha) and then
78 fell it when the trees reach a size that is easily harvested and handled (Mitchell et al., 1999).
79 Short rotation forestry is considered as encompassing woody crops grown for between 8 and
80 20 years, i.e. much shorter than traditional forestry practice, but longer than SRC. The aim of
81 SRF is to harvest the crop at an appropriate age and to remove only the stem wood. Leaving
82 the plant residues on site may have a positive impact from the aspect of reduced nutrient
83 removal as the wood contains less than 10% of the nutrients of the above-ground biomass of
84 the trees (Hardcastle, 2006).

85 Following afforestation, changes occur in the quality and quantity of C inputs (Romanyá et
86 al., 2000; Paul et al., 2002). The capacity of afforestation to increase soil C is highly variable,
87 and is dependent on edaphic (e.g. soil type), climatic (e.g. precipitation) and biotic (e.g.
88 species choice) factors, as well as land-use history (Paul et al., 2002; Laganière et al., 2010).

89 The balance between C inputs, in the form of litter and root exudates and/or fine root
90 turnover, and losses through decomposition determines whether the ecosystem is a sink or a
91 source of C. Evaluating the C dynamics of this type of system requires data on the size of the
92 C pool, the magnitude of the C input and output fluxes, as well as information about the
93 mechanisms involved in controlling flux dynamics. To promote the C sink status of tree

94 plantations, it is therefore imperative to determine the mechanisms involved in controlling
95 soil C dynamics and more specifically in the storage of C in the soil after afforestation
96 (Laganière et al., 2010). Despite the considerable soil C sequestration potential that
97 afforestation offers, many studies have reported contradictory findings (Mc Kay, 2011). The
98 magnitude and direction of the change in soil C after afforestation is strictly dependent to the
99 previous land use (arable/grassland), the soil type (mineral/organo-mineral) and land
100 preparation technique. Hence, afforestation could result in either a decrease (Ross et al.,
101 1999; Farley et al., 2004) or an increase in soil C (Del Galdo et al., 2003), or had a negligible
102 effect (Davis et al., 2007; Smal & Olszewska, 2008). Nevertheless, a trend appears to
103 emerge: afforestation frequently shows an initial loss in soil C during the first few years,
104 followed by a gradual return of C to levels comparable to those in the control soil, and then
105 increasing to generate net C gains in some cases (Paul et al., 2002; Davis et al., 2007).

106 Short rotation plantations do not usually replace undisturbed plant communities, but most
107 often are established on previously cultivated land, either those presently under arable crops
108 or under grass cover. In many cases, this is characterized as 'marginal crop land'. Such land
109 is likely to have lost 30% or more of the original soil C through cultivation and associated
110 erosion (Grigal and Berguson, 1997). The effect of land-use to short-rotation biomass
111 plantations on soil C has become relevant because of links to atmospheric CO₂ enrichment,
112 climate change, and related environmental issues. However, there is little current knowledge
113 of SRF in the UK and the lack of consistent data sets on afforested SRF systems (Rowe et al.,
114 2009), which in turn is mainly due to inconsistent experimental designs, sampling methods
115 and/or soil analysis techniques, results in high uncertainty on the effect of land-use change to
116 SRF on soil C.

117 Soil C sequestration is often estimated using numerical soil/ecosystem models. There are
118 many types of soil C decomposition models including: (1) single pool first order

119 decomposition rate models, (2) food-web models using nitrogen (N) and C interchanges
120 between soil organisms, (3) cohort models describing decomposition as a continuum and (4)
121 process based multi-compartment models such as RothC (Coleman & Jenkinson, 1999).
122 These models have varying levels of complexity and their utility will depend on the data sets
123 available for their parameterization (Dondini et al., 2010).

124 Several models have been developed in an attempt to quantify C from a vast range of mineral
125 soils. Process-based models have been developed from an understanding of how soil C is
126 affected by soil properties, land management and weather fluctuations. Incorporation of these
127 detailed processes and levels of understanding means these process-based models are
128 important, and often successful at predicting not just soil C but also greenhouse gas (GHG)
129 emissions at site level (Bell et al., 2012). However, model testing is often limited by a lack of
130 field data to which the simulations can be compared (Desjardins et al. 2010).

131 The requirement to simulate the C and N cycles using minimal input data on both mineral and
132 organic soils led to the development of the ECOSSE model (Smith et al. 2010a, b). ECOSSE
133 is a process-based model designed to simulate soil C and N dynamics and GHG emissions
134 from mineral and organic soils using only data that are commonly available at a regional
135 scale (Bell et al., 2012). The ECOSSE model has already been validated and applied spatially
136 to simulate land-use change impacts on soil C and GHG emissions over different soil types,
137 to simulate soil C change under energy crops and to simulate soil N and nitrous oxide (N₂O)
138 emissions in cropland sites in Europe (Bell et al., 2012; Smith et al., 2010b). However, it has
139 not previously been evaluated against a range of soils with varying organic content under
140 SRF plantations across GB.

141 This paper presents a field evaluation of ECOSSE and its suitability for estimating soil C
142 from British SRF soils after land-use change from conventional non-woody systems

143 (grassland with the exception of one field site which was arable). If measured and modelled
144 values are in agreement, the user can have more confidence that the model will correctly
145 simulate the processes. Evaluation of process-based models is often made difficult due to lack
146 of data from suitable study sites. The provision of data from eleven paired field sites in
147 Britain means that the mechanistic processes of ECOSSE can be evaluated thoroughly in this
148 study.

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163 **Materials and Methods**

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165 *ECOSSE model*

166 The ECOSSE model includes five pools of SOM, each decomposing with a specific rate
167 constant. Decomposition is sensitive to temperature, soil moisture and vegetation cover, and
168 so soil texture, pH, bulk density and clay content of the soil along with monthly climate and
169 land-use data are the inputs to the model (Coleman and Jenkinson, 1996, Smith et al., 1997).
170 The ECOSSE model simulates C and N cycle for four categories of vegetation: arable,
171 grassland, forestry and semi-natural. Short rotation forestry is commonly considered as
172 encompassing woody crops, therefore it is included in the forestry category of the model.

173 The soil input of the vegetation (SI) is estimated by a modification of the Miami model
174 (Lieth, 1972), which is a simple conceptual model that links the climatic net primary
175 production of biomass (NPP) to annual mean temperature (T) and total precipitation (P)
176 (Grieser et al., 2006). Separate estimates are obtained for NPP as a function of temperature
177 (NPPT) and precipitation (NPPP) according to empirical relationships, and the Miami
178 estimate of NPP is found as the minimum of these two estimates. In the present study NPP is
179 rescaled for each land cover type; for forest the rescaling factor is 7/8 of the Miami NPP
180 estimate (Del Grosso et al., 2008) and the SI is then estimated as a fixed proportion of the
181 NPP according to the land cover (value for forest is 0.15; Schulze et al., 2010). The linear
182 rescaling of the non-linear Miami functions is reasonable given the near-linear behaviour of
183 the Miami functions in the temperature and precipitation range of the UK.

184 For a full description of the ECOSSE model refer to Smith et al. (2010a).

185 The specific ECOSSE input requirements for large scale simulations are:

186 Climate/atmospheric data:

- 187 • 30 year average monthly rainfall, potential evapotranspiration (PET) and temperature,
- 188 • Monthly rainfall, temperature and potential evapotranspiration.

189 Soil data:

- 190 • Initial soil C content,
- 191 • Soil sand, silt and clay content,
- 192 • Soil bulk density,
- 193 • Soil pH.

194 Land-use data:

- 195 • Land-use for each simulation year.

