

1 **Modelling biodiversity distribution in agricultural landscapes to support ecological**
2 **network planning**

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22 **Abstract**

23 Strategic approaches to biodiversity conservation increasingly emphasise the restoration of
24 ecological connectivity at landscape scales. However, understanding where these connecting
25 elements should be placed in the landscape is critical if they are to provide both value for
26 money and for biodiversity. For such planning to be effective, it is necessary to have
27 information of the distributions of multiple taxa, however, this is of poor quality for many
28 taxa. We show that sparse, non-systematically collected biological records can be modelled
29 using readily available environmental variables to meaningfully predict potential biodiversity
30 richness, including rare and threatened species, across a landscape. Using a large database of
31 ad-hoc biological records (50 501 records of 502 species) we modelled the richness of
32 wetland biodiversity across the Fens, a formerly extensive wetland, now agricultural
33 landscape in eastern England. We used these models to predict those parts of the agricultural
34 ditch network of greatest potential conservation value and compared this to current strategic
35 network planning. Odonata distribution differed to that of other groups, indicating that single
36 taxon groups may not be effective proxies for other priority biodiversity. Our results
37 challenged previous assumptions that river channels should comprise the main connecting
38 elements in the Fens region. Rather, areas of high ditch density close to a main river are
39 likely to be of greater value and should be targeted for enhancement. This approach can be
40 adopted elsewhere in order to improve the evidence-base for strategic networks plans,
41 increasing their value for money.

42

43 **HIGHLIGHTS**

- 44 • We used ad-hoc biological data to model landscape-scale wetland species richness.
- 45 • Models were used to assess and improve a proposed ecological connectivity network.
- 46 • Our evidence-based network was shorter and connected areas of higher richness.
- 47 • Our results challenge previous assumptions of important network elements.
- 48 • Odonata were poor proxies for other groups of wetland species.

49

50 **1. Introduction**

51

52 Habitats are increasingly fragmented. Furthermore, in human landscapes, habitat patches are
53 often surrounded by land uses that are potentially hostile to dispersal, increasing functional
54 isolation (Nowicki et al., 2014). Such habitat fragmentation and isolation increase local
55 population vulnerability to extinction and reduced dispersal opportunities limit species'

56 ability to respond to climate change, further reducing biodiversity resilience (Hill et al.,
57 2002). Strategic approaches to conservation are, therefore, increasingly focused at the
58 restoration of landscape connectivity by the creation of movement corridors, stepping stones
59 or by improving landscape permeability (Dolman, 2012; Lawson et al., 2012; Saura et al.,
60 2014). However, the nature, size and placement of these connecting elements are critical if
61 investment of finite funds and land resources are to give optimal returns. There are several
62 key issues to the success of landscape connectivity; identifying what species should be
63 targeted within a landscape (Dolman et al., 2012), ensuring that the connectivity elements
64 comprise habitats that suit these species and establishing where these connecting elements
65 should be placed.

66

67 Ecological networks are often designed to enhance the metapopulation viability of individual
68 high profile species, such as top predators (Klar et al., 2012) or other mobile species (Bani et
69 al., 2002). However, the ability of such species to act as connectivity umbrellas for
70 assemblages of other species may be limited (Cushman & Landguth, 2012) because the
71 suitability of the habitat and type of connecting element differs amongst taxa. For example,
72 while linear field margins may provide connectivity to some generalist butterflies (Delattre et
73 al., 2010), they may act as sinks to other taxa (Krewenka et al., 2011). Similarly, hedgerows
74 are often purported to provide suitable corridors for woodland species, but may only provide
75 habitat for woodland edge species (Liira & Paal, 2013). The planning of landscapes to
76 provide resilience for assemblages of regional biodiversity therefore requires the
77 consideration of multiple relevant taxa (Zulka et al., 2014).

78

79 Decisions regarding the optimum placement of connecting elements should be made using
80 evidence of the current and potential distribution of a full complement of target species.
81 Existing protected sites that retain a concentration of rare species generally form the focus of
82 connectivity networks (Beier et al., 2011) and the existence of species within these fragments
83 is often well known. However, our understanding of the distribution of species throughout the
84 rest of the landscape is incomplete, with some locations receiving high levels of recording
85 effort and others very little. Poorly recorded areas that are nevertheless potentially suitable
86 for a species may harbour unrecorded residual populations, or be more likely to be colonised
87 if both habitat quality and connectivity are improved (Lawson et al., 2014). Unsystematically
88 collected biological data therefore do not provide a reliable assessment of conservation value
89 or potential across a landscape. This results in reliance on expert opinion in the design of

90 landscape connectivity (Beier et al., 2009; Eycott et al., 2011). However, if the patchy nature
91 of recording effort is accounted for in the analysis (Kéry, 2011), ad hoc biological data can be
92 exploited to provide more objective design of landscape connectivity.

93

94 In this study, we use the Fens, a formerly extensive wetland system in eastern England, to
95 demonstrate how connectivity planning can be informed by modelling ad hoc biological
96 records with easily obtainable, landscape-scale environmental data. Remaining wetland
97 habitat in the Fens is highly fragmented and isolated within an intensive agricultural
98 landscape, but there is a high potential for connectivity through enhancing management of
99 linear drainage ditches. Ditches in intense agricultural areas are often rather different to
100 natural streams (Herzon & Helenius, 2008), supporting lower biodiversity (Williams et al.,
101 2004); however, they can act as reservoirs for important regional wetland biodiversity (Simon
102 & Travis, 2011). Biological recording within the wider Fens landscape is extremely sparse, so
103 simple mapped biological richness cannot be used as an evidence base for selecting
104 potentially biodiverse ditches for improved management or in the design of connectivity
105 networks. Recent attempts at strategic planning (e.g. Fens for the Future Partnership, 2012)
106 have therefore relied on a combination of expert opinion and untested assumptions of where
107 this targeted management should be placed.

108

109 We take the approach of modelling potential biodiversity value in relation to underlying
110 environmental factors and landscape context, to predict where in the landscape targeted
111 management to enhance habitat quality will have greatest potential to support biodiversity
112 and enhance connectivity. We use an extensive but unevenly distributed database of 67,395
113 ad hoc biological records to model the richness of groups of wetland species across the Fens
114 landscape in relation to a range of coarse-scale environmental and landscape factors. Using
115 these models, we aim to: 1) predict and map the potential richness of groups of wetland
116 species in order to identify parts of the landscape of greatest potential conservation value; 2)
117 apply these maps of predicted biodiversity potential to assess current strategic planning maps.

