1 **Quantifying the contribution of riparian soils to the provision of ecosystem services**

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ABSTRACT

Riparian areas, the interface between land and freshwater ecosystems, are considered to play a pivotal role in the supply of regulating, provisioning, cultural and supporting services. Most previous studies, however, have tended to focus on intensive agricultural systems and only on a single ecosystem function. Here, we present the first study which attempts to assess a wide range of ecological processes involved in the provision of the ecosystem service of water quality regulation across a diverse range of riparian typologies. Specifically, we focus on 1) evaluating the spatial variation in riparian soils properties with respect to distance with the river and soil depth in contrasting habitat types; 2) gaining further insights into the underlying mechanisms of pollutant removal (i.e. pesticide sorption/degradation, denitrification, etc) by riparian soils; and 3) quantify and evaluate how riparian vegetation across different habitat types contribute to the provision of watercourse shading. All the habitats were present within a single large catchment and included: (i) improved grassland, (ii) unimproved (semi-natural) grassland, (iii) broadleaf woodland, (iv) coniferous woodland, and (iv) mountain, heath and bog. Taking all the data together, the riparian soils could be statistically separated by habitat type, providing evidence that they deliver ecosystem services to differing extents. Overall, however, our findings seem to contradict the general assumption that soils in riparian area are different from neighbouring (non-riparian) areas and that they possess extra functionality in terms of ecosystem service provision. Watercourse shading was highly habitat specific and was maximal in forests (ca. 52% shade cover) in comparison to the other habitat types (7-17%). Our data suggest that the functioning of riparian areas in less intensive agricultural areas, such as those studied here, may be broadly predicted from the surrounding land use, however, further research is required to critically test this across a wider range of ecosystems.

HIGHLIGHTS

- Habitat type is the main driver explaining riparian soil physicochemical variability.
- Riparian areas do not necessarily deliver greater ecosystem services.
- LiDAR data can support the identification of key areas to target to increase riparian shade. Riparian function can be largely predicted from neighbouring land use/soil type.
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1. Introduction

Ecosystem service-based approaches have been increasingly used to reduce pressure on natural resources and implement better land-management practices with respect to the environment (Van Looy et al., 2017). Riparian areas, the interface between land and freshwater ecosystems, are considered to play a pivotal role in the supply of regulating, provisioning, cultural and supporting services (Jones et al., 2010; Clerici et al., 2011; Aguiar et al., 2015). However, despite the fact that the number of studies referring to ecosystem services has increased by 38% in Europe over the last 20 years (Adhikari and Hartemink, 2016), riparian zones have received less attention than other land use types from an ecosystem services perspective. The few publications which have integrated an ecosystem service approach to the assessment of riparian areas have tended to address this from a modelling perspective (Clerici et al., 2014; Tomscha et al., 2017; Sharps et al., 2017). McVittie et al. (2015) proposed a model which aims to outline the fundamental ecological processes that deliver ecosystem services within riparian areas. Models provide a powerful and cost-effective tool to assess and map ecosystem services at the landscape scale, however,

they do not always provide a mechanistic process-level understanding. It is therefore important that models are supported and developed with robust underpinning data to correctly identify and describe the main factors affecting ecosystem services delivery within complex landscapes (i.e. those which may contain a diverse array of different riparian typologies). Little is known, however, about how inherent riparian properties and ecosystem functioning vary across different habitats within a catchment area (Burkhard et al., 2009). This uncertainty is largely due to the majority of riparian studies being focused on single sites, typically intensive agricultural systems (i.e. arable and grasslands) as these represent a major source of pollution (e.g. from fertilizers, livestock and pesticides) and because riparian zones associated with agriculture present pollution mitigation potential (Pierson et al., 2001; Rasmussen et al., 2011; Broetto et al., 2017). However, these studies tend to overlook the fact that riparian areas are inter-related systems and therefore changes (both natural and anthropogenic) occurring in headwater riparian zones across different habitat types could also affect riparian processes occurring downstream (Harper and Everard, 1998; Charron et al., 2008).

Among the many ecosystem services attributed to riparian areas, their role in water quality enhancement has grown in recognition over the years. Water quality has become a universal problem (Stephenson and Pollard, 2008) and is nowadays considered a priority objective for EU environmental sustainability (EEA, 2012). Increased loss of phosphorus (P) and nitrate (NO_3^-) from agricultural fertilizers has led to extensive eutrophication of surface and groundwaters (EEA, 2005), and contamination by pesticides and biological contaminants (e.g. bacteria) are regularly reported (Klapproth and Johnson., 2000; Troiano et al., 2001). Riparian areas are frequently proposed as a management strategy to reduce freshwater nutrient pollution (e.g. Coyne et al., 1995; O'Donnell and Jones, 2006; Stutter et al., 2009; Aguiar et al., 2015; Sgouridis and Ullah, 2015) and could also reduce the cost of drinking

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98   water purification (Klapproth and Johnson., 2000; Meador and Goldstein, 2003; Chase et al.,

99   2016). This pollution mitigation potential is often attributed to specific characteristics within

100   riparian soils (Mikkelsen and Vesho, 2000; Naiman et al., 2010). Table 1 summarizes the link

101   between riparian soil properties and the provision of ecosystem services found in the

102   literature. A better understanding of the causal factors for ecosystem services delivery will

103   provide an improved knowledge base on which to make land management decisions and

104   protection policies.

105   Many regulating services are highly affected by environmental conditions. For

106   example, temperature is known to directly and indirectly affect biological activity through its

107   impact on gaseous concentrations in soil (e.g. CO_2/O_2) and in the water column (Beschta,

108   1997; Verberk et al., 2016). It also plays an important role in determining the rate of key

109   ecosystem processes such as denitrification (Bonnett et al., 2013). Riparian buffers have

110   increasingly been used as a eutrophication mitigation tool by temperature regulation through

111   provision of shade (Nisbet and Broadmeadow, 2004; Burrell et al., 2014; Johnson and Wilby,

112   2015). Ghermandi et al. (2009) suggested that shading could viably be used as a management

113   option to improve water quality conditions in small and moderately-sized watercourses.

114   However, finding a cost-effective way to target vulnerable areas is challenging and has been

115   poorly explored to date.

116   The main focus of this study is to assess the link between riparian areas and the

117   regulating service of water purification through a wide range of ecological processes. In

118   particular, we aim to: 1) evaluate the spatial variation in riparian soils properties (i.e. general

119   nutrient status, soil acidity and conductivity, and microbial community size) with respect to

120   distance with the river and soil depth in contrasting habitat types; 2) gain further insights into

121   the underlying mechanisms of pollutant removal (i.e. pesticide sorption/degradation,

122   denitrification, etc) by riparian soils; and 3) quantify and evaluate how riparian vegetation

123   across different habitat types contribute to the provision of shade. This could help identify

124   areas especially vulnerable to excessive solar radiation and offer a cost-effective way to

125   improve ecosystem service provision (Ghermandi et al., 2009; De Groot et al., 2012). We

126   hypothesized that riparian areas would support a greater delivery of ecosystem services in

127   comparison to the upslope area, but that the balance of these services would be land use

128   specific within a catchment area.

129

130   **2. Methodology**

131   *2.1. Site description*

132   The Conwy catchment was chosen as a demonstration test site for this study due to its

133   extensive use in previous ecosystem service monitoring studies (Emmett et al., 2016). It is

134   located in North Wales, UK (3°50'W, 53°00'N) and comprises a total area of 580 km^2 (Fig.

135   1). The elevation ranges from sea level to 1060 m, with rainfall ranging between 500 to 3500

136   mm y^{-1} and the catchment has a mean annual temperature of 10 °C. Together, the

137   topography, parent material and climate have given rise to a wide range of soil types within

138   the catchment of which the dominant ones include Eutric Cambisols, Endoskelectic

139   Umbrisols, Albic Podzols and Sapric Histosols (WRB, 2014). It is predominantly a rural

140   catchment, with livestock farming (sheep and cattle) being the main land-uses. The two main

141   habitat types are improved (predominantly limed and fertilised) and unimproved grassland in

142   the lower altitudes to the east and mountain (exposed rock), heathland and bog in the western

143   part of the catchment. Extensive areas of coniferous (plantation) forestry and semi-natural

144   deciduous woodland can also be found in the upper reaches of the catchment.

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146   *2.2. Field sampling*

147   Five dominant habitat types (MHB = mountain, heath and bog; BW = broadleaf

148 woodland; CW = coniferous woodland; SNG = semi-natural grassland; IG = improved

149 grassland) were selected for soil sampling throughout the catchment. Habitat classification

150 was derived from the new Phase 1 National Vegetation Survey (Lucas et al., 2011) and

151 subsequently grouped, for simplicity, into the same broad habitat classes (see Appendix 1 for

152 details of groupings) defined in the UK's Land Cover Map 2007 (Morton et al., 2014).

153   Independent riparian sampling areas (n = 5) were selected from each of the 5 dominant

154 habitat types. At all sites, soil was collected at 2 m distance from a river and 50 m from a

155 river, which is regarded as the maximum extent of the riparian buffer zone and which

156 contained a different vegetation from that close to the river (De Sosa, 2017, unpublished

157 data). The sampling was designed to enable a direct comparison of how soil properties are

158 influenced by proximity to the river.

159   Intact soil cores (5 cm diameter, 30 cm long) were collected using a split tube sampler

160 (Eijklekamp Soil and Water, Giesbeek, The Netherlands) and separated into top- and sub-soil

161 fractions (0-15 cm and 15-30 cm depths respectively), stored in gas-permeable plastic bags

162 and transported to the laboratory for immediate analysis. These depths reflect the main

163 rooting zones in the soil profile (Glanville et al., unpublished data). In addition, the depths

164 were chosen to be consistent with those used in the national surveys for assessing changes in

165 soil ecosystem service delivery and which are used to directly inform land use policy at the

166 national-level (Countryside Survey, Glastir Monitoring and Evaluation Programme; Emmett

167 et al., 2010, 2016; Norton et al., 2012).

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169 *2.3. Soil characterisation*

170   Soil samples were sieved (< 2 mm) to remove stones and any visible plant material and

171 to ensure sample homogeneity (Jones and Willett, 2006). Samples were then stored at 4 °C

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172 prior to laboratory analysis. Soil water content was determined gravimetrically (24 h, 105 °C)

173 and soil organic matter (SOM) content was determined by loss-on-ignition (LOI) (450 °C, 16

174 h). Soil pH and electrical conductivity (EC) were measured using standard electrodes in a

175 1:2.5 (w/v) soil-to-deionised water mixture. Total available ammonium (NH_4-N) and nitrate

176 (NO_3-N) were determined with 0.5 M K_2SO_4 extracts (Jones and Willett, 2006) with

177 colorimetric analysis following the salicylate-based procedure of Mulvaney (1996) and the

178 VCl_3 method of Miranda et al. (2001), respectively. Available P was quantified with 0.5 M

179 acetic acid extracts (1:5 w/v) following the ascorbic acid-molybdate blue method of Murphy

180 and Riley (1962) and total C (TC) and N (TN) were determined with a TruSpec[®] elemental

181 analyser (Leco Corp., St Joseph, MI). Dissolved organic C (DOC) and total dissolved N

182 (TDN) were quantified in 1:5 (w/v) soil-to-0.5 M K_2SO_4 extracts using a Multi N/C 2100

183 TOC analyzer (AnalytikJena, Jena, Germany)(Jones and Willett, 2006). Microbial biomass C

184 and N was assayed by chloroform fumigation-extraction after a 72 h incubation using

185 conversion factors of $k_{ec} = 0.45$ and $k_{en} = 0.54$ (Vance et al., 1987).