196

197 The initialization of the model is based on the assumption that the soil column is at a stable
198 equilibrium under the initial land use at the start of the simulation. The model uses estimated
199 yearly plant inputs and measured initial soil C to estimate a soil turnover rate which would
200 maintain this equilibrium. Estimated plant inputs were calculated from a combination of the
201 net primary production (NPP) model MIAMI (Lieth, 1972; Lieth 1973) and land management
202 practices of the initial land use. The decomposition rate modifier, required to modify the
203 overall turnover rate, was estimated by numerically solving the analytical solution of the
204 decomposition equations (Bradbury et al., 1993). The solution was found using an iterative
205 method, using long term climate data, updating the decomposition rate modifier until the
206 system converges to a stable equilibrium and the change in soil carbon was zero. This method
207 produces relative carbon pool sizes of the decomposable plant material (DPM), resistant plant

208 material (RPM), microbial biomass (BIO) and humified organic matter (HUM), which along
209 with immobile soil C, is summed up to the measured soil C (Wong et al., 2013).

210

211 *Data*

212 In 2011/2012, eleven sites were sampled in Britain using a paired site comparison approach
213 (Keith et al., 2013). The sites and the relative measurements contribute to the ELUM
214 (Ecosystem Land Use Modelling & Soil Carbon GHG Flux Trial) project, which was
215 commissioned and funded by the Energy Technologies Institute (ETI). Each site consisted of
216 one reference field (arable or grassland, depending on the previous land-use of the SRF
217 fields) and one or more adjacent SRF fields, for a total of 29 transitions to SRF (Table 1). The
218 tree species included in the present study are: Alder (*Alnus incana* and *A. glutinosa*), Ash
219 (*Fraxinus excelsior*), Downy birch (*Betula pubescens*), Hybrid larch (*Larix x eurolepis*),
220 Poplar (*Populus* spp.), Scots pine (*Pinus Sylvestris*), Shining gum (*Eucalyptus nitens*), Cider
221 gum (*Eucalyptus gunni*), Silver birch (*Betula pendula*), Sitka spruce (*Picea sitchensis*), and
222 Sycamore (*Acer pseudoplatanus*). A full description of the sites can be found in Keith et al.
223 (2013). The change in soil C was assumed to be the difference in the forested and non-
224 forested pair.

225 Measurements of soil C, soil bulk density and soil pH, as well as information on the land-use
226 history, were collated for each field. A full description of the field sampling approach is
227 described in Keith et al. (2013). Briefly, for each field, fifteen soil cores to 30 cm depth were
228 taken using a split tube soil sampler with an inner diameter of 4.8 cm. A further, three deep
229 cores to 1 m were taken using a window sampler system with an inner diameter of 4.4 cm.
230 Samples were analysed for %C using a LECO Truespec CN analyser.

231 Air temperature and precipitation data at each location were extracted from the E-OBS
232 gridded dataset from the EU-FP6 project ENSEMBLES, provided by the ECA&D project
233 (Haylock et al., 2008). This dataset is known as E-OBS and is publicly available
234 (<http://eca.knmi.nl/>). For each location, monthly air temperature and precipitation for each
235 simulated year was collated and a long-term average was also calculated (Table 2). Monthly
236 potential evapotranspiration (PET) was estimated using the Thornthwaite method
237 (Thornthwaite, 1948), which has been used in other modelling studies when direct
238 observational data has not been available (e.g. Smith et al., 2005; Yokozawa et al. 2010; Bell
239 et al., 2012).

240 Soil texture data for the sites (Table 3) were extracted from the “Falloon” soil database (1 km
241 resolution) which is a collated soils dataset for England and Wales, Scotland and Northern
242 Ireland described in Bradley et al. (2005), and termed “Falloon” as it was first used to run
243 RothC in support of the Land-Use Change and Forestry (LULUCF) inventory (Falloon et al.,
244 2006).

245

246 *Model evaluation*

247 At each site, each transition from conventional crop (arable or grassland) to SRF was
248 modelled and the simulated soil C was compared to the measured soil C. Based on the site
249 information provided, the measured soil C at each reference arable/grassland site was used as
250 the starting C input to the model, assuming that the soil at the reference site had been in
251 equilibrium before the transition to SRF. All model parameters have been maintained
252 unvaried; therefore, the presented results are a test of the ability of the model to simulate soil
253 C under SRF as well as change in soil C from grassland/arable.

254 The model was evaluated using input data of measured soil C at the start of the simulation,
255 bulk density, and soil texture from the “Falloon” soil database. The simulations were done for
256 0-30 cm and 0-100 cm soil depth.

257 A quantitative statistical analysis was undertaken to determine the coincidence and
258 association between measured and modelled values, following methods described in Smith et
259 al. (1997) and Smith and Smith (2007). The statistical significance of the difference between
260 model outputs and experimental observations can be quantified if the standard error of the
261 measured values is known (Hastings et al. 2010). The standard errors (data not shown) and
262 95% confidence intervals around the mean measurements were calculated for all field sites.

263 The degree of association between modelled and measured values was determined using the
264 correlation coefficient (R). Values for R range from -1 to +1. Values close to -1 indicate a
265 negative correlation between simulations and measurements, values of 0 indicate no
266 correlation and values close to +1 indicate a positive correlation (Smith et al., 1996). The
267 significance of the association between simulations and measurements was assigned using a
268 Student’s t -test as outlined in Smith and Smith (2007).

269 The average size of the error was calculated as the root mean squared deviation (RMS) (Smith
270 et al., 2002). This is the average total difference between measured and modelled values and
271 is expressed in the same units as the analysed data. The lower the value of RMS , the more
272 accurate was the simulation.

273 The bias was expressed as a percentage using the relative error, E . The significance of the
274 bias was determined by comparing to the value of E that would be obtained at the 95%
275 confidence interval of the replicated values (E_{95}). If the relative error $E < E_{95}$, the model bias
276 cannot be reduced using these data.

277 Analysis of coincidence was undertaken to establish how different the measured and
278 modelled values were. The degree of coincidence between the modelled and measured values
279 was determined using the lack of fit statistic (*LOFIT*) and its significance was assessed using
280 an *F*-test (Whitmore, 1991) indicating whether the difference in the paired values of the two
281 data sets is significant. All statistical results were considered to be statistically significant at
282 $P < 0.05$.

283

284 **Results**

285 The model simulations of soil C showed a good fit against the measured soil C, for both
286 reference (Figure 1) and SRF fields (Figure 2), at 0-30 cm soil depth.

287 All the reference sites have been simulated for a time-period of ≥ 30 years without any land-
288 use change and using the field measurements as inputs to the model. Based on the site
289 histories, we assumed that all the reference sites were in equilibrium at the time of sampling.
290 The R value (1) of the reference sites at 0-30 cm soil depth showed a significant ($P < 0.05$)
291 association between modelled and measured values, as well as no significant model bias ($E <$
292 E_{95}).

293 Figure 2 shows the correlation between modelled and measured soil C at the SRF fields, at 0-
294 30 cm soil depth. Overall, the modelled soil C is highly correlated with the measured C
295 (Table 4). The R value (0.93) showed a significant ($P < 0.05$) association between modelled
296 and measured values.

297 The ECOSSE model simulates SRF as a single woodland vegetation type, but at all sites,
298 with the exception of Site 11, more than one SRF species was sampled. Therefore, for each
299 site, a single model simulation has been correlated to more than one measurement. To avoid
300 the lack of consistency between the number of model simulations and site measurements, the
301 results of each SRF species sampled at the same site have been averaged and the results of
302 the 0-30 cm soil depth presented in Figure 3.

303 At most of the sites, the modelled soil C at 0-30 cm soil depth was within the 95% confidence
304 interval of the measured soil C (error bars in Figure 3). At Site 1 and Site 4, the model
305 estimated a higher soil C content compared to the measured values (112.1 t C ha⁻¹ vs. 95.8 t C
306 ha⁻¹, 52.5 t C ha⁻¹ vs. 43.1 t C ha⁻¹, respectively), while for Site 10 the model simulated a
307 lower accumulation of C compared to the site measurements taken four years after

308 conversion from pasture (82.2 t C ha⁻¹ vs. 89.5 t C ha⁻¹). However, modelled soil C under
309 SRF showed a good fit against soil measurements, with an overall correlation value of $R =$
310 0.93 (Table 4).