118

119 **2. Methodology**

120 *2.1 Study area*

121 The Fens, covering almost 4,000 km² of eastern England (Fig. 1), was formerly an extensive
122 wetland area but only 1% of wetland habitat remains. This habitat is concentrated in six key
123 protected areas, which are each small (mean area 819 ha) and isolated within the country's

124 most important arable agricultural landscape. More than 20 million km of ditches and
125 drainage channels criss-cross the Fens landscape and by targeting selected ditches for
126 enhanced management, the ditch network presents an excellent opportunity for increasing
127 both habitat area and connectivity for wetland species. However, the current conservation
128 value of large parts of this landscape is poorly known.

129

130 *2.2 Biological data*

131 All available species observations (records) were collated for the period of 1987-2012 from
132 the 4147 1-km squares wholly or partly within the Fens Natural Character Area boundary
133 (Natural England, 2013), with an extension (3 km from the boundary) to include Chippenham
134 Fen, one of the three important relict fen sites in the Fens. Records were compiled from Local
135 Biological Records Centres, the National Biodiversity Network (NBN) gateway, national and
136 county natural history and recording societies whose records were not available via NBN, and
137 unpublished documents or reports. Records sent to Biological Records Centres and societies
138 are validated by expert county recorders. Although NBN data may include some unvalidated
139 records submitted by the public, our collated species lists were validated by a range of local
140 taxonomic experts. Records were managed using RECORDER 6 software (Joint Committee
141 for Nature Conservation, Peterborough, UK). The study period (1987-2012 inclusive) was
142 selected as a compromise between reflecting the current or recent distribution of wetland
143 species and including sufficient records in the dataset to capture rare species and the potential
144 distribution of sparsely recorded taxonomic groups. There may have been local extinctions
145 since 1987 due to local changes in habitat quality, nevertheless the landscape predictors we
146 consider will indicate the biodiversity potential should habitat and connectivity be restored.
147 The majority (74%) of records were resolved to a spatial resolution of 1 km or finer and these
148 were aggregated and analysed at the scale of 1-km grid cells. Tetrad records were assigned to
149 all of the four 1-km squares comprising the tetrad; species records at coarser spatial
150 resolutions were excluded. A small number of records of taxa not recorded to species level
151 were removed. Records of marine species were excluded, but those tolerant of brackish
152 conditions were retained. Following additional filters described below (e.g. removal of
153 coastal squares), a database of 255 291 records remained, of which 50 501 were records of
154 wetland plants (including conservation priority species) and conservation priority wetland
155 invertebrates. Conservation priority species were recognised as those designated as UK
156 Biodiversity Action Plan, Global and UK Red Data Book (except Least Concern), Nationally
157 Rare, Nationally Scarce or Nationally Notable A and B, according to JNCC (2012), plus

158 undesigned species with >25% of their UK distribution occurring in the Fens region –
159 hereafter referred to as ‘Fens Specialists’.

160

161 The richness of groups of wetland species were used as the biological response variables.
162 Seven widely recorded groups of wetland species were selected for modelling that were
163 considered good indicators of ditch quality: all Odonata (dragonflies and damselflies, 28
164 species), wetland plants (212 taxa), fully aquatic plants (137 taxa) and conservation priority
165 species (including plants and invertebrates) dependent on aquatic (fully aquatic and
166 submerged aquatic habitats, 90 species), littoral (aquatic margins, 109 species) or wetland
167 (208 species) habitats, and Fen Specialists (58 species). Wetland plant species were defined
168 as all Characeae (stoneworts, multi-cellular branched macro-algae) and those vascular plant
169 species associated with freshwater (aquatic, wetland or seasonally wet) habitats selected from
170 Hill et al. (2004) with Ellenberg moisture values ≥ 7 (species with Ellenberg salinity values
171 of >5 were excluded). Aquatic plants were a sub-set of the wetland plants, classified with
172 reference to existing lists by Palmer et al. (2013) and Mountford and Arnold (2006). The
173 autoecological requirements of conservation priority species and their association with
174 wetland, aquatic and littoral habitats, were classified following Mossman et al. (2012) and
175 Dolman et al. (2012).

176

177 *2.3 Environmental predictors*

178 The aim of this analysis was to predict the distribution of wetland species across the drainage
179 ditch network of the arable landscape based on readily-available, coarse-scale environmental
180 variables. Wetland Sites of Special Scientific Interest (SSSIs) were considered to be
181 reservoirs and potential sources of high quality biodiversity, therefore 1-km squares including
182 any part of a wetland SSSI were excluded from modelling. Wetland SSSIs were identified
183 based on the SSSI citation description (available at www.sssi.naturalengland.org.uk), with
184 wetland habitats considered to include ponds, gravel pits, wet woodland or carr, fen, bog,
185 grazing marsh and wet common.

186

187 Seventeen environmental predictors were initially selected as candidates for modelling (Table
188 1) based on ready availability across the study landscape and considered, *a priori*, to
189 potentially influence ditch biodiversity. A single value of each variable was calculated for
190 each 1 km square. The mean elevation above sea level, presence of an A or B road and the
191 distance from the centre of each 1-km square to the nearest wetland SSSI, Fenland island and

192 the edge of the Fen basin were calculated. Previous work has suggested that ditches with
193 highest conservation value are located near to the edge of the Fen basin or close to Fen
194 islands (Mountford & Arnold, 2006); the reasons for this are unclear, but may relate to high
195 water quality. Fen islands were delimited as areas of $>0.1 \text{ km}^2$ with an elevation of $\geq 5 \text{ m}$, and
196 the Fen basin defined as the 5 m contour boundary.

197

198 The soils of the Fens area are dominated by silt and peaty soil types. The percentages of each
199 1-km square comprising silt and selected peat soil types (Table 1) were calculated. Ditch
200 isolation from main channels and from tidal influence were considered potentially important
201 determinants of water quality, saline influence and thus of biodiversity richness. We
202 calculated the shortest network distance along the ditches and rivers network (extracted from
203 the Ordnance Survey (OS) surface water polylines, converted into a raster of 35m cells),
204 from the centre of each 1-km square to the nearest main channel/river and to the tidal
205 boundary, calculated in ArcGIS Spatial Analyst tools. Network distances were not weighted
206 by ditch size or type, such that all cells were assigned a value of 1. A cell size of 35 m was
207 sufficient to connect any small breaks in the polylines due to mapping error or underground
208 drains, but was considered small enough to prevent falsely connecting ditches in close
209 proximity that are not connected through surface water drainage. Some manual connections
210 were imposed on the network due to large breaks in the mapped surface, for example due to
211 bridges or pumping stations. Ditch density in each 1-km was calculated from OS polylines,
212 which defines both banks of ditches wider than 2 m; since ditches of $<2 \text{ m}$ in width are only
213 defined with one polyline, ditch density is an index that reflect both linear length and ditch
214 area.