186

187 *2.4. Process-level studies to measure ecosystem services*

188 A series of process-level studies were conducted to investigate how soils across

189 different habitats contribute to the regulation of important ecosystem services involved in

190 pollutant attenuation. In addition, we aimed to assess how habitat influences the provision of

191 shade and the impacts on temperature regulation. For all experiments, field-moist soil ($n = 5$)

192 was used to best represent field conditions.

193

194 *2.4.1. Phosphorus sorption to soil*

195 P adsorption isotherms were determined to estimate the soil's capacity for removing

196 dissolved P from solution, and hence assess the potential for soils to reduce the amount of P

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197 entering freshwaters. Sorption of P was determined following an adapted method of Nair et

198 al. (1984). In brief, 2.5 g of field-moist soil was shaken in 0.01 M $CaCl_2$ (1:5 w/v soil-to-

199 extractant ratio) containing known concentrations of P (0, 0.3, 1, 5, 10, 20 mg P l^{-1} as

200 KH_2PO_4) spiked with ^{33}P (PerkinElmer Inc., Walham, MA) (0.2 kBq ml^{-1}). These

201 concentrations were selected due to their likelihood of being encountered in the catchment

202 (DeLuca et al., 2015). Samples were shaken (2 h, 150 rev min^{-1}, 25 °C) on an orbital shaker.

203 This time was chosen to assess intermediate equilibrium conditions (Santos et al., 2011).

204 After 2 h, 1.5 ml of supernatant was removed, centrifuged (10,000 g, 5 min), and

205 subsequently, 1 ml of supernatant was mixed with 4 ml of Optiphase HiSafe 3 liquid

206 scintillation fluid (PerkinElmer Inc.). The amount of ^{33}P activity remaining in solution

207 measured using a Wallac 1404 liquid scintillation counter (Wallac EG&G, Milton Keynes,

208 UK) and the total amount of P adsorbed was determined as the difference between the

209 initial ^{33}P activity added and the final amount of ^{33}P remaining in solution. Any P not

210 recovered in the solution was assumed to be sorbed onto the soil's solid phase.

211 Sorption isotherms were examined according to the linearized form of the Langmuir

212 equation to estimate the P adsorption maxima and the P sorption binding energy for P (Reddy

213 and Kadlec, 1999; Mehdi et al., 2007):

214 $C/S = (1 / k \times S_{max}) + (C/S_{max})$ (Eqn. 1)

215 where S is the amount of P adsorbed (mg P adsorbed kg^{-1}), C is the equilibrium solution

216 concentration after 2 h (mg P l^{-1}), S_{max} is the P adsorption maximum (mg kg^{-1}), and k is a

217 constant related to the bonding energy (1 mg^{-1} P).

218

219 *2.4.2. Bacterial pathogen survival*

220 Soils from different habitat types were inoculated with human-pathogenic *Escherichia*

221 *coli* O157:H7 to investigate pathogen persistence in soils with respect to proximity to

222 waterbodies. Faecal samples, collected from a commercial beef farm in North Wales in

223 January 2016, were inoculated with *E. coli* O157:H7 to reproduce the natural vector by which

224 the pathogen is introduced into the environment (Jones, 1999; Williams et al., 2008). Samples

225 were transported to the laboratory and stored at 4.0 ± 0.1 °C prior to use. Both faecal and soil

226 samples were previously screened for the background *E. coli* O157:H7 cells using an

227 enrichment technique (Avery et al., 2008) and absence of *E. coli* O157:H7 was confirmed by

228 latex agglutination (Oxoid DR620; Oxoid Ltd., Basingstoke, UK). Prior to the start of the

229 experiment a basic characterization of the faecal samples was undertaken and moisture

230 content, organic matter, EC, pH, NO_3-N, NH_4-N and P determined as previously described.

231 The bacterial inoculum was prepared from a fresh overnight culture (LB broth; 18 h, 37 °C,

232 150 rev min^{-1} on an orbital shaker) of two environmental isolates of *E. coli* O157:H7 (strains

233 #2920 and #3704) (Campbell et al., 2001; Ritchie et al., 2003). A 40 ml aliquot of the *E. coli*

234 O157:H7 was added to 360 g of cow faecal samples and thoroughly mixed to deliver a final

235 concentration of approximately 10^8 cfu g^{-1} faeces (to reproduce the highest natural

236 concentration encountered; Besser et al., 2001; Fukushima and Seki, 2004). In brief, 5 g of

237 faeces spiked with *E. coli* O157:H7 was added to 5 g of soil in a sterile 50 ml polypropylene

238 tube and incubated at 10 °C (mean annual temperature for the catchment) for 1, 3, 7 and 14 d.

239 After each incubation time, samples were placed on an orbital shaker (150 rev min^{-1}, 15 min,

240 37 °C) with 20 ml of sterile quarter-strength Ringers solution (Oxoid Ltd.), followed by 4×3

241 s bursts on a vortex mixer. Serial dilutions were plated in duplicate onto Sorbitol MacConkey

242 agar (SMAC) (Oxoid Ltd.), then incubated (37 °C, 20 h) and colonies enumerated.

243 Presumptive *E. coli* O157:H7 colonies were confirmed via latex agglutination as described

244 previously.

245

246 *2.4.3. Pesticide sorption and degradation in soil*

247   The s-triazine herbicide, simazine ($C_7H_{12}ClN_5$; Water solubility, 5 mg l^{-1}; K_{ow}, 2.2;

248   pKa, 1.6), was selected to investigate the fate of a common pesticide when applied to soils

249   influenced by different environmental factors.

250   Simazine sorption followed the procedure of Jones et al. (2011). Briefly, 5 ml of [14]C-

251   labelled simazine (final concentration 0.5 mg l^{-1}; 0.02 kBq ml^{-1}) was added to 2.5 g of soil

252   contained in 20 ml polypropylene vials. The samples were then shaken (15 min, 200 rev min$^-$

253   ^{1}) to reflect instantaneous equilibrium conditions (Kookana et al., 1993). The extracts were

254   then centrifuged (10,000 g, 5 min) and the supernatant mixed with Scintisafe 3$^{®}$ scintillation

255   cocktail (Fisher Scientific, Leicestershire, UK). The ^{14}C activity remaining in solution was

256   then determined as described before. The simazine partition coefficient, K_d, was determined

257   as follows:

258    $K_d = C_{ads}/ C_{sol}$ (Eqn. 2)

259   where C_{ads} is the amount of simazine sorbed (mg kg^{-1}) and C_{sol} is the equilibrium solution

260   concentration (mg l^{-1}).

261   To determine how soil influences pesticide degradation, 5 g of soil was placed in

262   individual 50 ml polypropylene tubes and ^{14}C-labelled simazine was added to the soil at a

263   rate of 0.05 mg l^{-1} (0.25 µM; 0.2 kBq ml^{-1}). A 1 ml NaOH trap (1 M) was then placed into the

264   tube to capture any [14]CO_2 evolved. The tubes were hermetically sealed and placed at room

265   temperature (25 °C). The first NaOH traps were replaced after 24 h and then every 5 d for 30

266   d. On removal, NaOH traps were immediately mixed with Optiphase HiSafe 3 scintillation

267   fluid (PerkinElmer Inc.) and the amount of [14]CO_2 captured was determined using a Wallac

268   1404 liquid scintillation counter. Total simazine degradation was calculated as the cumulative

269   percentage of ^{14}C labelled CO_2 evolved at the end of the incubation period.

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271   *2.4.4. Nitrate loss from soil*

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272   Loss of nitrate via denitrification represents a major N loss pathway (Sgouridis and

273 Ullah, 2015). Denitrification capacity was estimated using the acetylene inhibition technique

274 (AIT) as described in Abalos and Sanz-Cobena (2013). Although the application of this

275 technique presents limitations (i.e. poor diffusion of C_2H_2 into the soil and inhibition of NO_3^-

276 production via nitrification), it has been widely used to give a qualitative estimate of

277 denitrification activity (Estavillo et al., 2002; Groffman and Altabet, 2006; Tellez-Rio and

278 García-Marco, 2015)).

279   In brief, 20 g of field-moist soil was placed in 150 ml gas-tight polypropylene

280 containers. Subsequently, KNO_3 (8 ml, 42.9 mM) was added to the soil to remove NO_3^-

281 limitation, the containers sealed and placed under vacuum and filled with O_2-free N_2 gas to

282 induce anaerobic conditions. Ten percent of the container headspace was then replaced with

283 acetylene to block the conversion of N_2O to N_2 gas. The containers were put on a

284 reciprocating shaker at 25 °C. After 0, 8 and 24 h, gas samples (10 ml) were removed with a

285 syringe and stored in pre-evacuated 20 ml glass vials, refilled with O_2-free N_2 gas. Nitrous

286 oxide was analysed by gas chromatography (GC) using a Clarus 500 GC equipped with a

287 headspace autoanalyzer Turbomatrix (HS-40) (PerkinElmer Inc.). Emission rates and

288 cumulative fluxes were determined as described by MacKenzie (1998) and Menéndez et al.

289 (2006), respectively.

290

291 *2.5. Water temperature regulation and riparian shading provision*

292   A GIS-based methodology was used to determine the extent to which vegetation

293 contributes to water channel shading in the different habitats. Based on the UK Environment

294 Agency 'Keeping River Cool' programme (Lenane, 2012), a LiDAR dataset (2 m resolution

295 Natural Resources Wales composite dataset) (Table 2) was used to provide a riparian shade

296 map to quantify how different habitat types and their associated riparian zones contribute to

297 shade provision. Using the ArcGIS Solar Radiation tool, we calculated the difference in

298 average incoming solar radiation during the summer months (1st May to 30th Sept.) between

299 two different elevation datasets to produce a measure of relative shade for the catchment. A

300 Digital Terrain Model (DTM) provided the 'bare earth elevation' whereas a Digital Surface

301 Model (DSM) provided the earth´s surface data including all objects on it. Differences in

302 incoming solar radiation between these datasets indicates the likely amount of shade created

303 by vegetation. Although the relative shade was calculated for the whole catchment, only the

304 parts which overlap with rivers were considered. The Zonal Statistics function (Arc GIS) was

305 used to attach the difference in solar radiation from the DTM and DSM to the water body

306 features (clipped using a 25 × 25 m grid in order to make small but similar sized units to

307 attach results) extracted from the OS Open Rivers dataset (Ordnance Survey, Southampton,

308 UK). The resultant shapefile was exported to Excel where shading differences were ranked

309 (1-20, with 1 being the least shaded and 20 the most shaded). The term "relative shading"

310 was used to refer to those areas that appear to have more or less than others due to the effect

311 of the vegetation. Finally, those areas which scored >10 on the ranking scale (higher

312 provision of shade) were then analysed to assess the influence of the habitat type on shade

313 provision. A 2 m margin was applied to each river, to ensure accurate intersection with the

314 adjacent Phase 1 habitat classification (Lucas et al., 2011) to estimate the percentage

315 occurrence of each habitat in relation to provision of shade.

316

317 *2.6. Statistical analyses*

318 For physicochemical soil properties, principal component analysis (PCA) was used to

319 explore the spatial relationships of selected soil properties for the different habitat types. A

320 two-way ANOVA was used to evaluate the interactions between physicochemical properties

321 with distance from river and soil depth within each habitat type. For each ecosystem process,

322   an independent t-test was performed to assess the influence of proximity to the river in terms

323   of ecosystem service provision. Pearson correlations were used to explore the relationships

324   between physicochemical properties and the results from the processing studies. All data

325   were analysed for normality and homogeneity of variance with Shapiro Wilk's tests and

326   Levene's statistics, respectively. Transformations to accomplish normality were done when

327   necessary. For all statistical tests, $P < 0.05$ was selected as the significance cut-off value.