311 The calculated statistical analysis of the model performance indicated that there is no
312 significant model bias ($E < E_{95}$) to simulate SRF and averaged SRF data. Similarly, the
313 *LOFIT* values showed that the model error was within (i.e. not significantly larger than) the
314 measurement error ($F < F$ (critical at 5%)).

315 The model simulations of the soil C at 0-100 cm soil depth again showed a good correlation
316 with the measured soil C, for both reference ($R = 0.99$, Figure 4) and SRF fields ($R = 0.82$,
317 Figure 5). Although the correlation between modelled and measured soil C at the SRF sites
318 was lower for the whole 100 cm soil profile compared to the 0-30 cm soil depth (Table 4), the
319 statistics of the soil C at the 0-100 cm soil depth reflected the good model performance found
320 for the top soil layer, with a high correlation between modelled and measured values and no
321 significant bias (Table 4).

322 The results of each SRF species sampled at the same site have been averaged and the results
323 are presented in Figure 6; the modelled and measured soil C at 0-100 cm soil depth followed
324 the same correlation among sites as for the 0-30 cm soil depth. The only exceptions are Site
325 5, Site 6, Site 9 and Site 11. The model underestimates the soil C at Site 5 and 9 by about 15-
326 20% of the measured values; whereas for Sites 6 and 11 the model overestimates the soil C at
327 0-100 soil depth by about 50% and 30%, compared to the measured values.

328 The change in soil C (ΔC) has been calculated as the difference between the soil C at the SRF
329 and the soil C at the reference site and the results are presented in Figures 7 and 8. These
330 results are important as they directly show the effect of the land-use transition itself. At 0-30
331 cm soil depth, the ΔC was within the 95% confidence intervals of the measured values

332 (Figure 7). Site 1 was the only site where the ΔC was not accurately simulated by the model.
333 At Site 1, the land-use change from arable has led to a decrease in soil C (16.3 t C ha^{-1}) after
334 8 years of land-use conversion to SRF; whereas, the results of the model simulations at Site 1
335 showed a small increase in soil C (0.6 t C ha^{-1}) after the transition.

336 Overall, at 0-100 cm, the ΔC simulated by the model followed the same direction of soil C
337 change as the simulated values (Fig. 8). The ΔC simulated by the model is within the 95%
338 confidence intervals of the measured values at four sites (Site 3, Site 7, Site 8 and Site 9;
339 Figure 8). The seven sites where the model did not match the measurements have all been
340 established recently (2004-2008).

341 Despite a lower correlation between modelled and measured soil C changes compared to the
342 soil C, the simulated changes in soil C are well associated with the measured values, with a
343 correlation factor of 0.66 and 0.72, at 0-30 cm and 0-100 cm soil depth respectively.
344 Furthermore, the statistical analysis on the ΔC showed no model bias ($E < E_{95}$) and a good
345 coincidence ($F < F$ (*critical at 5%*)) between modelled and measured changes in soil C after
346 transition to SRF (Table 4).

347

348 **Discussion**

349 The results of the present work revealed a strong correlation between modelled and measured
350 soil C and soil C changes to SRF plantations, at two soil depths (Table 4). Smith et al.
351 (2010a) presented an evaluation of the ECOSSE model to simulate soil C at national-scale,
352 using data from the National Soil Inventory of Scotland. This data set provided measurements
353 of soil C and soil C change for the range of soils, climates and land-use types found across
354 Scotland. The results of the present work are in agreement with the publication of Smith et al.
355 (2010a), which reported a high degree of association of the ECOSSE modelled values with
356 the measurements in both total C and change in C content in the soil.

357 As for the SRF plots, the soil C at the reference sites have been accurately simulated by the
358 model. The extremely high correlation for the reference fields shows a good performance of
359 the model spin-up. The spin-up is used by the model to reach a state of equilibrium under the
360 specified inputs. However, it is important to stress that it does not confirm that the reference
361 sites are in an equilibrium condition. Together, these results confirm the good performance of
362 the initialization method and the efficiency of the ECOSSE model in simulating soil C under
363 SRF.

364 Previous studies on ECOSSE have used large spatial datasets (Smith et al., 2010a,b) to
365 evaluate the model accuracy to simulate soil C. The present work is the first study to utilise
366 measured soil C at eleven different paired-sites in GB, to accurately test the ECOSSE model
367 performance in simulating soil C and soil C changes to SRF plantation. The statistical
368 analysis on results at both soil depths (0-30cm and 0-100cm soil depths) revealed no
369 significant error between modelled and measured soil C and soil C changes, as well as no
370 model bias, which suggests that the model cannot be further improved with the available data.

371 This is a promising result, given that this work is an independent evaluation of ECOSSE and
372 therefore, the model had not been further improved or parameterized to produce the outputs
373 presented in this paper.

374 Despite the good overall results, the analysis of the correlation between modelled and
375 measured soil C at specific sites showed that the model under/overestimated the measured
376 soil C at some of the SRF sites (Fig.3 and Fig.6). Since the change in soil C was determined
377 as the difference between the soil C at the SRF sites and the paired reference sites, such error
378 was also propagated in the soil C changes values (Fig. 7 and 8). This low correlation between
379 measured and modelled soil C is particularly manifested when comparing the soil C values of
380 the whole soil profile (0-100cm soil depth). One reason of the higher model inaccuracy at 0-
381 100cm compared to the 0-30 cm soil depth is the difference between the soil sampling
382 procedures. In fact, only three soil replicates were taken at one meter depth, which generated
383 a higher measurement uncertainty compared to data presented for the 0-30 cm soil depth (n=
384 15).

385 The young age of SRF plantations is also a factor that affected the simulation of the soil C.
386 The majority of transitions were less than 24 yrs old and four of the eleven sites were less
387 than 9 yrs old (e.g. Site 1, 4, 10 and 11). The decrease in the model accuracy to simulate the
388 soil C at some sites could therefore be caused by the imprecision of the processes described
389 in the model to capture the fast decrease in soil C that occurs during the first years of
390 cultivation. Similar issues to capture the decrease in soil C after afforestation were reported
391 for the parent model, RothC, by Romanyá et al. (2000). Romanyá et al. (2000) concluded that
392 the soil organic C that has become physically protected before land-use change loses its
393 protection from decomposition when the soil is converted to a new vegetation cover.

394 This process is not sufficiently described in the ECOSSE model, and could explain the loss in
395 soil C after land-use change measured at some experimental sites. It is important to notice
396 that at each sampled site, different SRF species have been sampled and this could have also
397 led to differences in soil C accumulation/depletion compared to the model simulations, which
398 in turn led to differences in soil C changes values. At Site 5, for example, the soil was
399 sampled on a Sitka spruce site together with two birch sites. The Sitka spruce site
400 accumulated an extremely high amount of soil C in 11 years, especially at the 30-100 cm soil
401 depth (122 t C ha^{-1}), but such high C content in deep soil layers was not captured by the
402 model. Previous studies on the effect of conversion from pasture to forest on soil C have
403 shown contrasting results on the direction and rate of change in soil C after land-use change
404 (Guo and Gifford, 2002; Poeplau et al., 2011; Poeaplau and Don, 2013). A meta analysis on
405 the influence of land use change on soil C concluded that when established pastures switch to
406 forest, soil C stocks decline under pine plantation, but are unaffected by broadleaf plantations
407 and that the time since conversion occurred influences the soil C stocks (Guo and Gifford,
408 2002). A recent review of 95 studies on the dynamics of soil C after land use change in
409 temperate zone (Poeplau et al., 2011) reported that the cultivation of grassland or forest
410 caused rapid soil C losses and the accumulation of soil C was a slow and continuous process
411 after establishment of grassland and afforestation of cropland. Finally, Poeaplau and Don
412 (2013) used a paired site approach on selected sites across Europe to measure changes in soil
413 C after different land use change types. In particular, they found a significant accumulation of
414 soil C after conversion of cropland to forest and no significant effect on the soil C converting
415 grassland to forest.