215

216 The grades of the Agriculture Land Classification were used as proxies for potential
217 agricultural productivity, land-use intensity and therefore quality of both water and
218 banksides; this is an ordinal scale (1-5) where grade 1 is best agricultural land. The combined
219 percentage cover of grades 3 and 4, comprising the lowest quality agricultural land and
220 therefore representing the lowest intensity of agricultural land-use (no land was classified as
221 grade 5 in our study region), was used as a candidate predictor. The dominant land use in the
222 Fens region is arable; the percentage of each 1-km square comprising un-intensively managed
223 grassland (defined from Land Cover Map (Morton et al., 2011)) classes of Rough/Neutral
224 Grassland) was therefore considered of interest. The percentage of urban land use was also
225 calculated from OS data.

226

227 Inter-correlation among predictor variables was investigated using Pearson's correlation
228 coefficient and considered large enough to potentially have an effect on the models if $r > 0.5$,
229 following Freckleton (2002). Distance to the Fen basin was strongly correlated with distance
230 to the nearest wetland SSSI ($r=0.533$), network distance to the tidal boundary ($r=-0.523$) and
231 percentage of silt soils ($r=0.536$). Distance to the Fen basin was therefore excluded from the
232 modelling, whilst the other variables were retained.

233

234 Due to comprehensive county flora, plant species recording effort was substantially greater in
235 Norfolk and Suffolk relative to other counties. Therefore, to avoid spurious identification of
236 any environmental factor that differed between these and other counties, when modelling the
237 response of wetland and aquatic plant variables to environmental and landscape context
238 indicators, we included the two county groups as a binary covariate (0 = no flora, 1 = flora).

239

240 A number of 1-km squares were excluded from the models because they contained no surface
241 water, the surface water was more than 70 m from the nearest surface water feature (thus
242 indicating the feature was likely to be a pond rather than a ditch, contained part of a wetland
243 SSSI, or comprised $>50\%$ coastal area (defined using the Wash SSSI). This resulted in 3,745
244 1-km squares being used in analyses.

245

246 *2.4 Model construction*

247 *2.4.1 Accounting for recording effort*

248 It is well known that not all species present at a site will be detected and that this poses
249 challenges for analysis (Chen et al., 2013), as species richness is underestimated and
250 coefficients with environmental variables are closer to zero. Spatial variation in recorder
251 effort can have severe consequences for models, as environmental variables that are
252 correlated with recording effort may be spuriously identified as being related to species
253 richness. Hierarchical occupancy modelling can address these problems by utilising repeated
254 visits to the same site to estimate detection probabilities (MacKenzie & Kendall, 2002) and
255 thus has applications for analysing citizen science data (Isaac et al., 2014). Despite extensions
256 to deal with multiple species (Dorazio & Royle, 2005), application to datasets such as ours is
257 challenged by, for example, uncertainty in defining what represents a discrete 'visit', and
258 absence of information on visits that did not contribute species records to the data. An
259 alternative approach to addressing spatial variation in recorder effort is to include a proxy for

260 recorder effort as a covariate (Hill, 2011), allowing the conditional effects of environmental
261 variables on species richness to be assessed while controlling for recorder effort. We use the
262 total number of records in a 1-km square (i.e. including non-wetland species) as a proxy for
263 recording effort. We expect this relationship to be saturating as species accumulation curves
264 tend to saturate at high numbers of species, so we explored models using either square root
265 number of records or a polynomial term for number of records, using the former as it
266 explained more deviance. Although our method accounts for spatial variation in recorder
267 effort, we are unable to estimate the probability of *not* detecting a species, so our estimates of
268 species richness should be taken as an index of relative richness.

269

270 *2.4.2 Predicting species richness*

271 Statistical analyses were performed using the computing environment R (R Core Team,
272 2012). Predictor variables were standardised prior to modelling, with the exception of the
273 number of records. For each response variable, we fitted generalised linear models, with a
274 quasi-poisson error structure to deal with over-dispersion, containing all 16 predictor
275 variables (17 for wetland and aquatic plants owing to the inclusion of county). The full model
276 was simplified by backward elimination, judging variable retention by the t-test of β
277 estimates, with a threshold of $\alpha < 0.05$. The resulting minimum models were used to predict
278 the richness of each of the seven wetland species groups in each 1-km square of the study
279 area, with recording effort standardised as the overall median (41 records per 1-km square).
280 For the wetland and aquatic plant response variables, we standardised for the presence of a
281 recent flora by setting the value for all squares as 1.

282

283 Following Legendre and Legendre (2012), we used variance partitioning to calculate the
284 proportion of total variation in species richness explained by recording effort (total number of
285 records) and by environmental variables. To do this, we constructed models including 1) only
286 environmental conditions, 2) only recording effort and 3) both environmental conditions and
287 recording effort.

288

289 *2.5 Comparison of predicted biodiversity richness to the current strategic planning maps*

290 The 1-km squares were ranked by the predicted species richness for each of the seven
291 biological response variables separately, where a high rank (low number) was given to
292 squares with high predicted biodiversity. The mean of these ranks was calculated and
293 mapped. The resulting map of predicted biodiversity was compared to the Fens for the Future

294 Partnership (FFFP) (2012) strategic connectivity plan. The strategic connectivity network
295 consisted of three types of corridors: primary, secondary and landscape (Fens for the Future
296 Partnership, 2012). The primary corridor was the priority corridor and aimed to connect three
297 core areas thought to have high biodiversity value, the southern Fens and Ouse Washes,
298 Holme and Woodwalton fens (and associated Great Fens Project restoration area of the
299 Wildlife Trusts), and the Nene Washes. Secondary and landscape corridors aimed to provide
300 additional landscape connectivity; for the purposes of this study, secondary and landscape
301 corridors were combined.