328   Statistical analyses were performed with SPSS version 22 for Windows (IBM Corp.,

329   Armonk, NY).

330

331   **3. Results**

332   *3.1. Soil properties*

333   Principal Component Analysis (PCA) of the soil physicochemical variables of all

334   samples across the five dominant habitat types (see Methods for acronyms) ($n = 100$,

335   irrespective of distance or depth) identified two principal components (PC) which, together,

336   explain 66% of the total variance within the dataset (Fig. 2). Soil pH, available P, total C,

337   total N, DOC and TDN correlated significantly ($P < 0.001$) with the positive axis of PC1,

338   whilst microbial-N correlated significantly ($P < 0.001$) with the positive axis of PC2. Soil

339   moisture, organic matter, available NH_4-N and microbial-C correlated significantly ($P <$

340   0.01) with both PC1 and PC2.

341   Results of the PCA showed that habitat type (represented by cluster centroids, average

342   score on each PC1 and PC2 with standard errors) was an important predictor of soil

343   physicochemical variables. In terms of soil properties, BW and CW, and IG were closely

344   associated to each other in the Conwy catchment, although IG displayed overall higher total

345   C and N content (Table 3). At the other end of the spectrum (positive axis of PC1), the MHB

346   habitat was driven by moisture content (2.5 times more compared to woodlands and IG and

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347 1.5 times greater than SNG) and total C (ranging between 3.5 times greater than IG and 9.5

348 for BW) (Table 3). The SNG habitat resembled MHB in the sense that it had a greater

349 moisture content, total C and N compared to woodlands and IG habitats. However, they were

350 more influenced by microbial biomass showing larger variability in their vertical component.

351 The sites IG, SNG and BW were characterized by more alkaline pH values (ca. 5.2), whilst

352 MHB and CW displayed a more acidic pH (ca. 4.5) (Table 3).

353   As the objective of this work was to assess the influence of the river and soil depth in

354 terms of ecosystem service provision and not to compare different habitats, from this point

355 onwards we will focus on the influence of these factors within each habitat type.

356   The influence of soil depth and distance from river on physicochemical properties

357 within each habitat type is summarised in Tables S1-S5. Overall, soil depth showed no

358 significant effect on any of the soil physicochemical properties across habitat types, with

359 some exceptions. Microbial biomass-C was three times greater in the topsoil than subsoil in

360 MHB ($P < 0.01$) while microbial biomass-N differed approximately two-fold in the topsoil

361 compared to the subsoil in CW and SNG ($P < 0.05$). Total C showed a 72% change from top-

362 to sub-soil in IG ($P < 0.001$).

363   Available P was three times greater close to the river than 50 m away ($P < 0.01$) in

364 MHB but it was in the topsoil where the most noticeable difference was seen. The BW

365 habitat displayed the greatest difference when comparing physicochemical properties with

366 respect to distance. The BW habitat displayed 1.5 times greater EC away from the river,

367 whereas total N decreased by 1.5 times with distance from the river. Inorganic N (NH_4-N and

368 NO_3-N) showed a statistically significant increase (27% ($P = 0.042$) and 64% ($P = 0.004$)

369 respectively) away from the river whereas microbial biomass-N was 1.7 times less close to

370 the river.

371   The pH within the CW habitat showed a significant variation ($P = 0.002$) with a 10%

372   increase close to the river, whereas DOC was 1.5 times greater away from the river. Distance

373   had no effect in physicochemical properties in SNG and IG habitats with the exception of

374   microbial biomass-C in SNG which was 6-times greater close to the river, although the

375   standard error was quite high. Total N within the IG habitat showed an increase of 62% close

376   to the river ($P < 0.05$).

377   As depth was shown to have very little effect on soil physicochemical properties, this

378   factor was removed from the subsequent assessment of ecosystem services delivery.

379   *3.2. Ecosystem service provisioning*

380   *3.2.1. Phosphorus sorption to soil*

381   P sorption across all habitat types was generally well described by the Langmuir model

382   ($r^2 = 0.92 \pm 0.01$). P sorption maxima, S_{max}, ranged on average from 85 to 382 mg P kg^{-1}

383   across the five habitat types, showing the lowest sorption capacity with BW and the highest

384   in MHB. Results showed that MHB had consistently higher values of maximum P sorption

385   than the other habitats. Nonetheless, the binding parameter, k, that reflects the strength of P

386   sorption, was found to be highly variable and reduced for MHB whilst the rest of the habitat

387   types displayed a similar trend (Table 4).

388   Although river proximity did not have a significant effect on S_{max} ($P > 0.05$), SNG and

389   IG showed a tendency of greater P sorption closer to the river (Table 4). Significant positive

390   correlations ($P < 0.001$) were observed between S_{max} and moisture content, organic matter,

391   available forms of N and P, C content and microbial biomass. In contrast, S_{max} correlated

392   negatively with bulk density ($P < 0.001$). The most striking relationship was between S_{max}

393   and DOC and TDN, suggesting that organic matter might play a key role in P sorption

394   capacity.

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396   *3.2.2. Human bacterial pathogen survival in soil*

397   Overall numbers of *E. coli* O157:H7 declined significantly ($P < 0.001$) between the

398   first and the second harvest dates across all habitat types. After 24 h post-inoculation, a

399   decrease of ca. 20% of pathogen numbers were observed at all sites. Numbers then remained

400   relatively stable in the soil for all habitat types with the exception of SNG in which the final

401   percentage ($49 \pm 2\%$) differed significantly from the rest of the habitat types. The final

402   percentage decrease across the other sites was ~ 70%, suggesting different controlling factors

403   within SNG sites. In terms of distance from river, there was no significant effect ($P > 0.05$)

404   on persistence of *E. coli* O157:H7 colony counts and therefore, both values (close and far)

405   were amalgamated (Fig. 3).

406   *3.2.3. Pesticide sorption to soil*

407   Average K_d values, irrespective of distance to river, ranged from 11 to 484 l kg^{-1} across

408   all habitat types. The pesticide sorption capacity in MHB soils was 45 and 23 times greater

409   than in the woodland (BW and CW, respectively) soils and between 6 and 30 times greater

410   than SNG and IG sites (Fig. 4). Woodland (BW, CW) and IG habitats showed similar K_d

411   values (11 ± 2, 21 ± 3 and 16 ± 6 kg^{-1}, respectively) and the average K_d value for SNG was

412   79 ± 28 kg^{-1} which is midway between the MHB and woodland habitats. K_d values displayed

413   fairly similar trends ($P > 0.05$) when comparing results from close and far away from the

414   river (Fig. 4). Organic matter and moisture content correlated significantly ($P < 0.001$) with

415   K_d which might explain the higher sorption capacities within MHB and SNG habitat types.

416

417   *3.2.4. Pesticide degradation in soil*

418   After 30 d of incubation, the total percentage of simazine degradation ranged from 2.7

419   to 8.8% of the total ^{14}C-simazine activity added across habitat types irrespective of distance

420   from the river. The amount of simazine mineralized was noticeably less in the MHB sites

421 compared with the rest of the habitats. Across all habitats and distances, the rate of simazine

422 mineralization was maximal in the first week of incubation and then progressively decreased

423 over the 30 d incubation period. No significant differences were noted for MHB and IG with

424 respect to distance from the river. In contrast, significant differences with distance from the

425 river were observed in the two woodland habitats (Fig. 5; $P = 0.041$ for BW and $P = 0.035$

426 for CW). However, while the final percentage of simazine mineralized tended to be higher

427 close to the river in CW, the opposite trend was seen for BW. Across habitat types, the most

428 striking relationships between simazine degradation and soil physicochemical properties were

429 a positive correlation with pH ($P < 0.01$) and negative correlation with DOC ($P < 0.001$).

430 Simazine degradation also correlated negatively with N inorganic forms (NH_4-N, $P = 0.002$,

431 NO_3-N, $P = 0.003$) and available P ($P = 0.008$).

432

433 *3.2.5. Denitrification potential in soil*

434 Denitrification potential (DP) ranged between 0.25 and 1.94 mg N_2O-N m^{-2} d^{-1} across

435 habitat types based on a 24 h incubation. Overall, IG showed the highest DP, being 3 and 7.5

436 times higher than the MHB and the woodlands, respectively.

437 The influence of river proximity revealed no significant differences in N_2O emissions

438 ($P > 0.05$). Very different emission patterns were observed within each habitat, as indicated

439 by the large error bars in Figure 6, reflecting the spatial complexity and the presence of

440 denitrification hot spots across all habitat types. When hot spot values were removed from the

441 analysis, N_2O emissions were the same irrespective of proximity to the river for MHB, BW

442 and CW habitat types. Although not significant, emissions rates tended to be higher further

443 away from the river for SNG and CW whereas the opposite trend was found for MHB and

444 BW.

18

445   Overall, significant positive correlations ($P < 0.05$) were found between N$_2$O emissions

446   ($n = 50$) and bulk density and pH. Higher denitrification rates were found between pH 5 and

447   6 and bulk densities of 0.6 and 0.8 g cm^{-3}.

448

449   *3.2.6. Provision of riparian shade*

450   When evaluated across the whole catchment, the presence of woodland (CW and BW)

451   shaded 52.4% of the water channel. In contrast, in the MHB habitat the vegetation only

452   provided 7.6% shade cover. In the IG and SNG habitats the vegetation provided 17.4% and

453   12.9% shading respectively, however, this was partially due to the presence of isolated

454   hedges, trees and shrubs which were present within these habitats (Fig. 7).

455

456   **4. Discussion**

457   *4.1. General approach*

458   Our study investigated the spatial diversity of riparian soils and the ecological

459   processes that regulate the ecosystem service related to improving water quality. Soil

460   physicochemical properties were compared between samples taken close to (2 m) and distant

461   (50 m) from the river to further our understanding of how riparian specific soil characteristics

462   vary across different habitat types. Additionally, we explored different mechanisms of

463   pollutant removal (i.e. sorption, degradation and denitrification) and shading involved in

464   water quality enhancement with respect to riparian areas. We acknowledge that significant

465   gradients may exist across riparian areas, however, our sampling approach was designed to

466   simply compare soils in and out of the riparian zone. This approach reflects existing broad-

467   scale soil surveys which are used to measure and predict ecosystem service delivery at the

468   national scale (Emmett et al., 2010, 2016; Norton et al., 2012; Jones et al., 2014)

469   *4.2. Riparian soil physicochemical properties*

470 Many studies have linked the provision of riparian ecosystem services to their unique

471 intrinsic characteristics (Vought et al., 1994; Natta and Sinsin, 2002; Groffman and

472 Crawford, 2003). Riparian soils may have higher organic C contents (Figueiredo et al., 2016;

473 Graf-Rosenfellner, 2016), greater amounts of nutrients and fine-grained sediments (Lee et al.,

474 2000; Mayer et al., 2007), increased moisture contents (Lewis et al., 2003; Zaimes et al.,

475 2007) and microbial biomass (Naiman et al., 2010) than adjacent non-riparian areas. Contrary

476 to expectations, our findings contradict the frequently held assumption of riparian area

477 'uniqueness'. We observed little or no effect of the proximity to the river on the soil

478 physicochemical properties measured, despite major differences in vegetation community

479 composition and exposure to different hydrological regimes. General soil physicochemical

480 properties across habitat types followed the same trends as previous studies undertaken in the

481 catchment (Ullah and Faulkner, 2006; Sgouridis and Ullah, 2014; ;2015) and the inherent

482 habitat characteristics proved to be the main drivers explaining soil physicochemical

483 variability in riparian areas. In support of our findings, Richardson et al. (2005) also noticed

484 little difference in soil properties between riparian and upslope areas along small streams in

485 temperate forested areas of the Pacific Northwest. In addition, riparian studies have

486 commonly focussed on agriculturally-managed grasslands and more specifically on riparian

487 buffer strips as management tools (Pierson et al., 2001; Hefting and Bobbink, 2003; Hickey

488 and Doran, 2004), even though this habitat type has shown less value in terms of ecosystem

489 service provision (Maes et al., 2011; 2012). Stutter et al. (2012) and Smith et al. (2012) found

490 significant differences when comparing soil physicochemical properties of riparian buffers

491 versus adjacent fields. However, the comparison was undertaken between areas which

492 possessed vastly different management regimes and in which the vegetation cover changed

493 dramatically. Similarly, Burger et al. (2010) also showed differences in soil properties

494 between agriculturally impacted riparian areas and ones conserved in pristine natural

495   conditions. Most of the habitats assessed in our study have little or no management

496   intervention so natural or semi-natural habitat conditions remained consistent across the

497   upslope and riparian area. This was true even for the areas subject to agricultural practices

498   (improved and to a lesser extent semi-natural grassland), although it should be stated that

499   these agricultural areas generally have good soil quality (unlike those under arable cropping;

500   Emmett et al., 2016). It is possibly for this reason that we did not identify any significant

501   change in soil physicochemical properties as reported by others. Further studies are therefore

502   needed to take into account management intensity and to include seasonal patterns as they

503   may also represent an important component in riparian dynamics (Dhondt et al., 2002; Greet

504   et al., 2011).