416 Another common source of error when studying soil C, and particularly soil C changes after
417 transition to a new vegetation system, is the selection of paired sites. Inexact pairing is a
418 frequent source of discrepancy, which is mainly due to the lack of information on the land-

419 use history of fields (Goidts et al., 2009). In our study, 29 transitions have been simulated
420 based on extended information on the selected sites. The only improper pair was found at
421 Site 6. At this site the reference field was an arable crop, which was converted to pasture in
422 1994. The pasture site was sampled as a reference site, but was planted at the same time as
423 the SRFs (1994-1996), therefore it is not a good reference for this site. In fact, the
424 measurements showed a lower soil C under the SRFs compared to the reference site, while
425 the model predicted around the same C content at the two paired sites.

426 In the present study, a range of SRF species has been modelled, including Eucalyptus (Site 1
427 and 4). However, the results of the modelled soil C did not agree with the measured values at
428 either Eucalyptus sites or at either soil depth. In addition, at site 1, the establishment of
429 Eucalyptus species involved the use of strip plastic mulch mats for weed suppression, which
430 may have led to a reduction in volume of leaf litter material being incorporated into the
431 humic soil horizon. There is very little research from Europe and GB on Eucalyptus litter
432 and soil chemistry effects (Hardcastle, 2006). It has however been reported that the various
433 species of Eucalyptus have widely different canopy density and potential growth rate (Pryor,
434 1976), which affect the soil C behaviour under this SRF species. The ECOSSE model has
435 previously been parameterized for forest as a land use category (Smith et al, 2010a), but no
436 parameterization have been made for exotic species such as Eucalyptus. It is therefore likely
437 that the model does not describe the soil C behaviour under Eucalyptus as well as under the
438 other SRF species reported in the present work. Further model developments are therefore
439 needed to include this vegetation type in the model parameters.

440 This paper reinforces previous studies on the ability of ECOSSE to simulate soil C and N and
441 test its accuracy to simulate changes in soil C after land-use change to SRF. The use of this
442 process-based model is an improvement on empirical models, with simulations of aggregate
443 monthly data producing high degrees of association with measured data. With further

444 modification to capture the decrease in soil C which often occurs in the early stage of a new
445 transition and with better parameterisation for Eucalyptus and coniferous species, ECOSSE
446 would be expected to be a very useful tool for quantitatively predicting the impacts of future
447 land-use on soil C, GHG emissions and climate change.

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636 **Figure legends**

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638 **Figure 1:** Correlation between measured and modelled soil C at the reference sites at 0-30
639 cm soil depth. Error bars represent 95% confidence interval of measured values. Dotted line
640 represents 1:1 correlation between measured and modelled values.

641 **Figure 2:** Comparison between modelled and measured soil C at the SRF sites at 0-30 cm
642 soil depth. Error bars represent 95% confidence interval of measured values. Dotted line
643 represents 1:1 correlation between measured and modelled values. SRF species are
644 represented by different colours.

645 **Figure 3:** Modelled and measured soil C at the study sites (0-30 cm soil depth). Results are
646 averaged soil C values for the SRF fields at each site. Error bars represent 95% confidence
647 interval of measured values.

648 **Figure 4:** Comparison between measured and modelled soil C at the reference sites at 0-100
649 cm soil depth. Error bars represent 95% confidence interval of measured values. Dotted line
650 represents 1:1 correlation between measured and modelled values.

651 **Figure 5:** Comparison between modelled and measured soil C (0-100 cm soil depth) at the
652 SRF sites. Error bars represent 95% confidence interval of measured values. Dotted line
653 represents 1:1 correlation between measured and modelled values. SRF species are
654 represented by different colours.

655 **Figure 6:** Modelled and measured soil C at the study sites (0-100 cm soil depth). Results are
656 averaged soil C values for the SRF fields at each site. Error bars represent 95% confidence
657 interval of measured values.

658 **Figure 7:** Measured and modelled change in soil C at 0-30 cm soil depth. Results are
659 averaged change in soil C values between the SRF fields at each site. Error bars represent
660 95% confidence interval of measured values.

661 **Figure 8:** Measured and modelled change in soil C at 0-100 cm soil depth. Results are
662 averaged change in soil C values between the SRF fields at each site. Error bars represent
663 95% confidence interval of measured values.

Tables

Table 1

Site no.	Transition unit (previous land use in bold)	Duration of the SRF stands since transition to year of sampling (years)	Latitude , Longitude
1	Arable		
	Eucalyptus Gunnii	8	55.2, -1.5
Eucalyptus Nitens	8		
2	Pasture		
	Hybrid Larch	23	52.0, -3.6
Sycamore	23		
3	Rough Pasture		
	Alder	56	54.3, -0.5
	Scots pine	58	
	Silver birch	56	
Beech	56		
4	Rough Pasture		
	Eucalyptus Gunnii	6	53.34, -1.0
Eucalyptus Nitens	6		
5	Rough Pasture		
	Downy Birch	13	57.6, -3.2
	Silver Birch	13	
Sitka spruce	12		
6	Pasture		
	Poplar	17	57.7, -3.3
	Alder	15	
Ash	15		
7	Rough Pasture		
	Alder	55	54.0, -2.4
	Scots pine	55	
Sitka spruce	20		
8	Pasture		
	Sycamore	23	56.9, -2.6
	Scots pine	23	
Hybrid Larch	23		
9	Pasture		
	Alder	21	55.8, -3.6
	Poplar	21	
Sitka spruce	21		
10	Pasture		
	Ash	4	54.7, -2.8
	Sycamore	4	
Alder	4		
11	Rough Pasture		
	Scots pine	4	56.1, 3.6

Table 1: Details of vegetation type, duration of the SRF stands since transition and location of the study sites.

Table 2

Rainfall (mm/month)											
Month	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Site 8	Site 9	Site 10	Site 11
January	52.6	134.5	61.2	48.3	52.0	57.1	142.7	70.2	126.0	138.9	102.7
February	44.3	104.7	47.8	37.3	51.1	53.8	102.9	61.5	96.9	98.7	72.6
March	48.4	96.5	48.6	40.6	45.9	45.3	107.8	54.5	85.2	101.1	74.2
April	47.2	82.1	47.9	45.4	44.9	47.7	82.9	54.2	61.8	68.3	52.6
May	46.1	75.7	49.3	45.2	49.1	51.3	81.3	53.7	61.8	69.4	60.9
June	58.4	75.4	55.9	60.3	55.5	57.2	87.4	58.2	67.0	72.6	60.2
July	59.3	96.4	58.5	46.6	57.2	63.0	96.6	60.6	76.6	83.8	66.6
August	62.6	97.9	68.0	53.0	62.9	63.7	117.0	66.8	86.2	94.9	76.9
September	58.1	95.3	59.4	49.2	61.9	68.2	120.3	62.7	85.2	101.2	84.4
October	62.4	144.9	60.7	55.9	79.6	80.7	141.2	97.7	121.5	134.5	100.1
November	69.0	141.8	69.5	52.6	65.8	72.0	142.6	84.4	113.0	136.0	93.8
December	58.5	138.5	64.7	52.0	55.4	58.9	150.5	67.5	112.2	138.1	91.1