302

303 We designed a new connectivity network that met with the objectives of the strategic
304 connectivity network and the following criteria. Corridors must connect areas of known high
305 biodiversity richness (wetland SSSIs) and presumed high richness, defined as those wetland
306 Local Wildlife Sites (LWS) that were ≥ 0.25 km² and occurred in areas of high predicted
307 biodiversity (richest $\geq 50\%$ of 1-km squares). A single primary corridor was placed to connect
308 the three core sites identified by the FFFP (2012). All corridors must join to form a
309 continuous network across the region and, where possible, achieve such connectivity by
310 passing through areas of greater predicted biodiversity.

311

312 The potential conservation effectiveness of the original strategic plan was compared to that of
313 the corridor network we proposed on the basis of the predicted distribution of wetland
314 biodiversity richness. These were assessed for each corridor strata (primary, secondary) in
315 terms of the length within each quartile of predicted species richness (for each 1-km square,
316 the mean of ranked richness across all the seven species groups). Proposed networks were
317 deemed to be more effective if a greater proportion of the corridors lay within the quartiles
318 predicted to be the most species-rich.

319

320 **3. Results**

321 *3.1 Effect of the environment on wetland biodiversity richness*

322 Overall, the minimum models explained 27.2 – 63.9% (mean = 40.3%) of the variation in
323 species richness of the seven groups (Table 2), performing best in predicting the richness of
324 wetland plants and aquatic plants (63.9% and 59.8%, respectively). A substantial part of the
325 explained variance was attributed to the independent effect of recorder effort (27.3 – 76.2%).
326 However, 17.1 – 52.8% of explained variance was attributed to the independent effect of
327 environmental variables, and a further 2.5-26.4 % to the joint effect of recorder effort and

328 environmental variables (Table 2). Species groups with the highest proportion of variance
329 explained by the environmental variables were Odonata, aquatic species and littoral species
330 (53%, 36% and 33% respectively).

331

332 The effects of many environmental predictors were consistent among species groups. Mean
333 elevation above sea level and percentage of urban area were not significant predictors of the
334 richness of any group (Fig. 2). A greater percentage of silt soil was negatively related to
335 species richness of all groups, compared to all types of peat soil (Fig. 2), although the
336 richness of wetland and littoral species were also lower with a greater percentage of deep
337 sand over peat or peat (Fig. 2).

338

339 Richness of all groups, except Fen Specialists, was greater closer to existing wetland SSSIs.
340 The richness of all groups except wetland plants, increased significantly with increasing
341 values of the index of ditch density (Fig. 2). The percentage of grade 3 and 4 agricultural land
342 (i.e. lower land-use intensity) was positively related to richness of Odonata, but not
343 significantly related to the richness of other groups. The richness of Odonata also increased
344 further from the tidal boundary; in contrast, the richness of aquatic species, and wetland and
345 aquatic plants was higher closer to the tidal boundary. The distance to a main river was not
346 significantly related to the richness of Fen Specialists and littoral species. Richness of the
347 remaining groups was highest closer to a main river, although predicted richness generally
348 decreased when main rivers were located on silt soils or were further from a wetland SSSI
349 (Fig. 3). The predicted richness of all groups was low around the coast (Fig. 3).

350

351 *3.2 Biodiversity potential of the proposed network corridors*

352 The combined predicted richness of ditch species suggests that the corridors of the proposed
353 strategic network are generally well placed (Fig. 4, 5). However, comparison of the strategic
354 map and the predicted biodiversity richness indicated that proposed corridors do pass
355 through some areas of lower biodiversity potential (Fig. 4). In contrast, our suggested map
356 achieved a greater proportion of connectivity in areas of high predicted richness (88% of our
357 corridors were located in the richest 50% of squares, compared to 66% of the FFTF corridors)
358 for a shorter overall length (27% shorter, combined primary and secondary corridors) (Fig.
359 5).

360

361 **4. Discussion**

362 Landscape connectivity and conservation plans are often developed with a reliance on
363 environmental and land cover data (Brooks et al., 2004a), but such broad data can be poor
364 surrogates for biodiversity (Araujo et al., 2001; Schindler et al., 2013), particularly for rare or
365 specialist species (Lombard et al., 2003). Ecological planning should consider the identity,
366 distribution and requirements of target species in that region, rather than being based on
367 untested assumptions of where species occur (Brooks et al., 2004b) as such assumptions can
368 lead to inappropriate selection of habitat type or placement of the connecting elements. For
369 example, the previous landscape connectivity plan in the Fens that was based on expert
370 opinion selected the main river channels as a key connecting component (FFTP 2012).
371 Whilst we found that species richness was higher closer to main river channels, rivers
372 flowing through areas of silt soils had particularly low predicted species richness, so
373 improvements to management or connectivity in these areas may have limited benefits for
374 wetland biodiversity. This has important implications for other landscapes where a single
375 land cover variable has been the focus of network planning, because without validating with
376 biological data the use of single features can prevent selection of optimal connectivity.

377

378 Increasing ditch density was a significant predictor of species richness for all groups, except
379 wetland plants. The ditch density was a particularly strong predictor of priority species (those
380 with a conservation designation) associated with littoral margins. Littoral species are of
381 particular conservation importance in the Fens region, but are often overlooked by
382 conservation interventions compared to submerged aquatic species (Mossman et al. 2012).
383 Thus specifically targeting areas of high ditch density close to rivers for improved
384 management, rather than the main river channels themselves, would substantially add
385 conservation value. This highlights the importance of considering the identity and
386 requirements of the species that are the priorities for conservation and connectivity in a
387 region or a landscape.

388

389 Several broad and readily available landscape variables, such as distance to a protected site
390 (SSSI) and cover of silt soils, were important predictors of biodiversity. Thus, such variables
391 can be used to select areas for restoration or connectivity. The consistent negative response of
392 species richness to silt soils may be related to reduced water quality, since sediment nutrient
393 concentrations are higher in finer particle soils (Ockenden et al., 2012), or may reflect the
394 contrasting deposition and landuse histories, with peat soils indicating the historic extent of
395 freshwater marshes and earlier reclamation compared to the marine or riverine deposition of

396 silts that were reclaimed for agriculture more recently. Previous studies have found peat
397 substrates to have distinct flora (Mountford & Arnold, 2006) and support rare invertebrate
398 species (Foster et al., 1989); the richness of species groups in this study were not strongly
399 correlated with peat substrates.