505

506   *4.3. Ecosystem service provision*

507   In comparison to the surrounding region, riparian areas are usually considered to have

508   extra functionality in terms of ecosystem service provision through enhanced flood control,

509   water purification or biodiversity (Salo and Theobald, 2016; Sutfin et al., 2016; Xiang et al.,

510   2016). However, in our study there was no evidence that fundamental differences exist

511   between riparian zones and the adjacent land. This is supported by the clear segregation of

512   results according to habitat types and not by riparian areas (Fig. 8). Main habitat

513   characteristics and not distance from the river was the driving factor in all cases. In this

514   respect, Table S6 summarizes the soil habitat physicochemical properties which are most

515   likely to be driving the ecosystem service delivery in this study. Together with that, we also

516   include other factors that, despite not being measured, should be considered in future riparian

517   studies to predict the spatial and temporal variation in ecosystem service delivery. These

518   processes could be responsible for creating 'hot spots and moments' within riparian zones

519   (McClain et al., 2003; Vidon et al., 2010). For example, erosion is more prevalent in riparian

520   areas due to the exposure to a more dynamic water regime (McCloskey, 2010). This can

521   cause a large release of N, P and C into the water column producing similar loads to those

522   induced by fertilizer application (Quinton et al., 2010). Likewise, water table fluctuations that

523   modifies oxygen levels and nutrient availability, and the presence of macrophytes are also

524   good examples that could potentially alter ecosystem service delivery dynamics in riparian

525   areas (Naiman and Decamps, 1997; Hill, 2000; Lewis et al., 2003; Ng and Chan, 2017).

526

527   *4.3.1 Pollutant removal via sorption*

528   Values of S_{max} (P sorption) and K_d (simazine sorption) resulted in good agreement with

529   other values found in the literature across habitat types (Dunne et al., 2005; Flores et al.,

530   2009). Analysis suggested that simazine and P sorption was driven by high organic matter

531   content as has been highlighted in previous studies (Li et al., 2003; Hogan et al., 2004; Kang

532   and Hesterberg, 2009; Alister and Kogan, 2010). Particularly for P sorption, some authors

533   attribute this affinity of P for organic matter to the co-occurrence of Al and Fe oxides, which

534   can sorb high amounts of P (Pant et al., 2001; Kang and Hesterberg, 2009). We had expected

535   that the riparian areas would be wetter, have a lower redox status and would contain a lesser

536   amount of oxidsed forms of Fe and thus a lower P retention capacity, however, this was not

537   apparent in our soils. Barrow (2017) illustrated different pathways for P sorption according to

538   soil pH but due to the relatively small shifts in pH relative to the distance to the river, no such

539   effect was found in this study.

540   Comparing the results obtained in this study is challenging as most studies within

541   riparian areas try to identify the most cost-effective buffer width depending on the pollutant

542   load in agricultural systems or constructed wetlands. This is motivated by the fact that land

543   managers do not want to sacrifice more productive land than they have to (Wenger, 1999;

544   Shearer and Xiang, 2007). Consequently, the centre of attention has been on comparing

545   inputs versus outputs of pollutants in runoff through vegetative buffer strips (Schultz et al.,

546   2000; Maillard and Imfeld, 2014). Results found in the literature about the long-term

547   effectiveness of riparian buffers in trapping pollutants are contradictory as riparian areas can

548   vary from being sources to sinks depending mostly on physicochemical soil properties and

549   hydrology (Hickey and Doran 2004; Fisher and Acreman, 2004; Stutter et al., 2009; Maillard

550   and Imfeld, 2014). Some studies (e.g. Miller et al., 2016) reported different P retention

551   capacities with distance from the river. However, it was only true for samples included inside

552   a concentrated flow path that was visually identified prior to sampling. In contrast, samples

553   outside this concentrated flow path did not reveal any differences in P retention across the

554   transect.

555   The similar pollutant sorption capacities relative to distance from the river found in this

556   study, combined with fact that simazine and P retention by soil can only occur when they are

557   in direct contact with the adsorbent suggest that the soil potential data alone is not very useful

558   in predicting the pollutant retention capacity (Reddy and Kadlec, 1999). Thus, the study of

559   transport pathways, potential sources of pollutant loads, ease of degradation, desorption

560   potential from the soil, shifts in temperature that controls simazine solubility or pH that

561   controls P precipitation may contribute more efficiently to understanding riparian pollutant

562   attenuation.

563

564   *4.3.2 Pollutant removal through degradation*

565   Degradation, together with sorption, is one of the main processes determining the fate

566   of pollutants within the environment (Gunasekara et al., 2007; Maillard and Imfeld, 2014). In

567   our study, we investigated the degradation of a pesticide and loss of the biological

568   contaminant, *E. coli* O157, which are of concern in terms of their impact on human health

569   (Holden et al., 2017). Sorption and transport of pollutants, and the extension of buffer strips

570   on agricultural and wetland systems has often been the focus of attention (Vellidis et al.,

571   2002; Hickey and Doran 2004; Rasmussen et al., 2011), but processes influencing pollutant

572   degradation in riparian areas are much less well understood (Vidon et al., 2010). Microbial

573   activity has long been identified as a critical factor determining the fate of pesticides in the

574   environment (Kaufman and Kearney, 1976; Anderson, 1984), and it is suggested that

575   microbial populations within riparian areas are able to degrade pesticides due to their

576   continuous exposure to such chemicals through runoff from agricultural lands (Vidon et al.,

577   2010). Overall, simazine degradation in this study showed a similar percentage decrease (of

578   the total of ^{14}C-simazine added) to other studies (Laabs et al., 2002; Gunasekara et al., 2007;

579   Jones et al., 2011). Laabs et al. (2002) and Cox et al. (2001) found a negative correlation

580   between simazine degradation rates and organic matter content due to the residue binding to

581   organic matter reducing herbicide movement in the soil. This fact could explain the minimal

582   amount of simazine degraded in MHB sites in this study. Previous studies have demonstrated

583   enhanced pesticide degradation within riparian areas (Mudd et al., 1995; Staddon et al.,

584   2001). However, the riparian buffer strips in these previous studies differed considerably

585   from the adjacent habitat (i.e. bare or highly modified fields versus vegetated buffer strips).

586   In our study, only the woodlands showed a different pattern in terms of pesticide degradation

587   when comparing sites close and distal to the river. However, we hypothesized that the

588   negative correlation between simazine degradation and N and P inorganic forms content

589   could explain this spatial variability as the use of pesticides as a source of energy in areas

590   with low nutrient status has been identified (Błaszak et al., 2011). In addition, it has been

591   shown that some organisms (e.g. *Pseudomonas*) are able to mineralise simazine more rapidly

592   (Regitano, 2006; Błaszak et al., 2011) and therefore a more diverse microbial population

593   associated with a higher above-ground plant diversity could be involved in different

594   ecosystems. Our results may therefore reflect the spatial heterogeneity of microbial

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24

595 populations within these habitat types rather than a specialization of microbial population in

596 riparian areas. This fact is endorsed by studies like Widenfalk et al. (2008) where an effect on

597 microbial composition due to pesticide exposure could not be identified. Our results reveal

598 that there is a need for linking functional soil biota groups with the maintenance of ecosystem

599 services to better explain the inherent spatial heterogeneity (Brussaard, 1997; Graham et al.,

600 2016).

601   Along with pesticides, biological contaminants, in particular faecal coliform bacteria

602 (FCB), have become an important source of water contamination from human and animal

603 wastes applied to land (Bai et al., 2016). Although the use of riparian buffer strips for

604 reducing FCB transport into streams has been explored (Coyne et al., 1995; Parkyn et al.,

605 2003; Sullivan et al., 2007), bacterial survival and behaviour in terrestrial systems has

606 received less attention than in water ecosystems (Jones, 1999). Our results corroborate

607 previous studies that show *E. coli* O157 can survive for long periods (more than 120 d) in a

608 diverse range of soils and under a wide range of environmental conditions (Bogosian et al.,

609 1996; Kauppi and Tatini, 1998; Jones, 1999). Some studies have suggested that moisture

610 status and organic matter are the principal factors controlling *E. coli* survival (Jamieson et al.,

611 2002). However, the lack of correlation between soil properties and pathogen survival in this

612 study suggest that other factors, such as predation or the presence of elements highlighted in

613 other studies (Al, Zn; Avery et al., 2008), might better explain the lower survival rate found

614 in semi-natural grassland sites.

615

616 *4.3.3 Pollutant removal through denitrification*

617   Denitrification, as a mechanism for permanent removal of NO_3^- from ecosystems, has

618 important implications for both water quality and greenhouse emissions (Groffman et al.,

619 2009). It has been extensively studied in riparian areas due to the frequency of locally anoxic

25

620 conditions and labile organic C which trigger denitrification (Bettez and Groffman, 2012). In

621 our study, rates of N_2O emissions across habitat types followed similar trends to those

622 described in Sgouridis and Ullah (2014). However, we could not find any clear evidence that

623 leads us to identify more efficient patterns of NO_3^- removal by denitrification with proximity

624 to the river. We also observed a high degree of spatial variability in denitrification with some

625 extremely high rates as has been observed in other studies and described as 'hot spots or

626 moments' controlled by oxygen, NO_3^- and C availability (Parkin, 1987; McClain et al., 2003;

627 Groffman et al., 2009; Vidon et al., 2010). Previous riparian studies have also reported no

628 clear spatial patterns in denitrification rates (Martin et al., 1999). In our study, it was clear

629 that the addition of NO_3^- was not sufficient to trigger large amounts of N_2O production,

630 indicating that factors other than NO_3^- limitation were playing a key role. Sgouridis and

631 Ullah (2015) describe significant relationships between denitrification rates and pH and bulk

632 density, and the same pattern was found in our study. However, those factors do not explain

633 the high variability encountered within habitat types, and it was not possible to demonstrate

634 significantly increased N_2O production rates within riparian areas as demonstrated in

635 previous studies (Hanson et al., 1994; Groffman et al., 2000; Groffman and Crawford, 2003).

636 Further research is therefore required to better understand why denitrification is so spatially

637 variable and the spatial/temporal existence of 'hot spots or moments'.

638

639 *4.3.4 Riparian shading*

640 Riparian shading is gaining increased recognition for its potential to alleviate water

641 pollution (Ghermandi et al., 2009; Warren et al., 2017). For example, Hutchins et al. (2010)

642 found that the reduction of nutrient pollution was less effective at suppressing phytoplankton

643 growth than establishing riparian shading. Bowes et al. (2012) also noticed a potential

644 reduction of 50% of periphyton accrual rate through shading in the River Thames.