Temperature (C°/month)											
Month	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6	Site 7	Site 8	Site 9	Site 10	Site 11
January	6.6	3.9	2.9	4.1	3.6	3.3	2.2	2.9	3.4	2.3	2.9
February	7.0	4.1	3.0	4.4	3.8	3.5	2.3	3.1	3.9	2.6	3.13
March	9.2	5.5	4.8	6.5	5.2	4.9	4.0	4.5	5.5	4.1	4.88
April	11.5	7.3	6.9	8.6	7.3	7.3	6.3	6.4	7.8	6.3	7.16
May	14.2	10.5	9.9	11.6	9.7	9.6	9.3	9.0	10.5	9.4	9.9
June	17.0	12.8	12.8	14.5	12.3	12.3	12.1	11.8	13.0	12.0	12.8
July	19.4	14.7	14.8	16.7	14.3	14.3	13.8	13.7	14.7	14.0	14.4
August	19.2	14.7	14.9	16.5	14.1	14.1	13.6	13.5	14.6	13.6	14.2
September	16.7	12.6	12.9	14.1	12.0	12.1	11.6	11.4	12.3	11.3	11.9
October	12.9	9.7	9.7	10.6	9.0	9.0	8.6	8.2	9.0	8.3	8.9
November	9.2	6.5	5.8	6.9	5.8	5.8	5.0	5.0	5.9	5.0	5.3
December	6.9	4.1	3.7	4.4	3.2	2.9	2.9	2.6	3.0	2.8	3.2

Table 2: Long-term (30 years) monthly rainfall and temperature at the location of the study sites.

Table 3

Site	Reference field	0-30 cm soil depth					0-100 cm soil depth				
		Soil C (t C ha ⁻¹)	Bulk density (g/cm ³)	Clay (%)*	Silt (%)*	Sand (%)*	Soil C (t C ha ⁻¹)	Bulk density (g/cm ³)	Clay (%)*	Silt (%)*	Sand (%)*
1	Arable	112.0	1.3	23	33	44	151.9	1.3	39	33	29
2	Pasture	76.2	0.9	23	49	29	81.0	1.0	23	51	26
3	Rough Pasture	101.4	0.6	6	29	64	115.3	1.1	4	25	71
4	Rough Pasture	54.0	1.2	8	17	75	64.5	1.4	4	9	87
5	Rough Pasture	94.6	0.8	10	24	66	169.6	1.0	10	24	66
6	Pasture	39.3	1.1	8	22	70	58.0	1.2	6	15	79
7	Rough Pasture	117.2	0.7	23	33	44	239.6	1.2	23	36	42
8	Pasture	80.7	0.7	9	33	58	90.6	0.9	8	29	62
9	Pasture	122.9	1.0	20	27	52	285.5	1.2	25	29	46
10	Pasture	83.0	1.0	19	30	51	164.8	1.0	29	32	39
11	Rough Pasture	83.2	1.2	5	56	39	123.9	1.2	5	58	37

Table 3: Measured soil C, measured bulk density, percentage of clay, silt and sand at 0-30 cm and 0-100 cm soil depth for the reference fields.

* Data extracted from “Falloon” soil database.

Table 4

		<i>R</i>	<i>t</i> value	<i>t</i> value at P = 0.05	<i>E</i>	<i>E</i> (95% Confidence Limit)	<i>F</i> value	<i>F</i> value (Critical at 5%)
0-30 cm	Reference	1.00	52.02	2.26	0	24	0.00	2.03
	SRF	0.93	13.48	2.05	-4	27	0.00	1.55
	Averaged SRF	0.96	10.58	2.26	-4	16	0.00	2.03
	Averaged ΔC	0.66	2.61	2.26	93	-2003	0.18	2.03
0-100 cm	Reference	0.99	17.84	2.26	0	58	0.00	2.03
	SRF	0.82	7.23	2.06	-3	72	0.01	1.56
	Averaged SRF	0.87	5.39	2.26	-13	52	0.02	2.03
	Averaged ΔC	0.72	3.15	2.26	91	-1068	0.07	2.03

Table 4: ECOSSE model performance at simulating soil C and soil C changes (ΔC) at the reference, SRF and averaged SRF fields for two soil depths (0-30 cm and 0-100 cm). Averaged SRF represents statistical analysis on averaged soil C values of the SRF fields at each site. Averaged ΔC represents averaged change in soil C of the SRF fields at each site. Association is significant for $t > t$ (at P=0.05). Model bias is not significant for $E < E_{95}$. Error between measured and modelled values is not significant for $F < F$ (critical at 5%).