400

401 Environmental factors, such as water quality (Twisk et al., 2000) and flow rate (Leslie et al.,
402 2012), and ditch management type and frequency (Milsom et al., 2004), are known to be
403 important determinants of ditch biodiversity. The inclusion of such variables would certainly
404 improve the predictive power of our models. However, such data were not available at
405 suitable resolution across our study area, and the case is likely to be the same in other
406 regions. We suggest that our predictive modelling approach is used in other regions to predict
407 areas of high potential biodiversity value. Following this, the collation or collection of
408 detailed environmental or habitat data may assist the selection of specific sites for
409 management interventions (such as dredging and cutting), within those areas highlighted by
410 the predictive mapping.

411

412 The effects of many environmental predictors were remarkably consistent among species
413 groups. For example, the richness of all groups was significantly greater closer to existing
414 wetland SSSIs. This may be because the high quality SSSI sites have acted as reservoirs of
415 wetland species, although there may be other conditions not included in this study (e.g. water
416 quality) that are also correlated with the distance to the SSSIs. Whilst the responses of most
417 groups were consistent, the richness of Odonata increased further from the tidal boundary; in
418 contrast, the richness of aquatic species, and wetland and aquatic plants was higher closer to
419 the tidal boundary. This is an important contrast, such that network planning must either take
420 a mixed approach, or select to prioritise either Odonata or remaining groups. Similarly, the
421 value of wooded connectivity networks is rather different for birds, bats and beetles
422 (Boughey et al., 2011; Davies & Pullin, 2007). This adds to previously stated concerns over
423 the use of single taxonomic groups as proxies for other biodiversity (Noss, 1990). Recent
424 work has demonstrated that the addition of habitat characteristics to multi-taxa proxy groups
425 substantially improves the performance of biodiversity surrogates in spatial planning (Di
426 Minin & Moilanen, 2014).

427

428 Biological records can be modelled with environmental variables to predict biodiversity
429 richness across landscapes and such models have been widely used to link species

430 distributions from atlas data to land cover data (e.g. Atauri & de Lucio, 2001; Virkkala et al.,
431 2005). Their use here to model species richness of priority biodiversity across multiple taxa
432 in the Fens allowed previously held assumptions about the importance of landscape features
433 to be tested. However, the use of such methods has been limited by the lack of detailed atlas
434 data for many taxa in many regions, with data for rare and threatened species and for poorly
435 recorded taxonomic groups (i.e. other than vascular plants, butterflies and odonatan)
436 particularly limited. We show that this problem can be overcome by modelling groups of
437 priority taxa with shared ecological requirements, which allowed us to include species that
438 would be too rare and/or sparsely recorded to model individually. This addresses a significant
439 gap in previous large-scale studies that have omitted due to insufficient data, the rare species
440 that are intended to benefit from the conservation measures. Our approach could be applied to
441 any region or landscape where there has been widespread, albeit patchy biological recording.

442

443 We were then able to predict potential species richness, including multi-taxa groups of
444 priority species, at a landscape scale and used the model predictions to make an evidence-
445 based landscape connectivity plan, an improvement on previous plans based on untested
446 expert judgement. Our models predict areas that have the potential for high biodiversity
447 richness, based on their soil and other landscape variables, and we have linked these together
448 with our proposed corridors. However, we do not know if the cells of our predicted corridor
449 currently realise that biodiversity potential with their existing habitat, which could still be
450 improved through enlargement or management, *or* if they currently have low habitat
451 suitability despite high potential on the basis their landscape variables. However, in either
452 case, we predict the potential to enhance biodiversity value and connectivity of those areas to
453 be greater than in areas with lower intrinsic potential and thus we are recommending these
454 areas should be targeted for enhancement.

455

456 The previous attempt to map a strategic connectivity network in the Fens (FFTP 2012)
457 largely concurred with areas of high predicted biodiversity richness. However, our evidence-
458 based map connected a greater proportion of areas with higher potential for biodiversity
459 richness (22% more of our corridors were located in areas of the highest potential richness)
460 and for a shorter overall length. Targeting areas of higher potential richness over a shorter
461 connectivity length is more cost-effective, allowing remaining funding to be targeted to
462 habitat management, a key influence on ditch biodiversity (Milsom et al., 2004). For
463 example, our evidence-based predictive map provides confidence in the strategic targeting of

464 agri-environmental measures and other means to enhance ditch management to those areas of
465 the wider agricultural landscape that have greatest biodiversity potential for aquatic and
466 wetland species.

467

468 Evidence-based predictive models, such as those in this study, could also be further
469 developed to inform optimal connectivity plans. For example, predicted potential species
470 richness can be used as a cost surface for circuit theory and other graph theory based models
471 (Galpern et al., 2011; Rayfield et al., 2011). Although we note that the practical realization of
472 any connectivity plan (subjective or objective) will be dependent on opportunity, landowner
473 and other stakeholder interest, and cost (Bergsten and Zetterberg, 2013), it is crucially
474 important to start negotiations based on evidence. Our methodology utilises ad-hoc records,
475 and thus could be applied in any landscape or region where biological records are available,
476 to provide an evidence-base for network planning, including rare species for which
477 conservation actions are most needed.

478

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TABLE LEGENDS

Table 1. Definition and data source of environmental predictors used to model the distribution of Fens biodiversity.

Table 2. Variation in the richness of wetland groups of species explained by the minimum models.

Table 1. Definition and data source of environmental predictors used to model the distribution of Fens biodiversity.

Environmental predictor	Abbreviation used in Figure 2	Source
Mean elevation above sea level: mean elevation of all 50 m x 50 m cells within the 1 km square	Elevation	
Distance to nearest fenland island: Fenland island defined as areas >0.1 km ² with an elevation of ≥ 5 m (excluding coastal cliffs at Skegness and islands within large urban areas). Several large ‘islands’ within 1000 m of the fenland basin were incorporated into the basin, i.e. not considered islands.	Distance to Fen Island	Edina Digimap Ordinance Survey (OS) PANORAMA DTM (Digital Terrain Model) 1:50,000, 50m cells
Distance to fenland basin: basin was defined as the 5 m contour boundary, unless the area had been defined as a fenland island.		
Presence of either an A or B road within a square	Presence of a road	OS Meridian 2 (1:50 000)
Distance to nearest SSSI comprising wetland habitats	Distance to wetland SSSI	Natural England GIS Digital Boundary Datasets
Percentage of square comprising urban areas. Urban defined from OS Strategic 1:250,000	% urban	
Network distance along ‘ditches’ to the nearest ‘main river’/coastline: calculated using network cost distance. Ditch was defined using the VectorMap District <i>Surface_Water</i> polyline for accurate mapping of small ditches and open water, and the <i>Tidal_Boundary</i> (High/Low Water Mark) polyline because the surface water data stop at the tidal boundary.	Distance to river	Edina Digimap Ordinance Survey Strategic 1:250,000 VectorMap District (1:25,000)