645   The shade mapping approach presented here provides an easy tool to identify

646   watercourse exposure to solar radiation. As described in Lenane (2012), the maps generated

647   using this approach, offer the guidance necessary to help with riparian management plans and

648   decision-making strategies. Identifying whether riparian vegetation is providing effective

649   shade is fundamental for environmental protection. Furthermore, the size of this area required

650   to provide shade has economic implications as it takes the land out of production (Sahu,

651   2010). The shade evaluation undertaken in this study differs from others in which field

652   monitoring are required (Boothroyd et al., 2004; Halliday et al., 2016) and consequently it

653   avoids excessive costs associated with field measurement campaigns. However, it does not

654   predict water quality changes as proposed by Ghermandi et al. (2009) which combines

655   available flow measurements with biochemical and shade models.

656   As expected, in our study the effects of shading were more significant in woodlands

657   than in any other habitat type. Woodland riparian zones are likely to offer the greatest

658   influence on water temperature within a catchment. Any assessment, however, should also

659   consider excessive shading, mostly caused by abandoned woodlands (Suzuki, 2013) which

660   can be detrimental to aquatic ecosystems by excessively reducing water temperature. This

661   can have a direct impact on aquatic fauna and result in a loss of shade-intolerant plants

662   (Forestry Commission, 2004; Hédl et al., 2010). Shading may also reduce the UV radiation-

663   induced photooxidation of many pesticides within the water column.

664

665   **5. Conclusions**

666   Recommendations and guidance about riparian zone management are frequently

667   undertaken without an accurate evaluation of their status and the ecosystem services that they

668   actually provide. Consequently, many previous environmental protection measures involving

669   riparian management remain too general and untargeted and may offer little environmental

670   benefit. Through a series of laboratory experiments and GIS-based mapping, this study has

671   shown that across a diverse range of habitats, riparian soils diverge from their capacity to

672   deliver the specific ecosystem service of water purification. However, contrary to

673   expectation, riparian soils did not differ greatly in their ability to provide this service in

674   comparison to neighbouring upslope (non-riparian) soils. We ascribe this to our habitats

675   being in a close to natural or semi-natural state rather than the more frequently studied

676   riparian areas in degraded agricultural systems. Further work should focus on validating our

677   findings using an even greater range of ecosystem services (e.g. inclusion of CH_4/CO_2

678   emissions, metal attenuation, biodiversity), using in situ measurements, encompassing inter-

679   annual variation and over a wider range of ecosystem types.