Figure 1

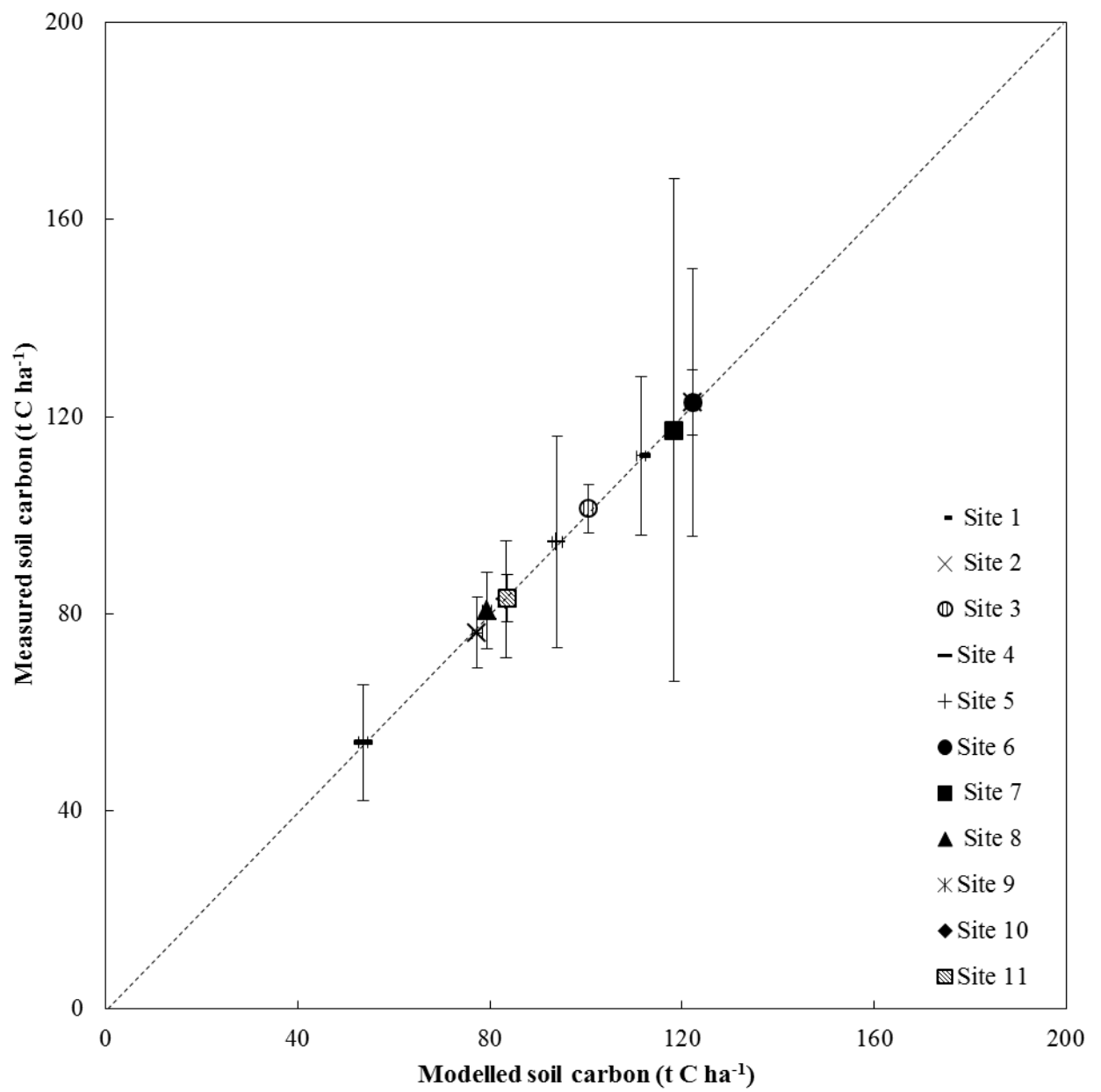


Figure 2

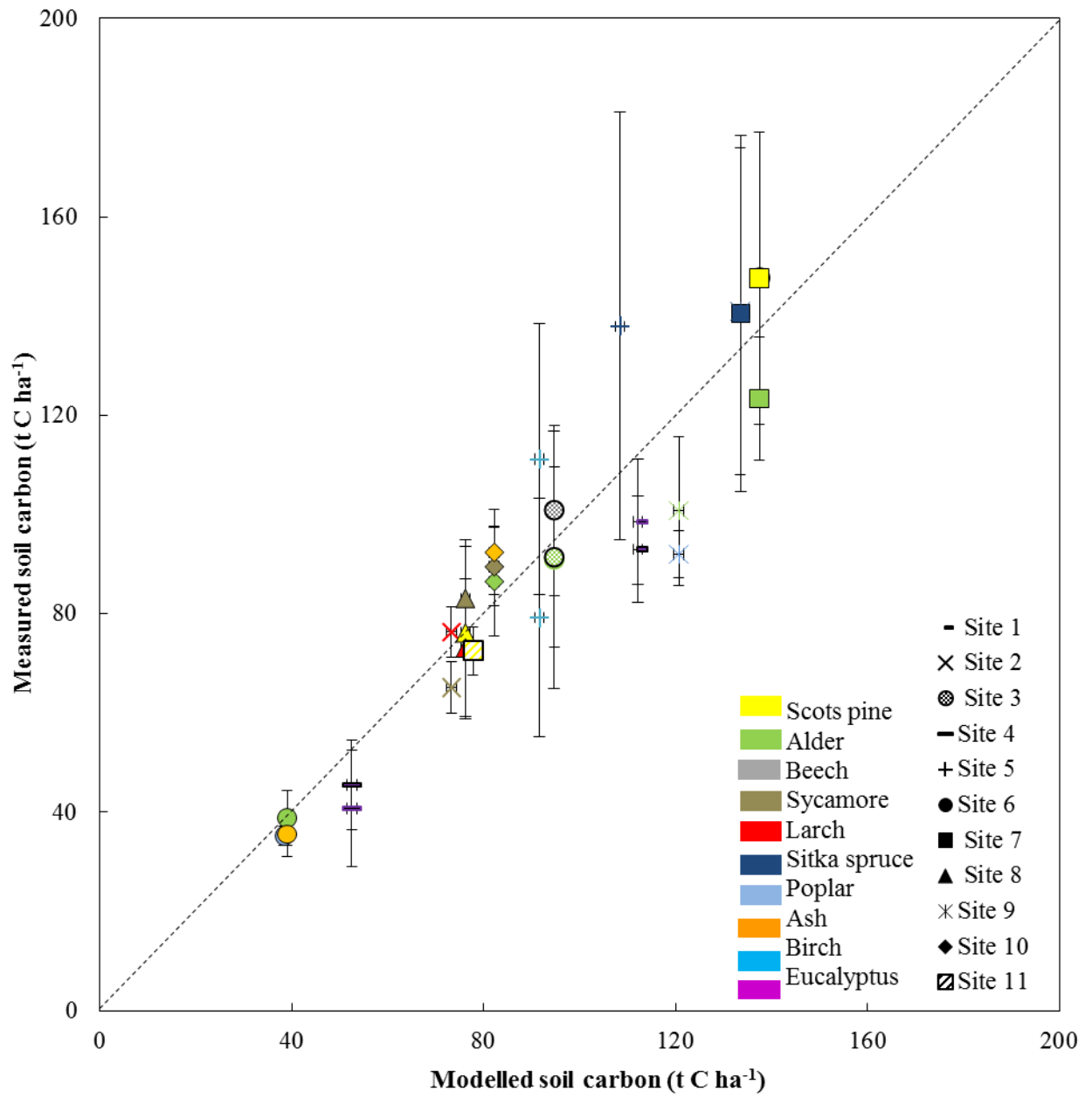


Figure 3