Network distance along ditch/river to the tidal boundary: calculated using network cost distance (see below for full description). Ditch/river defined using the Edina Digimap <i>River_polyline</i> and VectorMap District <i>Surface_Water</i> polyline. Tidal boundary was defined as the high water mark using the VectorMap District <i>Tidal_Boundary</i> polyline.	Distance to tidal boundary	
Index length of all ditches per 1 km square: ditches were defined as above. This is considered an index because polylines defined each bank of wide ditches or rivers, resulting in double-counting, as such the lengths are not accurate.	Length of surface water	
Percentage of rough and neutral grassland	% rough/neutral grassland	Land Cover Map 2007. Centre for Ecology and Hydrology
Percentage of grades of Agricultural Land Classification: summed percentage area of grades 3 and 4	% grades 3 & 4	Natural England GIS Digital Boundary Datasets
Percentage of each peat soil type defined using Cranfield Soil Class; Peat; Seasonally wet deep peat to loam; Seasonally wet deep clay over peat (marine alluvium and fen peat) and Seasonally wet deep sand over peat (glaciofluvial drift and peat).	% peat; % peaty loam; % deep sand over peat; % deep clay over peat	NATMAP Cranfield University
Percentage of silt soil, defined as the Cranfield Soil Class “Seasonally wet deep silty”	% silt	
Occurrence of a county flora: 0/1 if in a flora recorded county	County flora	

Table 2. Variation in species richness explained by the minimum models.

	Total r ²	% variation of total r ² explained		
		Independent effect of recording effort	Independent effect of environment	Joint effect of recording and environment
Odonata	30.4	27.3	52.8	20.0
Fen Specialists	27.2	46.4	27.1	26.4
Aquatic species	30.5	49.2	35.5	15.3
Aquatic plants	59.8	75.4	19.9	4.8
Littoral species	31.5	64.4	33.1	2.5
Wetland species	39.0	57.8	25.5	16.7
Wetland plants	63.9	76.2	17.1	6.7

LIST OF FIGURES

Figure 1. (a) The location of the Fens region within the UK, and b) the intensity of recording effort within the Fens, shown as number of records per 1-km square. Class intervals calculated using Jenks natural breaks.

Figure 2. Mean (\pm SE) standardised effect size (β values) of environmental predictor variables on the richness of ditch indicator groups. Only significant ($p < 0.05$) effects are shown. Predictor abbreviations are provided in Table 1.

Figure 3. Predicted richness per 1-km square for a) Odonata species, b) littoral priority species, c) aquatic priority species, d) aquatic plants, e) wetland priority species, f) wetland plants, and g) Fens Specialists. White areas denote 1 km squares that were excluded from models. Class intervals calculated using Jenks natural breaks.

Figure 4a. Mean of the ranks of predicted species richness per 1 km square of the seven wetland biological indicator groups. A low rank (high number) is given to squares with low predicted biodiversity and high rank (tied, highest = 44) to areas with high biodiversity. Main rivers and (a) connectivity corridors proposed by the Fens for the Future Partnership (excluding the Landscape Corridor) (FFFP 2012), and (b) connectivity corridors re-routed through areas of higher wetland species richness are shown. White areas denote 1 km squares excluded from models. Class intervals calculated using Jenks natural breaks.

Figure 5. Length (km) of primary and secondary connectivity corridors originally proposed by the Fens for the Future Partnership and alternative corridors selectively routed through areas of predicted higher wetland richness. Bars are shaded according to quartiles of the mean of ranks of biodiversity richness per 1-km square across seven indicator groups (Q1: 44-950, Q2: 951-1561, Q3: 1562-2372, Q4: 2373-3000).