680

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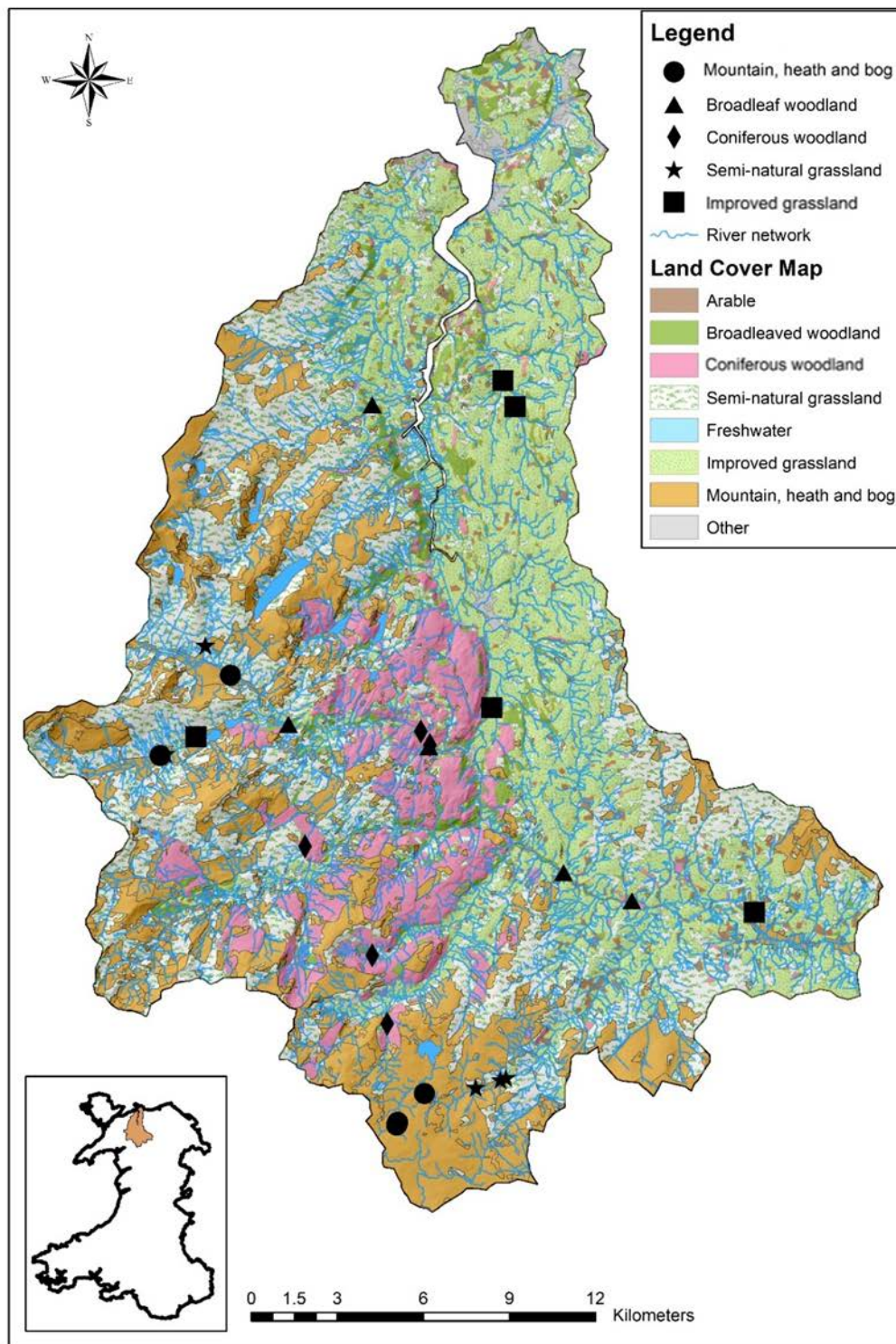
Figure 1. The Conwy catchment, North Wales, UK showing location of sample points, land cover classes (Lucas et al., 2011) and river network. Samples sites were distributed within the five dominant habitat types in the Conwy catchment (mountain, heath and bog, broadleaf and coniferous woodlands, semi-natural grassland and improved grassland) and each symbol represents a pair of sample points, one at 2 m and another at 50 m distance from the river system (*n* = 10).
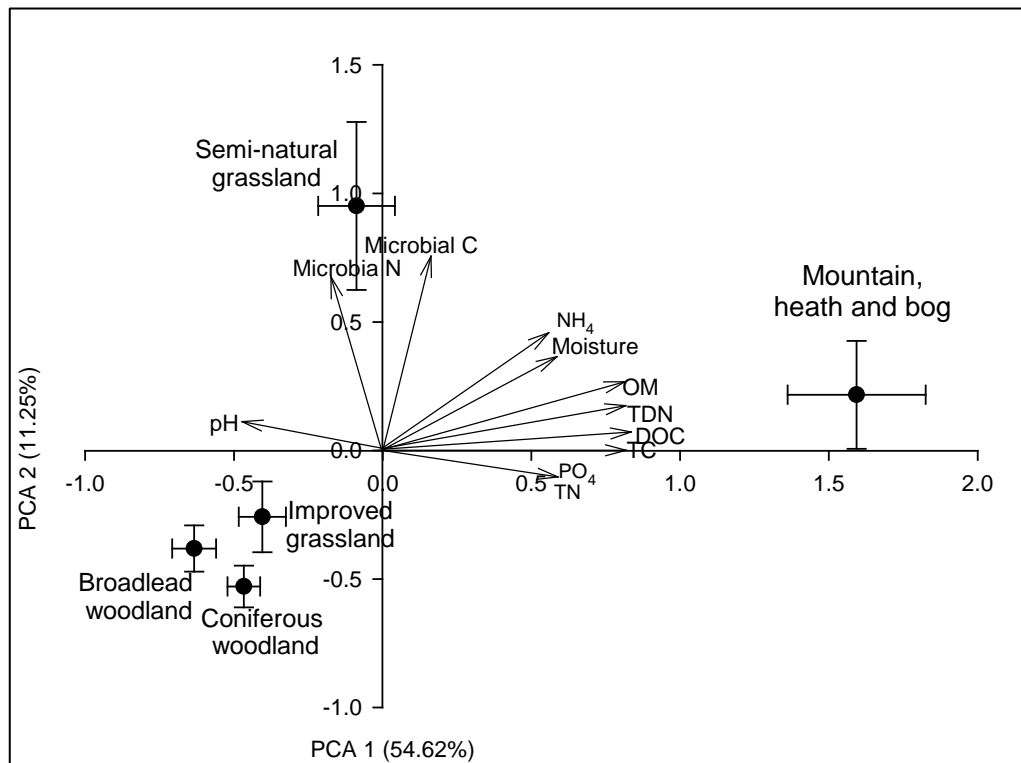
Figure 2. Correlation bi-plot from the principal component analysis (PCA) on soil physicochemical variables according to their dominant habitat type and irrespective of distance and depth ($n = 100$). Correlation of soil properties with the main axes are given by arrows and habitat types by cluster centroids (average score on each horizontal principal component (PC1) and vertical principal component (PC2) with standards errors). Organic matter (OM). Total carbon (TC). Total nitrogen (TN). Dissolved organic carbon (DOC). Total dissolved nitrogen (TDN).
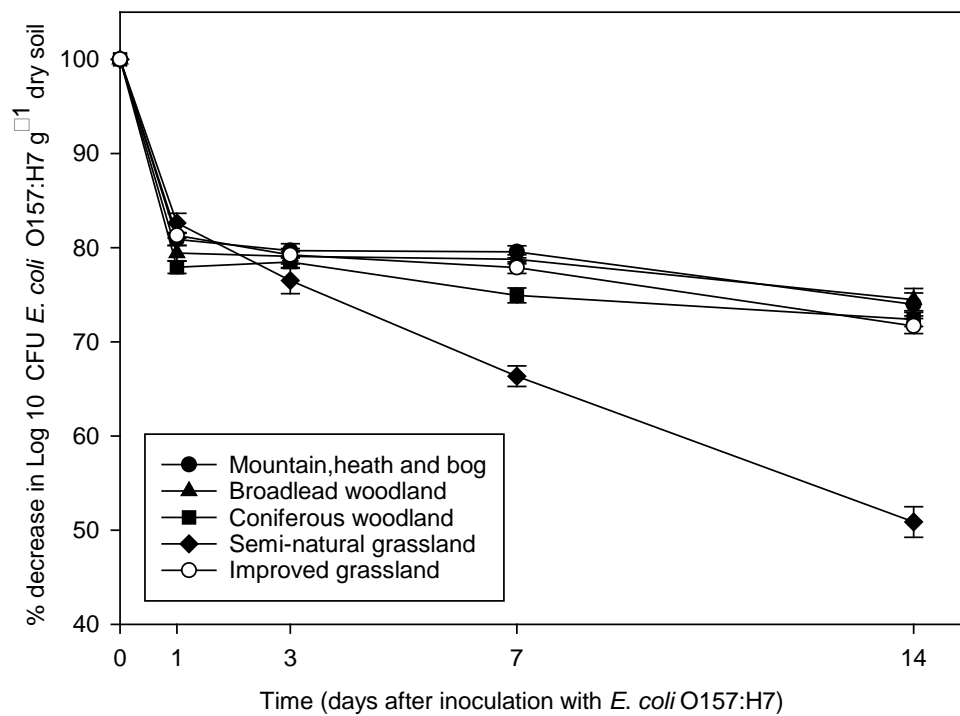
Figure 3. Survival of *Escherichia coli* O157:H7 following the application of pathogen-contaminated cattle slurry to the soil from different habitat types amalgamating distance from the river. Data points represent mean values ($n = 10$) ± standard error of the mean (SEM).
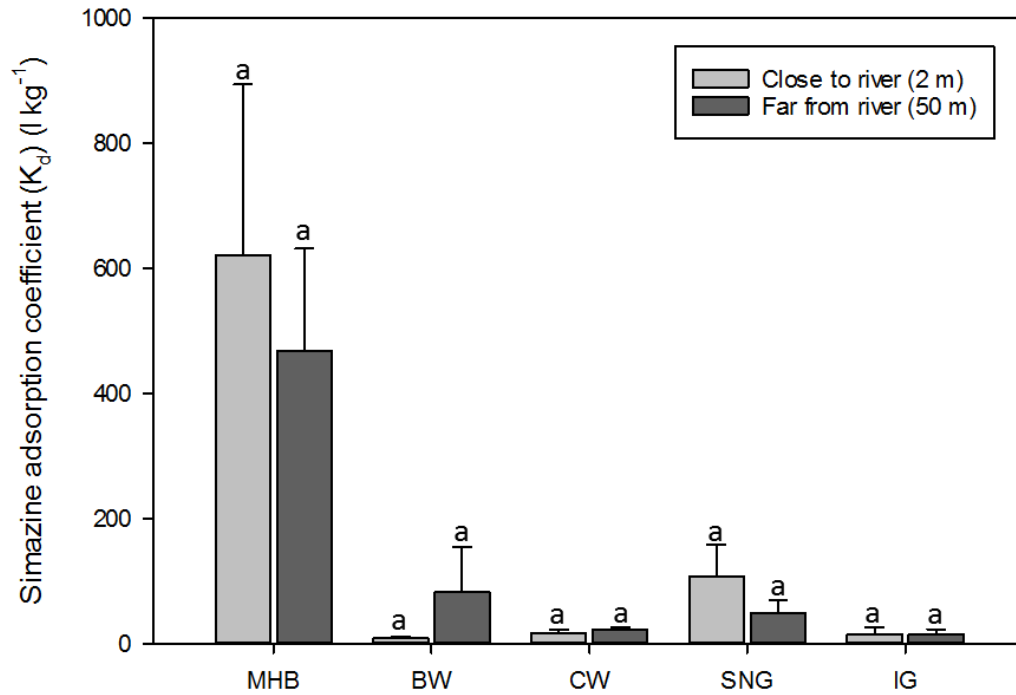
Figure 4. Simazine adsorption coefficient (K_d) across habitat types (MHB: mountain, heath and bog; BW: broadleaf woodland; CW: coniferous woodland; SNG: semi-natural grassland; IG: improved grassland) with respect to distance from the river. Same lower-case letters indicate no significant difference ($P > 0.05$) between distance from the river and simazine adsorption coefficient according to independent t-test within each habitat type. Bars represent mean values ($n = 5$) ± standard error of the mean (SEM).
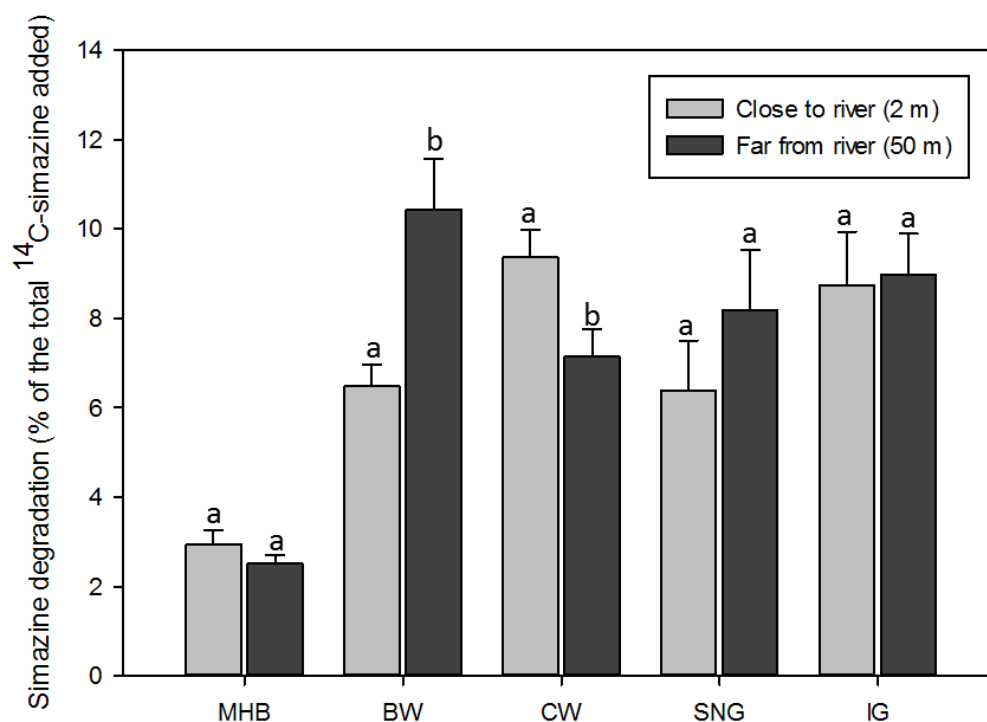
Figure 5. Simazine degradation across habitat types (MHB: mountain, heath and bog; BW: broadleaf woodland; CW: coniferous woodland; SNG: semi-natural grassland; IG: improved grassland) with respect to distance from the river. Values are expressed as the cumulative percentage of the total ^{14}C-simazine added. Same lower-case letters indicate no significant difference ($P > 0.05$) between distance from the river and simazine degradation according to independent t-test within each habitat type. Bars represent mean values ($n = 5$) ± standard error of the mean (SEM).
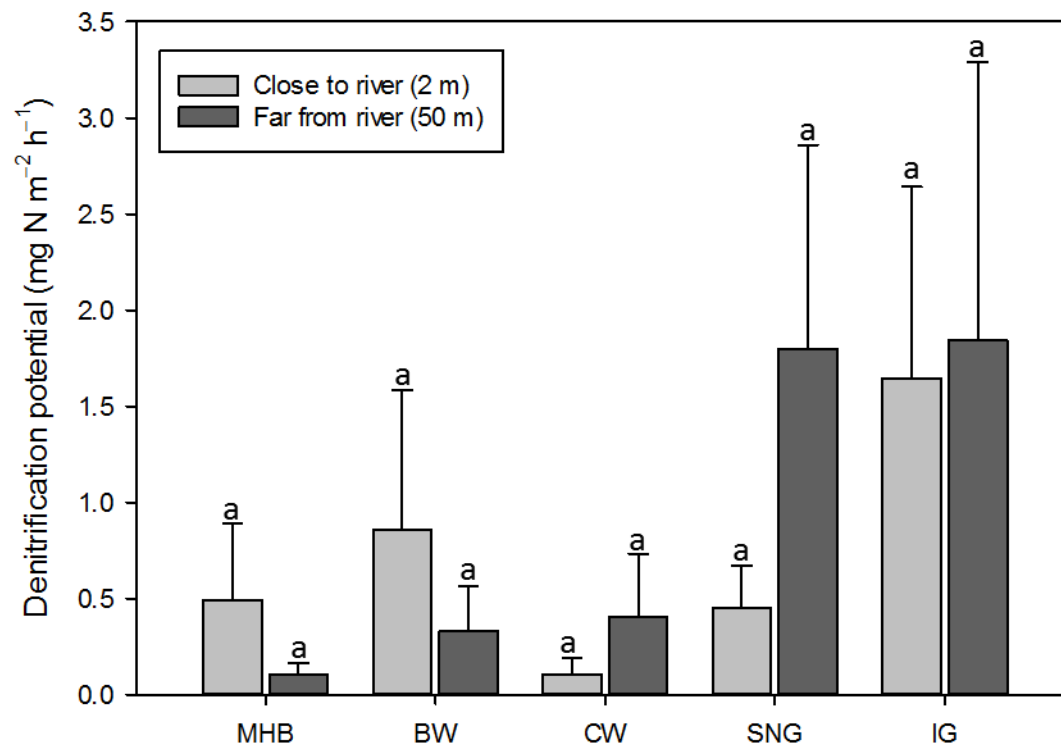
Figure 6. Rate of potential denitrification after 24 h across dominant habitat types (MHB: mountain, heath and bog; BW: broadleaf woodland; CW: coniferous woodland; SNG: semi-natural grassland; IG: improved grassland) w with respect to distance from the river. Same lower case letters indicate no significant differences ($P > 0.05$) respective to distance from the river according to the independent t-test. Bars represent mean values ($n = 5$) $\pm$ standard error of the mean (SEM).
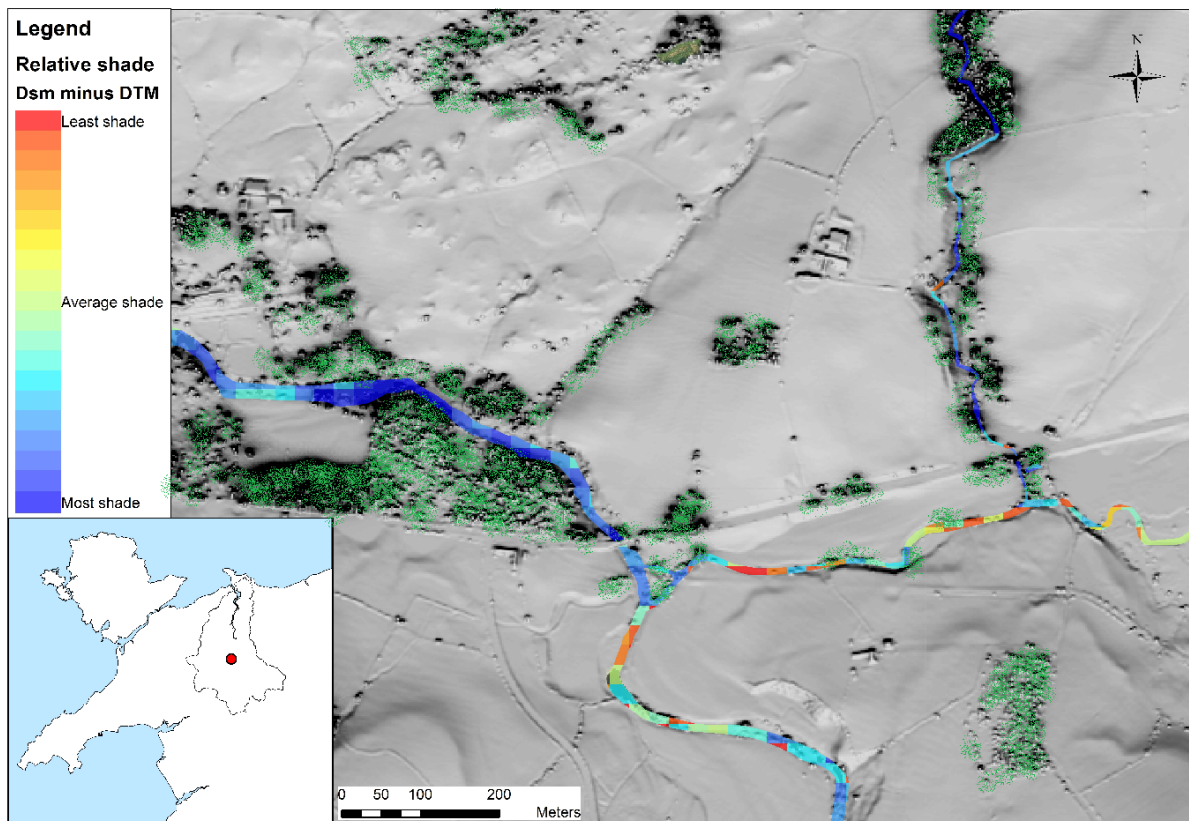
Figure 7. An example image showing the areas with the least (red) and greatest (blue) amount of shade from solar radiation, generated using a Digital Elevation Model (DEM) to represent the bare surface without objects (i.e. vegetation and other objects) and the Digital Surface Model representing the earth's surface including vegetation and other objects. Areas with dense vegetation are coloured in green.
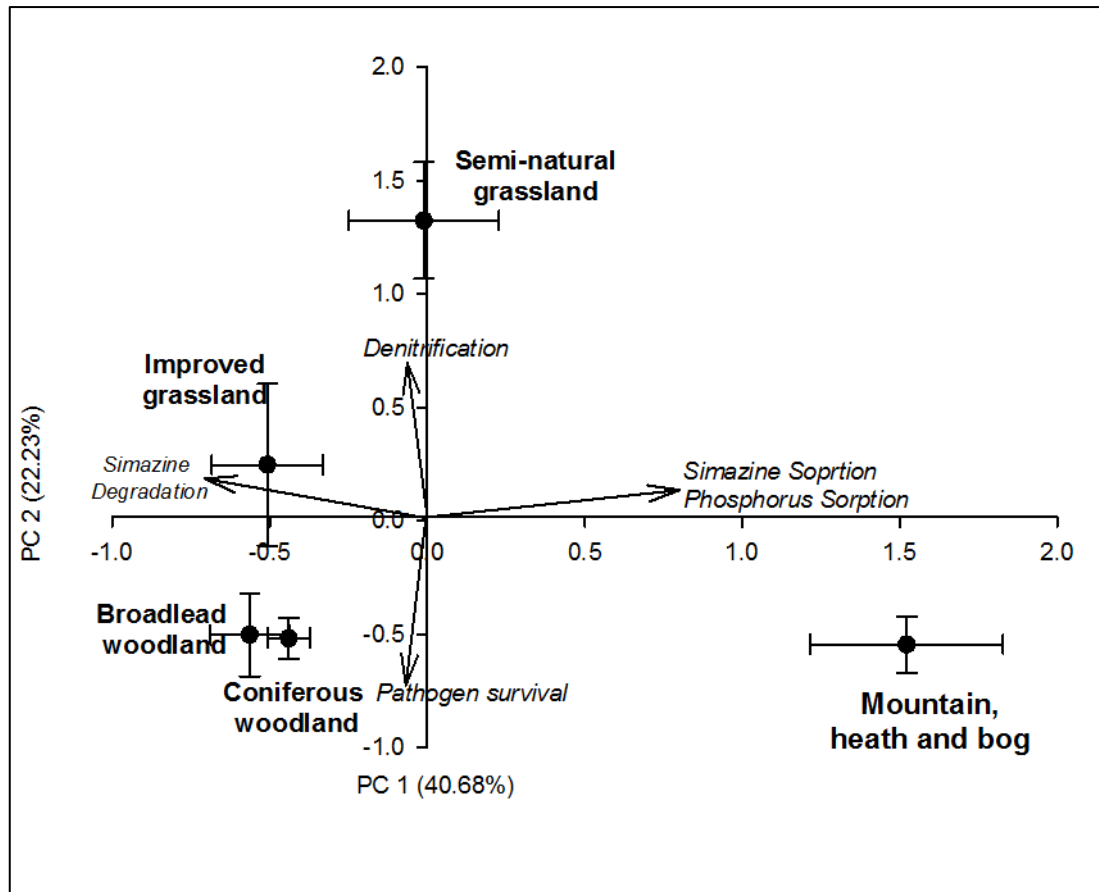
Figure 8. Correlation bi-plot from the principal component analysis (PCA) on ecosystem services evaluated in this study irrespective of the distance from the river. Correlation of ecosystem services with the main axes are given by arrows and habitat types by cluster centroids (average score on each horizontal principal component (PC1) and vertical principal component (PC2) with standards errors, $n = 10$).