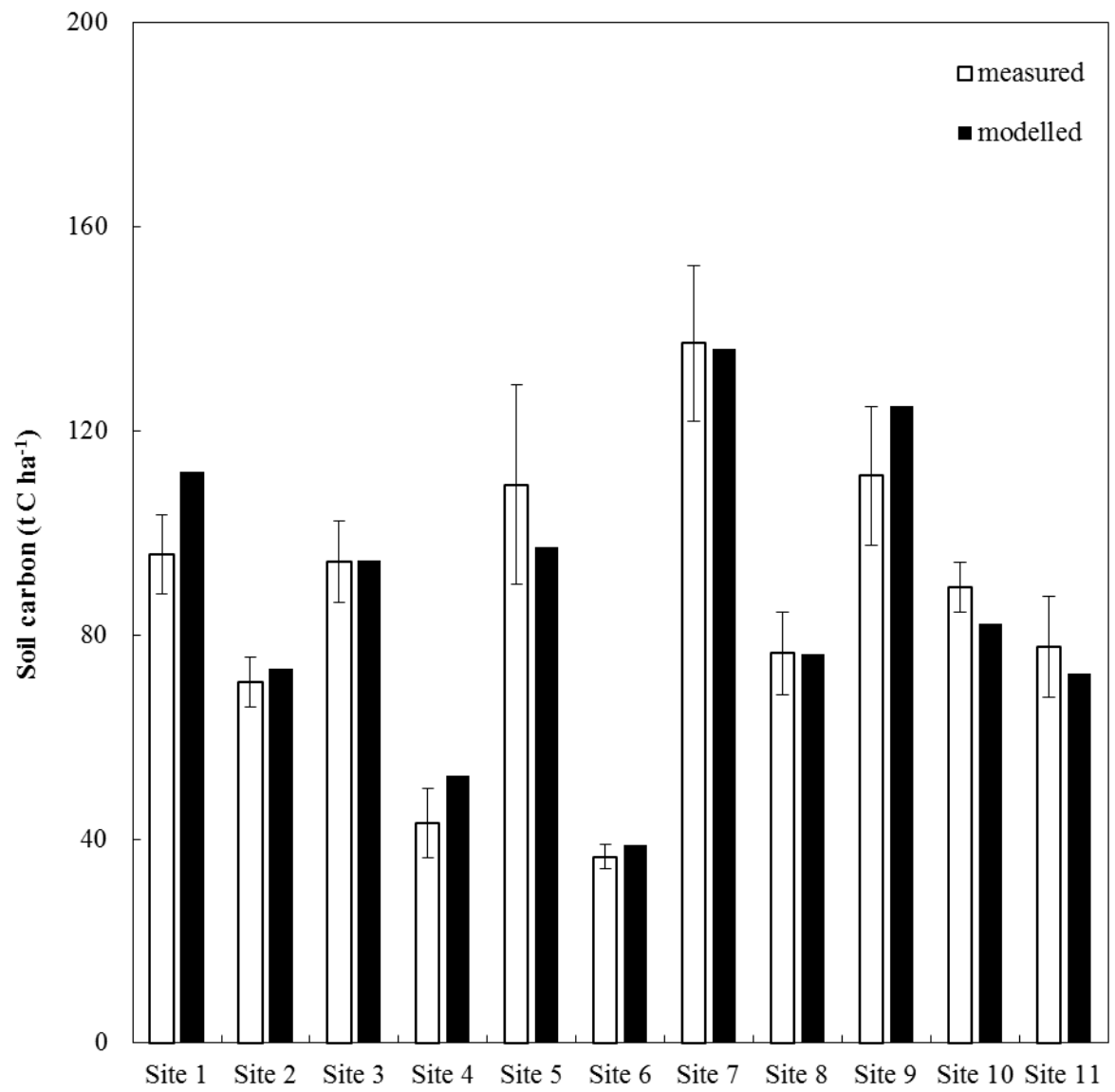


Figure 4

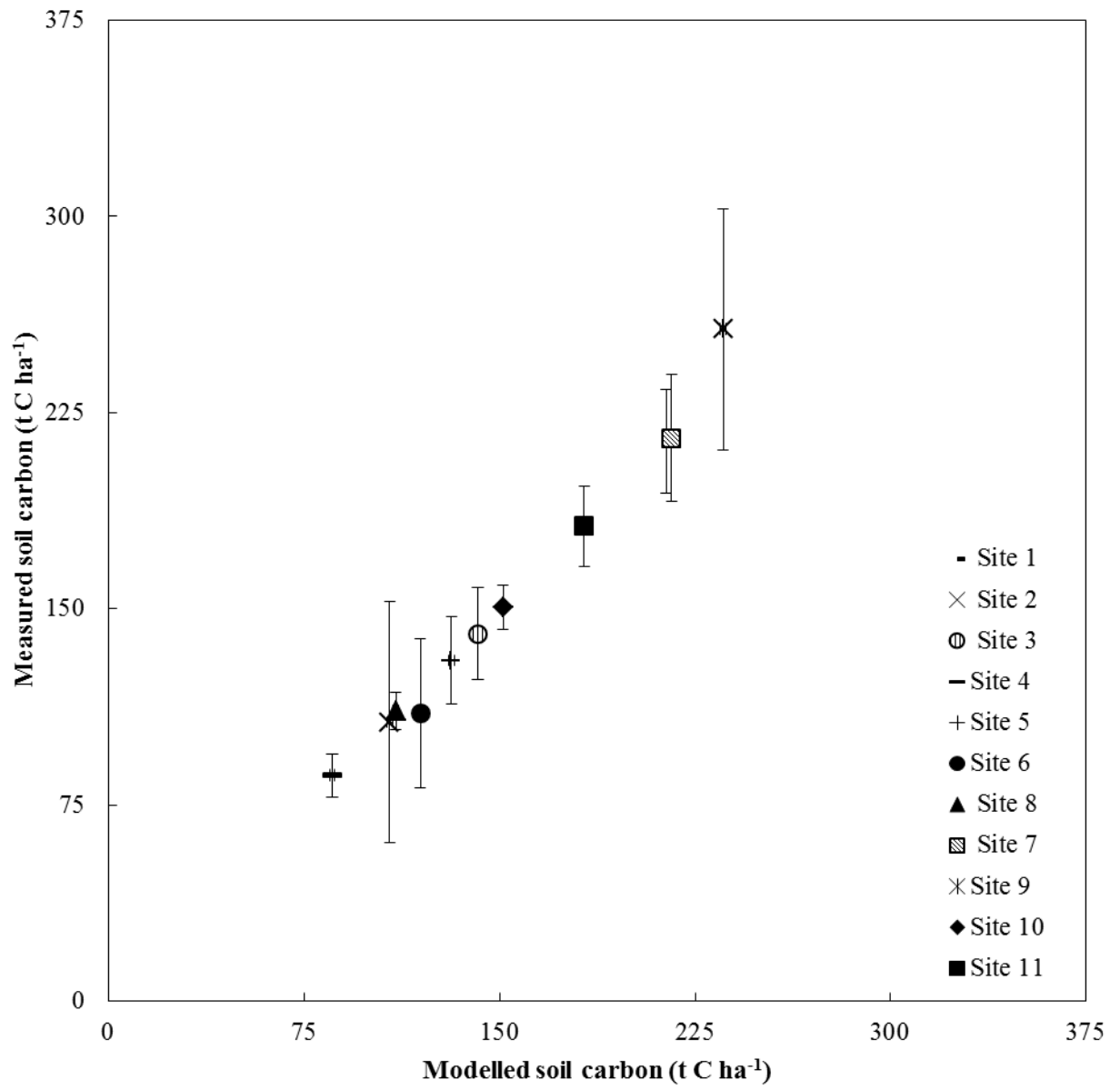


Figure 5

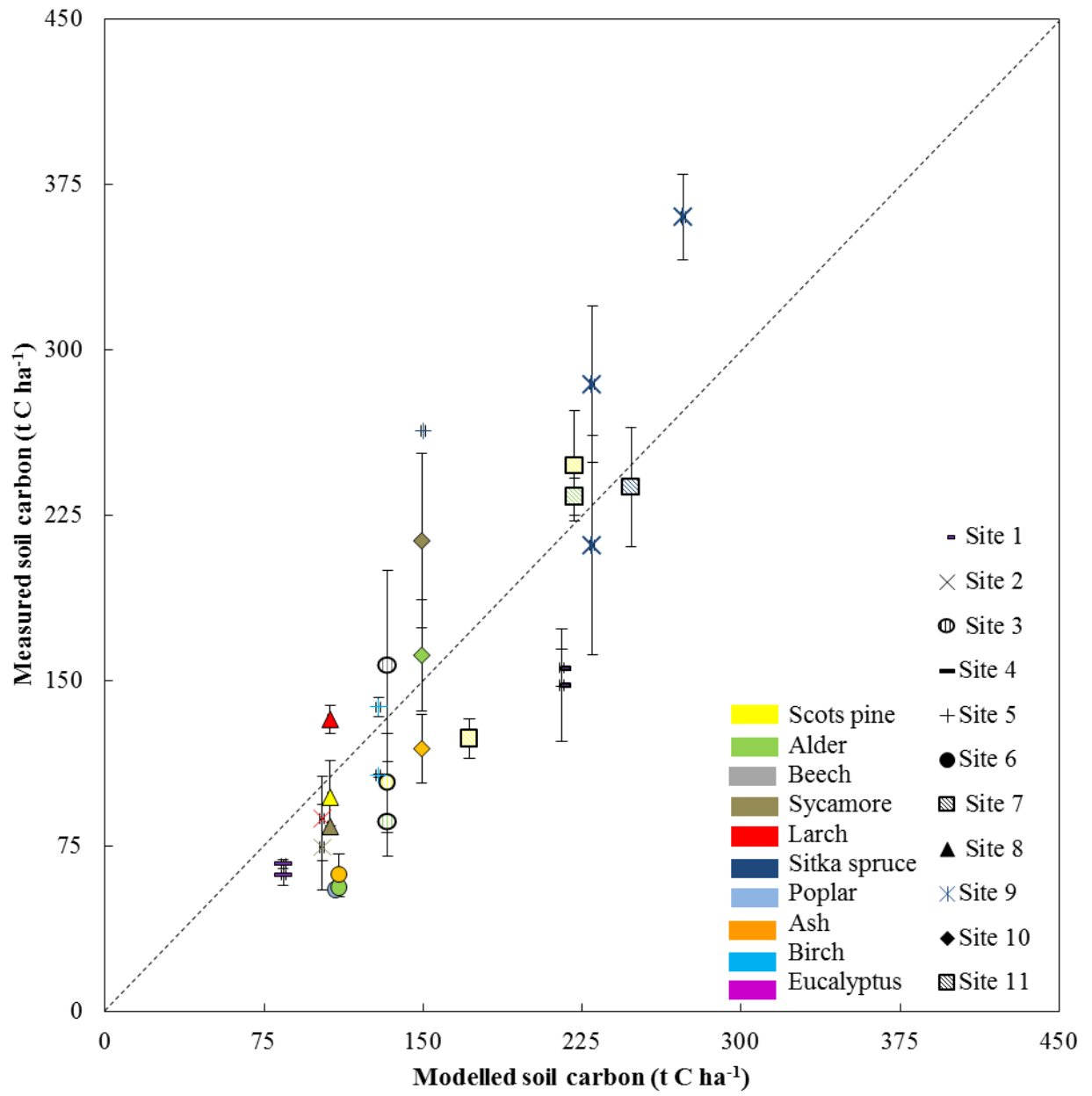