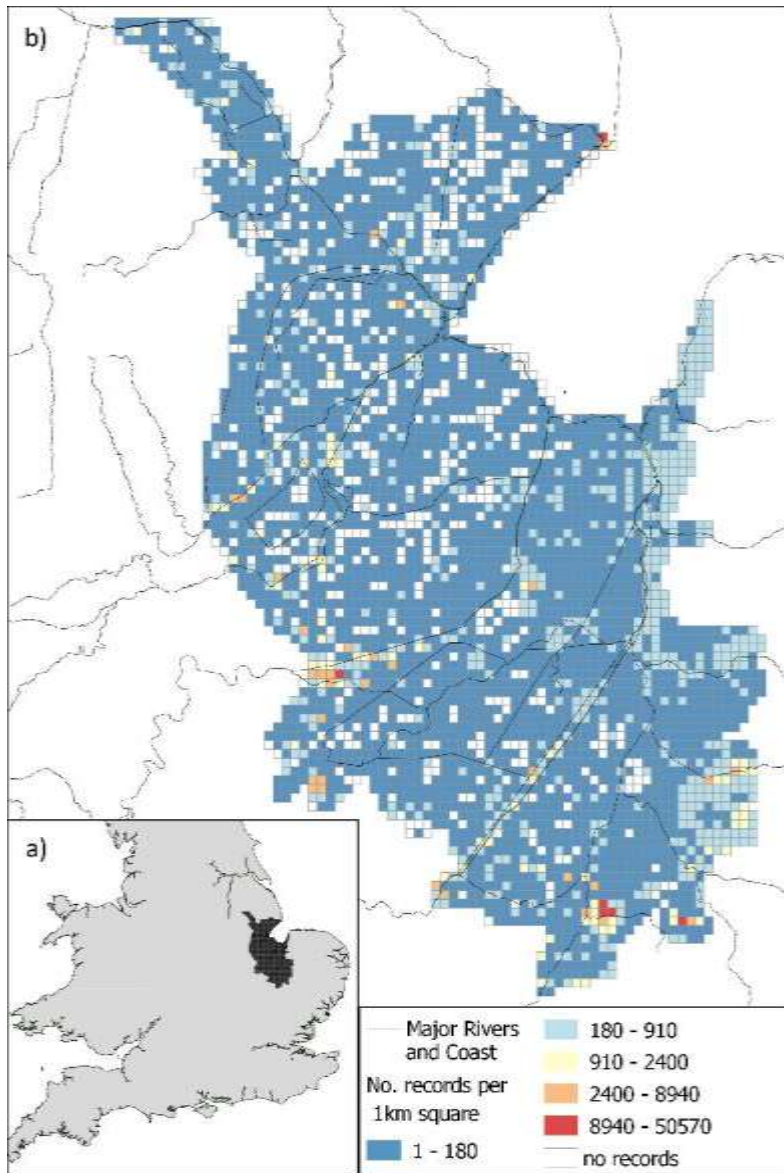


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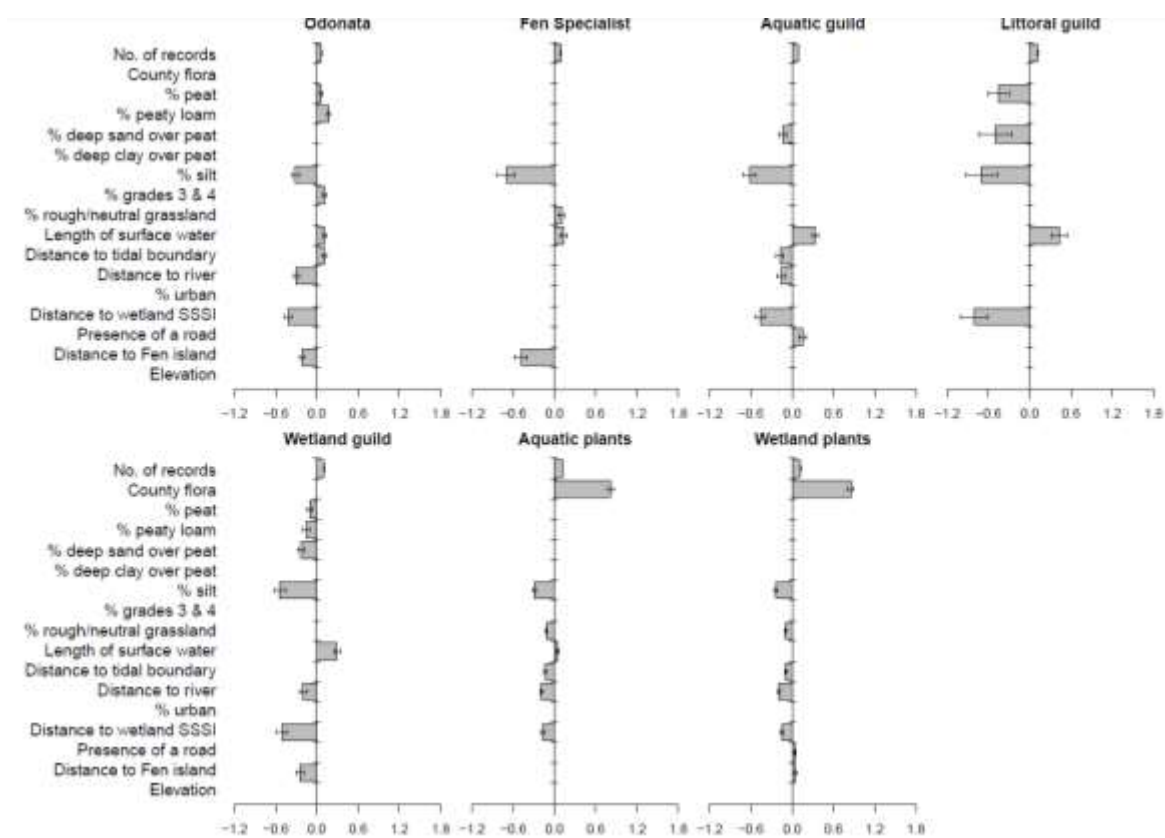


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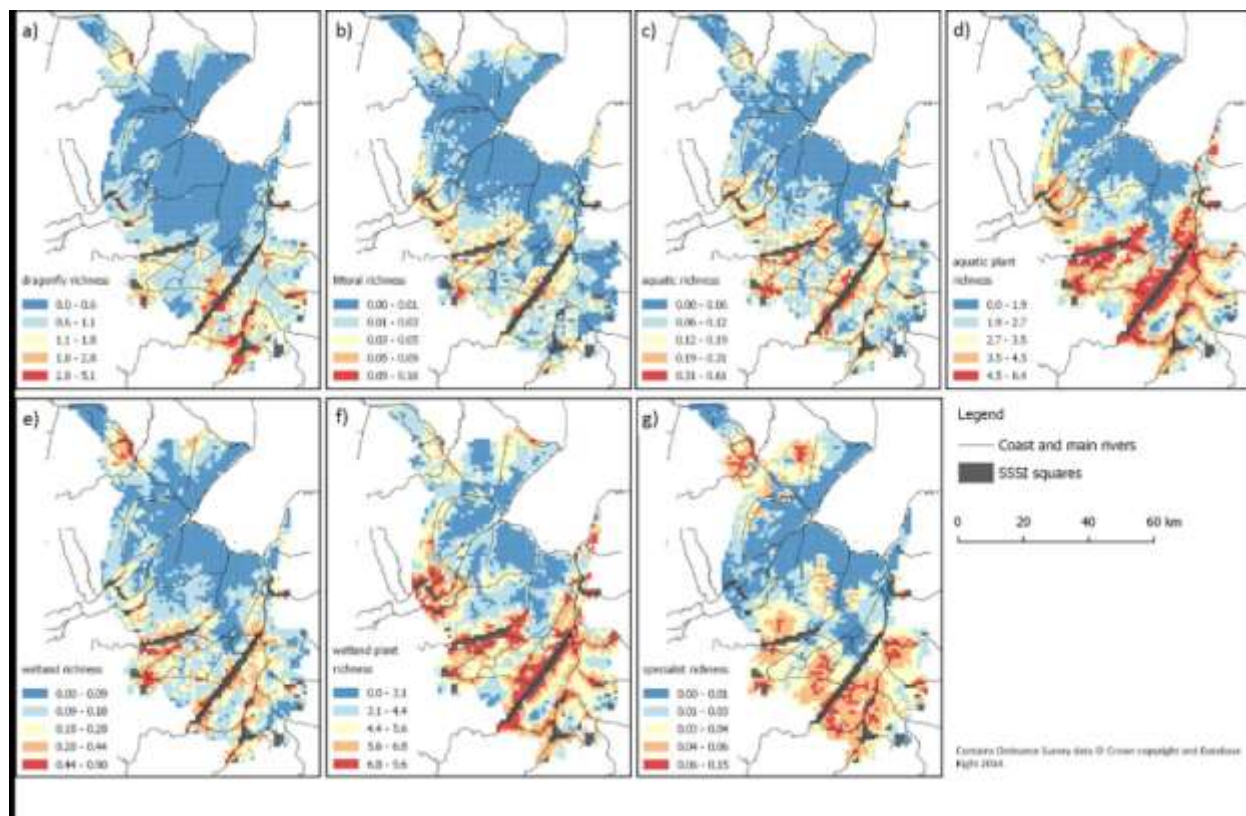