Table 1

Summary of riparian soil characteristics and their associated provision of ecosystem services.

Ecosystem services	Causal factor	Resulting soil characteristics
Supporting services Soil formation Nutrient cycling **Regulating services** Water purification by reducing non-point source pollutants Flood and erosion regulation by slowing and spreading flood water	• Periodic sediment deposition together with flushes of organic litter during floods events • Large variation of soil chemical composition mainly due to filtration and nutrient removal from terrestrial upland and aquatic ecosystems	Heterogeneity (Mikkelsen and Vesho, 2000)
Supporting services Biodiversity **Regulating services** Carbon sequestration **Provisioning services** Shading by vegetation	• High vegetation density and diversity associated with higher moisture and organic matter content which leads to more microbial activity • Provide (roots, fallen logs) refuge for aquatic and terrestrial fauna	Biological diversity (Naiman et al., 2010)
Supporting services Soil formation **Regulating services** Carbon sequestration	• New material (organic matter fluxes and sediments) being deposited by flood events and water fluctuation • Regular inundation of soils by river water preventing horizon formation	Undeveloped soils (Zaimes et al., 2007)
Regulating services Water storage	• Their proximity with the river enhances water storage and infiltration	High moisture content (Lewis et al., 2003)
Regulating services Fast engineering resilience[1]	• Anthropogenic activities such as farming, water abstraction, livestock and deforestation • Frequent environmental disturbances such as floods or droughts	Disturbance driven (Klemas, 2014)

[1] Speed with which a system returns to equilibrium after a disturbance (Holling, 1996).

Table 2

Dataset	Scale	Data Type	IPR holder	Description
Digital Terrain Model	2 m	Raster	Natural Resources Wales	This dataset is derived from a combination of all data that is at 2 m resolution or better which has been merged and re-sampled to give the best possible coverage. Available at: https://data.gov.uk/dataset/lidar-terrainand-surfaces-models-wales
Digital Surface Model	2 m	Raster	Natural Resources Wales	This dataset is derived from a combination of all data that is at 2 m resolution or better which has been merged and re-sampled to give the best possible coverage. Available at: https://data.gov.uk/dataset/lidar-terrainand-surfaces-models-wales
OS Open Rivers	1:25,000	Shapefile	Edina Digimap	Water bodies polygons within the catchment.

Data inputs and sources for the computational GIS tool.

Table 3

Main soil physicochemical characteristics for the five different habitat types. Sampling depth and distance from the river were amalgamated together as there was no significant differences from the result of a factorial analysis with habitat, depth and distance as the main factors (see Tables S1-S5). Data are mean values ($n = 10$) $\pm$ standard error of the mean (SEM).

	Mountain, heath and bog (MHB)	Broadland woodland (BW)	Coniferous woodland (CW)	Semi-natural grassland (SNG)	Improved grassland (IG)
pH	4.5 ± 0.1	5.2 ± 0.1	4.6 ± 0.1	5.1 ± 0.1	5.3 ± 0.1
EC (μS cm^{-1})	32.5 ± 3.3	31.8 ± 2.9	35.7 ± 3.6	33.3 ± 3.0	93.1 ± 20.5
Bulk density (g cm^{-3})	0.08 ± 0.01	0.74 ± 0.06	0.43 ± 0.1	0.23 ± 0.06	0.66 ± 0.07
Moisture content (%)	86.6 ± 0.6	32.2 ± 1.5	31.9 ± 3.0	64.1 ± 5.0	35.5 ± 2.7
Organic matter (%)	82.4 ± 2.6	10.6 ± 0.8	14.6 ± 2.2	35.3 ± 5.7	11.4 ± 1.4
NH$_4^+$-N (mg kg^{-1} soil)	18.0 ± 0.76	4.77 ± 0.39	5.06 ± 0.38	12.48 ± 2.21	4.47 ± 0.75
NO$_3^-$-N (mg kg^{-1} soil)	50.3 ± 8.32	3.07 ± 0.47	5.31 ± 0.76	10.6 ± 1.42	12.7 ± 3.14
P available (mg kg^{-1} soil)	4.92 ± 1.28	0.31 ± 0.07	0.32 ± 0.06	0.78 ± 0.14	1.27 ± 0.31
Total C (g kg^{-1} soil)	522 ± 27	54 ± 5	73 ± 12	121 ± 24	149 ± 31
Total N (g kg^{-1} soil)	20.5 ± 1.11	3.45 ± 0.26	4.01 ± 0.55	6.86 ± 1.00	9.10 ± 1.58
Dissolved organic C (g kg^{-1} soil)	1.01 ± 0.11	0.19 ± 0.02	0.27 ± 0.02	0.39 ± 0.05	0.17 ± 0.01
Total dissolved N (g kg^{-1} soil)	0.15 ± 0.02	0.03 ± 0.003	0.03 ± 0.002	0.06 ± 0.01	0.05 ± 0.01
Microbial biomass C (g kg^{-1} soil)	2.31 ± 0.44	0.93 ± 0.07	1.31 ± 0.19	3.58 ± 1.03	1.63 ± 0.22
Microbial biomass N (g kg^{-1} soil)	0.34 ± 0.07	0.23 ± 0.03	0.16 ± 0.02	0.47 ± 0.09	0.29 ± 0.04

Table 4

Maximum adsorption values (S_{max}), binding energy constant (k) and correlation coefficients (R^2) as estimated by Langmuir isotherm with respect to distance from the river. Data are mean values ($n = 5$) ± standard error of the mean (SEM).

| | Langmuir model | | | | |
| | Maximum P sorption S_{max} (mg kg^{-1}) | | Binding strength k (l kg^{-1}) | | R^2 |
	Close to river	Far from river	Close to river	Far from river	
Mountain, heath and bog (MHB)	379 ± 74	385 ± 137	3.6 ± 2.5	7.3 ± 5.1	0.90 ± 0.03
Broadleaf woodland (BW)	88 ± 10	82 ± 7	42.2 ± 8.0	28.7 ± 9.6	0.87 ± 0.04
Coniferous woodland (CW)	81 ± 6	114 ± 15	31.6 ± 5.3	25.3 ± 5.1	0.91 ± 0.04
Semi-natural grassland (SNG)	246 ± 62	172 ± 55	22.8 ± 8.1	23.7 ± 6.8	0.95 ± 0.04
Improved grassland (IG)	148 ± 68	86 ± 9	14.6 ± 5.1	19.9 ± 3.2	0.97 ± 0.01

Quantifying the contribution of riparian soils to the provision of ecosystem services

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Supplementary on-line information

Table S1. Soil physicochemical properties in mountain, heath and bog (MHB) land use type with respect to the distance from the river and soil depth in the Conwy Catchment. Data are mean values ($n = 5$) ± standard error of the mean (SEM). Significant differences are shown according to two-way ANOVA (One-way ANOVA for bulk density) with distance and depth as main factors. No interactions between depth and distance were found in the analysis. No significant differences were found by the interaction of distance with depth.