Figure 6

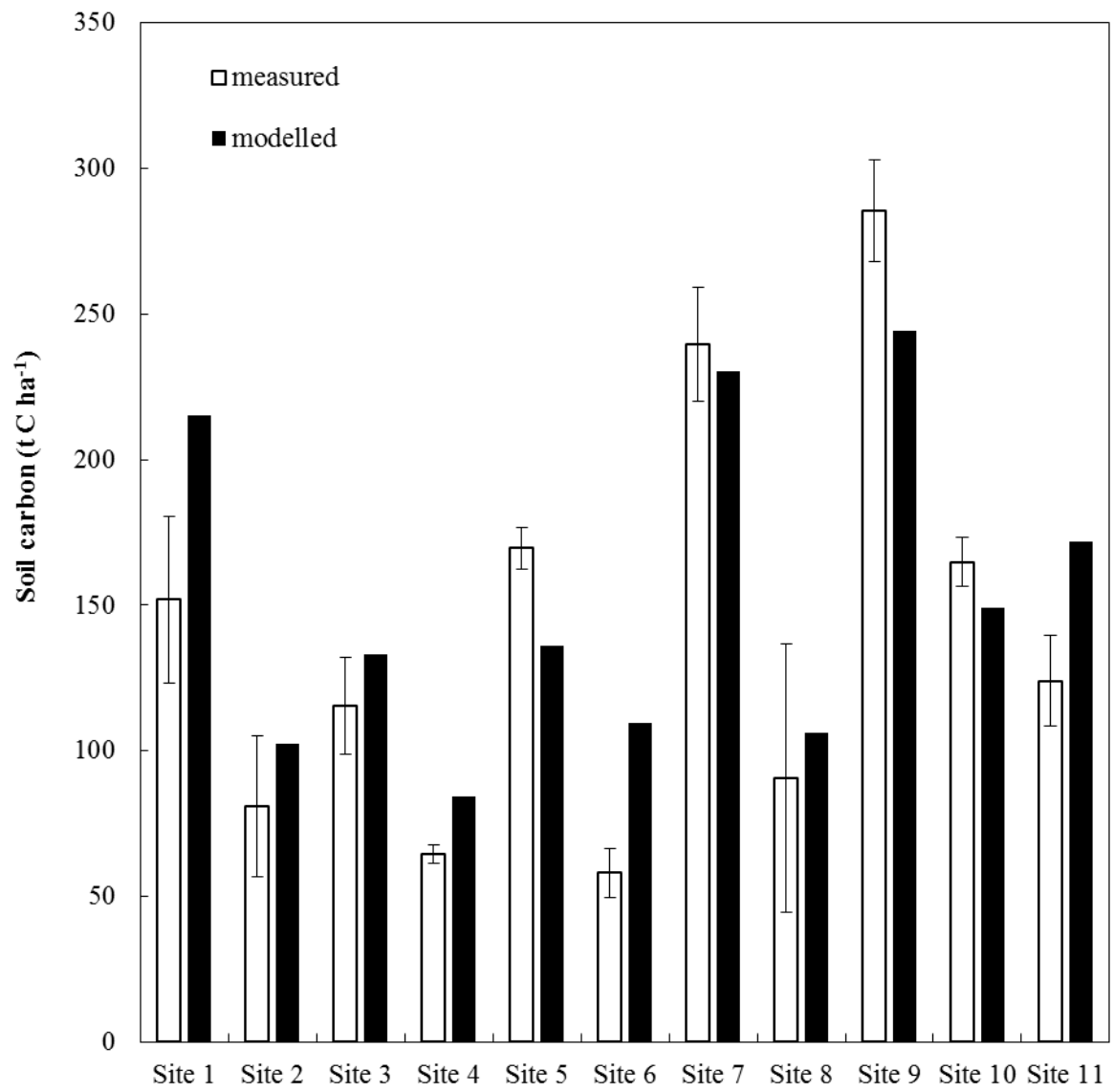


Figure 7

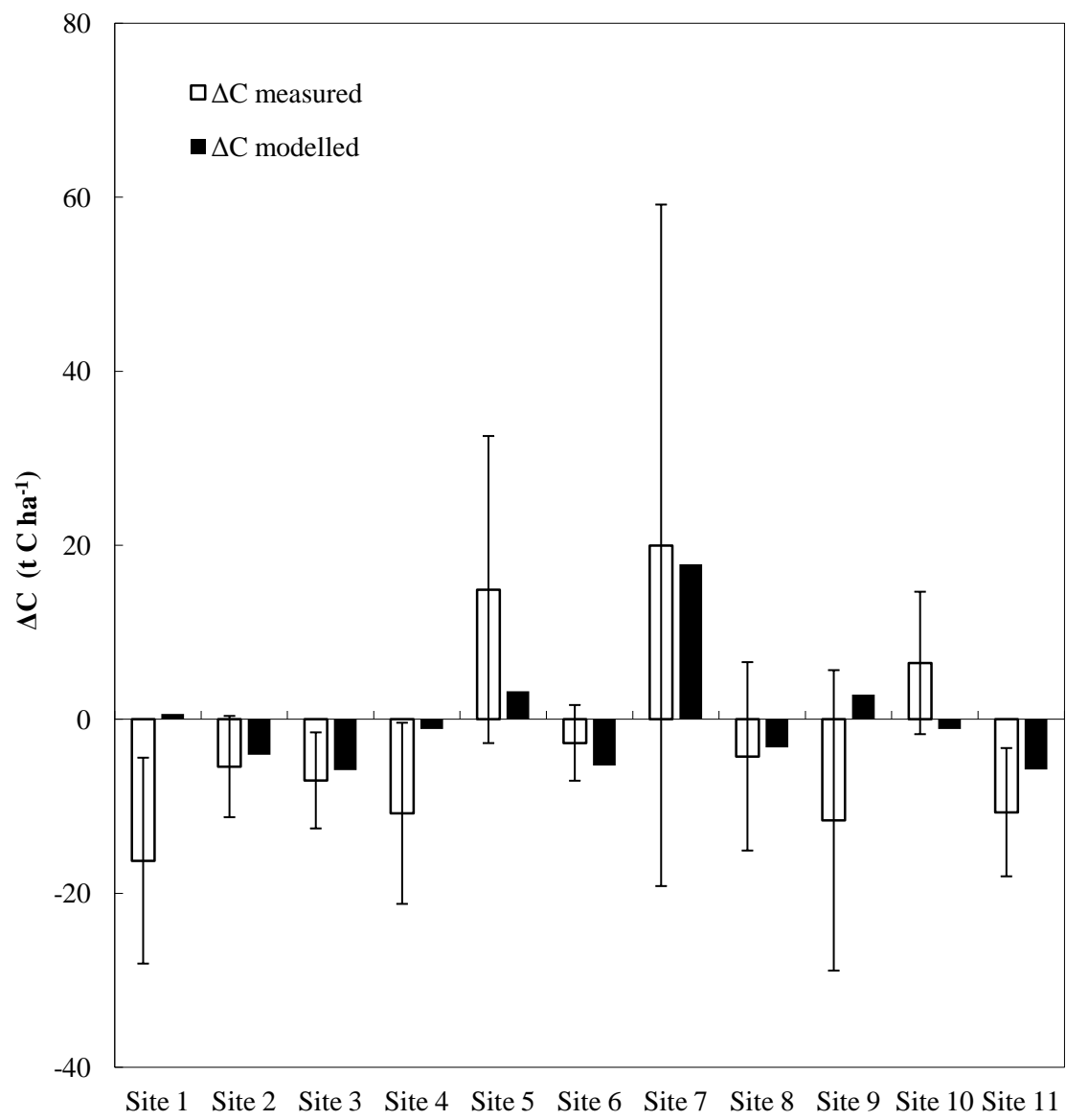


Figure 8