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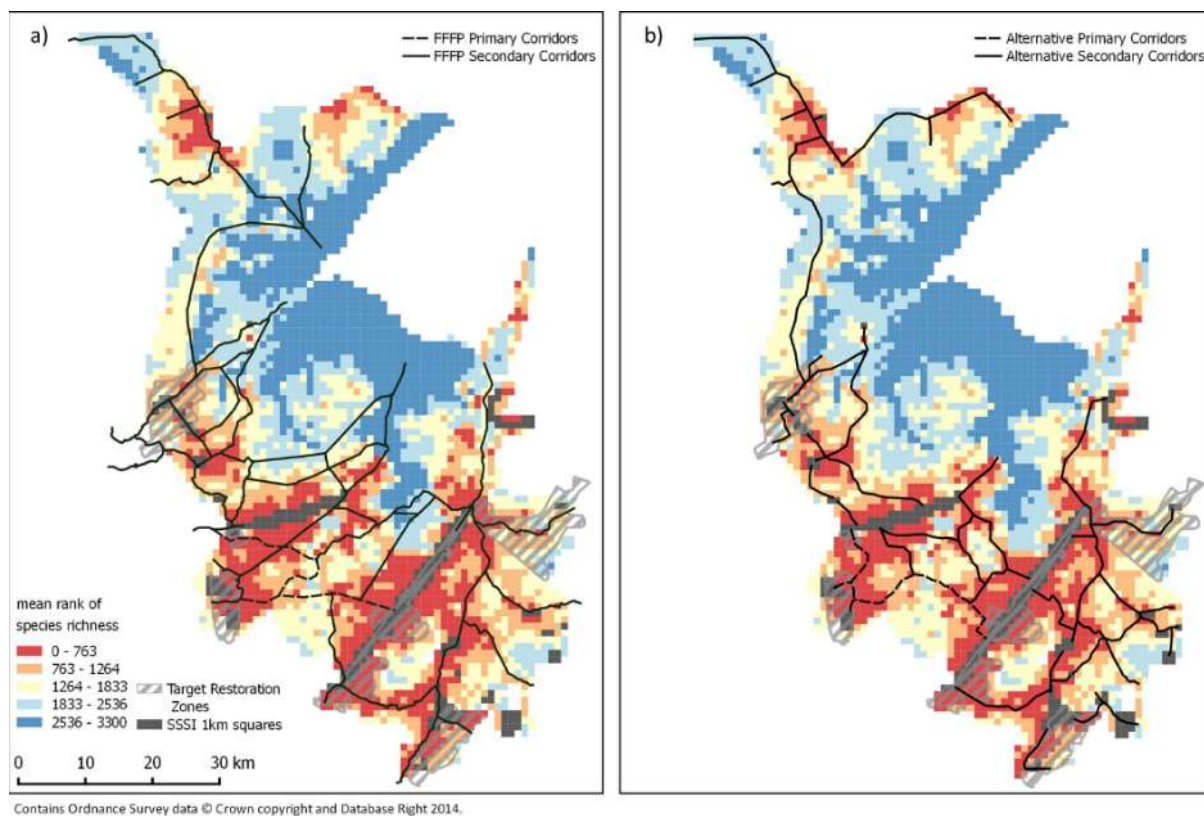


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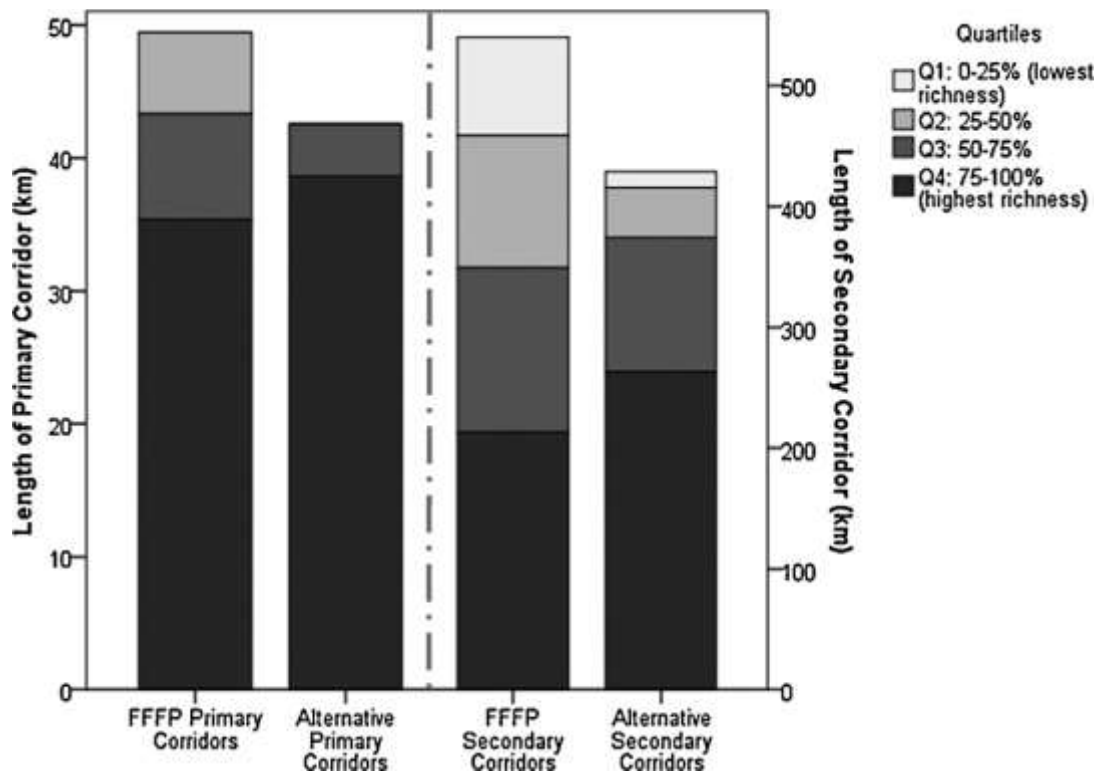


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