| | Riparian distance | | | | *P*-values | |
| | Close to river (2 m) | | Far from river (50 m) | | | |
	0-15 cm	15-30 cm	0-15 cm	15-30 cm	Distance	Depth
pH	4.85 ± 0.40	4.92 ± 0.40	4.34 ± 0.20	4.46 ± 0.20	ns	ns
EC (µS cm^{-1})	33.2 ± 6.3	26.8 ± 5.4	37.1 ± 4.4	24.0 ± 5.0	ns	ns
Bulk density (g cm^{-3})	0.07 ± 0.01	ND	0.09 ± 0.02	ND	ns	ND
Moisture content (%)	87.7 ± 0.8	87.4 ± 0.5	87.4 ± 1.7	84.2 ± 1.1	ns	ns
Organic matter (%)	78.7 ± 6.8	86.1 ± 5.6	86.3 ± 3.5	78.6 ± 5.9	ns	ns
NH$_4^+$-N (mg kg^{-1} soil)	19.8 ± 1.3	18.4 ± 1.2	20.7 ± 4.0	18.1 ± 1.2	ns	ns
NO$_3^-$-N (mg kg^{-1} soil)	51.5 ± 18.7	50.5 ± 19.3	56.8 ± 15.1	42.5 ± 12.1	ns	ns
Available P (mg kg^{-1} soil)	10.8 ± 4.04	3.11 ± 1.49	3.42 ± 0.53	2.29 ± 0.72	0.002	ns
Total C (g kg^{-1} soil)	453 ± 102	456 ± 147	545 ± 30	524 ± 40	ns	ns
Total N (g kg^{-1} soil)	17.8 ± 3.1	21.6 ± 1.8	13.8 ± 4.5	21.1 ± 2.2	ns	ns
Dissolved organic C (g kg^{-1} soil)	0.95 ± 0.30	1.00 ± 0.30	1.07 ± 0.20	1.01 ± 0.20	ns	ns
Total dissolved N (g kg^{-1} soil)	0.14 ± 0.03	0.16 ± 0.04	0.17 ± 0.04	0.14 ± 0.01	ns	ns
Microbial biomass C (g kg^{-1} soil)	3.20 ± 0.89	1.04 ± 0.41	3.81 ± 1.07	1.20 ± 0.19	ns	0.005
Microbial biomass N (g kg^{-1} soil)	0.26 ± 0.11	0.28 ± 0.11	0.43 ± 0.24	0.38 ± 0.08	ns	ns

EC, electrical conductivity; ND, not determined.

Table S2. Soil physicochemical properties in broadleaf woodland (BW) land use type with respect to the distance from the river and depth in the Conwy Catchment. Data are mean values ($n = 5$) ± standard error of the mean (SEM). Significant differences are shown according to two-way ANOVA (One-way ANOVA for bulk density) with distance and depth as main factors. No interactions between depth and distance were found in the analysis. No

	Riparian distance				P-values	
	Close to river (2 m)		Far from river (50 m)			
	0-15 cm	15-30 cm	0-15 cm	15-30 cm	Distance	Depth
pH	5.14 ± 0.30	5.18 ± 0.20	5.07 ± 0.30	5.24 ± 0.30	ns	ns
EC (µS cm^{-1})	26.6 ± 5.0	25.2 ± 4.2	42.9 ± 6.2	31.5 ± 5.4	0.047	ns
Bulk density (g cm^{-3})	0.74 ± 0.11	ND	0.73 ± 0.06	ND	ns	ND
Moisture content (%)	30.0 ± 3.0	27.2 ± 5.0	41.0 ± 7.8	34.3 ± 2.8	ns	ns
Organic matter (%)	14.3 ± 4.8	8.4 ± 1.9	24.8 ± 12.5	10.1 ± 0.7	ns	ns
NH$_4^+$-N (mg kg^{-1} soil)	3.75 ± 0.8	4.25 ± 0.7	6.37 ± 0.5	4.70 ± 0.8	0.042	ns
NO$_3^-$-N (mg kg^{-1} soil)	1.99 ± 0.6	1.77 ± 1.1	7.01 ± 1.6	3.49 ± 1.0	0.004	ns
P available (mg kg^{-1} soil)	0.31 ± 0.11	0.41 ± 0.20	0.57 ± 0.28	0.19 ± 0.12	ns	ns
Total C (g kg^{-1} soil)	57 ± 13	44 ± 10	76 ± 8	42 ± 6	ns	ns
Total N (g kg^{-1} soil)	3.38 ± 0.60	4.47 ± 0.30	2.72 ± 0.40	3.21 ± 0.20	0.016	ns
Dissolved organic C (g kg^{-1} soil)	0.19 ± 0.05	0.19 ± 0.05	0.26 ± 0.06	0.14 ± 0.02	ns	ns
Total dissolved N (g kg^{-1} soil)	0.03 ± 0.01	0.03 ± 0.01	0.04 ± 0.005	0.02 ± 0.002	ns	ns
Microbial biomass C (g kg^{-1} soil)	0.26 ± 0.11	0.28 ± 0.11	0.43 ± 0.24	0.38 ± 0.08	ns	ns
Microbial biomass N (g kg^{-1} soil)	0.16 ± 0.03	0.18 ± 0.02	0.26 ± 0.03	0.32 ± 0.11	0.024	ns

significant differences were found by the interaction of distance with depth.

EC, electrical conductivity; ND, not determined.

Table S3. Soil physicochemical properties in coniferous woodland (CW) land use type with respect to the distance from the river and depth in the Conwy Catchment. Data are mean values ($n = 5$) ± standard error of the mean (SEM). Significant differences are shown according to two way ANOVA (One way ANOVA for bulk density) with distance and depth as main factors. No interactions between depth and distance were found in the analysis. No significant differences were found by the interaction of distance with depth.

	Riparian distance				P-values	
	Close to river (2 m)		Far from river (50 m)			
	0-15 cm	15-30 cm	0-15 cm	15-30 cm	Distance	Depth
pH	4.75 ± 0.20	4.95 ± 0.10	4.23 ± 0.10	4.52 ± 0.10	0.002	ns
EC (µS cm^{-1})	28.9 ± 4.8	27.0 ± 3.2	43.6 ± 7.5	45.0 ± 10.1	ns	ns
Bulk density (g cm^{-3})	0.45 ± 0.15	ND	0.41 ± 0.16	ND	ns	ND
Moisture content (%)	36.4 ± 9.9	36.2 ± 10.7	39.3 ± 5.8	32.9 ± 7.5	ns	ns
Organic matter (%)	13.5 ± 5.8	12.9 ± 6.6	18.9 ± 3.4	13.3 ± 1.6	ns	ns
NH$_4^+$-N (mg kg^{-1} soil)	5.62 ± 0.90	4.79 ± 0.60	5.08 ± 0.90	4.75 ± 0.80	ns	ns
NO$_3^-$-N (mg kg^{-1} soil)	4.95 ± 1.2	4.11 ± 1.4	7.54 ± 2.2	4.63 ± 5.9	ns	ns
Available P (mg kg^{-1} soil)	0.27 ± 0.08	0.34 ± 0.20	0.40 ± 0.08	0.28 ± 0.03	ns	ns
Total C (g kg^{-1} soil)	71 ± 33	56 ± 36	109 ± 13	58 ± 11	ns	ns
Total N (g kg^{-1} soil)	4.21 ± 1.40	5.38 ± 0.50	3.32 ± 1.60	3.11 ± 0.40	ns	ns
Dissolved organic C (g kg^{-1} soil)	0.22 ± 0.04	0.22 ± 0.04	0.32 ± 0.03	0.33 ± 0.03	0.011	ns
Total dissolved N (g kg^{-1} soil)	0.03 ± 0.004	0.03 ± 0.005	0.04 ± 0.004	0.04 ± 0.004	ns	ns
Microbial biomass C (g kg^{-1} soil)	1.09 ± 0.38	0.85 ± 0.41	2.15 ± 0.23	1.15 ± 0.28	ns	ns
Microbial biomass N (g kg^{-1} soil)	0.20 ± 0.05	0.10 ± 0.02	0.22 ± 0.03	0.13 ± 0.04	ns	0.019

EC, electrical conductivity; ND, not determined.

Table S4. Soil physicochemical properties in semi-natural grassland (SNG) land use type with respect to the distance from the river and depth in the Conwy Catchment. Data are mean values ($n = 5$) ± standard error of the mean (SEM). Significant differences are shown according to two way ANOVA (One way ANOVA for bulk density) with distance and depth as main factors. No interactions between depth and distance were found in the analysis. No significant differences were found by the interaction of distance with depth.

	Riparian distance				P-values	
	Close to river (2 m)		Far from river (50 m)			
	0-15 cm	15-30 cm	0-15 cm	15-30 cm	Distance	Depth
pH	4.95 ± 0.20	5.07 ± 0.10	5.25 ± 0.40	5.27 ± 0.20	ns	ns
EC (µS cm^{-1})	35.1 ± 5.3	26.9 ± 4.6	44.4 ± 8.4	28.1 ± 4.6	ns	ns
Bulk density (g cm^{-3})	0.16 ± 0.05	ND	0.31 ± 0.12	ND	ns	ND
Moisture content (%)	73.0 ± 7.6	68.9 ± 10.1	62.7 ± 9.3	51.7 ± 13.0	ns	ns
Organic matter (%)	41.4 ± 11.6	39.9 ± 12.2	33.9 ± 11.4	25.9 ± 13.2	ns	ns
NH$_4^+$-N (mg kg^{-1} soil)	15.5 ± 4.9	14.1 ± 4.4	12.9 ± 5.9	7.40 ± 2.3	ns	ns
NO$_3^-$-N (mg kg^{-1} soil)	14.6 ± 5.6	14.7 ± 4.2	13.7 ± 5.1	9.10 ± 1.9	ns	ns
Available P (mg kg^{-1} soil)	1.06 ± 0.36	0.64 ± 0.25	0.63 ± 0.21	0.57 ± 0.24	ns	ns
Total C (g kg^{-1} soil)	74 ± 35	218 ± 67	101 ± 25	83.3 ± 20	ns	ns
Total N (g kg^{-1} soil)	5.47 ± 1.9	7.46 ± 1.5	11.03 ± 3.7	12.28 ± 4.0	ns	ns
Dissolved organic C (g kg^{-1} soil)	0.40 ± 0.10	0.41 ± 0.14	0.42 ± 0.10	0.35 ± 0.1	ns	ns
Total dissolved N (g kg^{-1} soil)	0.07 ± 0.02	0.07 ± 0.02	0.07 ± 0.01	0.06 ± 0.008	ns	ns
Microbial biomass C (g kg^{-1} soil)	6.84 ± 2.40	5.50 ± 2.68	1.05 ± 0.38	0.94 ± 0.30	0.050	ns
Microbial biomass N (g kg^{-1} soil)	0.90 ± 0.23	0.29 ± 0.08	0.43 ± 0.11	0.27 ± 0.10	ns	0.014

EC, electrical conductivity; ND, not determined.

Table S5. Soil physicochemical properties in improved grassland (IG) land use type with respect to the distance from the river and depth in the Conwy Catchment. Data are mean values (n = 5) ± standard error of the mean (SEM). Significant differences are shown according to two way ANOVA (One way ANOVA for bulk density) with distance and depth as main factors. No significant differences were found by the interaction of distance with depth.

	Riparian distance				P-values	
	Close to river (2 m)		Far from river (50 m)			
	0-15 cm	15-30 cm	0-15 cm	15-30 cm	Distance	Depth
pH	5.19 ± 0.30	5.28 ± 0.30	5.39 ± 0.10	5.43 ± 0.20	ns	ns
EC (μS cm^{-1})	104 ± 37	34 ± 7	131 ± 55	101 ± 47	ns	ns
Bulk density (g cm^{-3})	0.60 ± 0.11	ND	0.71 ± 0.10	ND	ns	ND
Moisture content (%)	39.0 ± 6.9	35.4 ± 8.9	44.0 ± 5.3	30.6 ± 2.8	ns	ns
Organic matter (%)	13.3 ± 3.7	12.6 ± 6.3	20.0 ± 4.4	10.0 ± 2.0	ns	ns
NH$_4^+$-N (mg kg^{-1} soil)	5.18 ± 1.7	3.42 ± 1.1	5.87 ± 2.1	3.39 ± 1.1	ns	ns
NO$_3^-$.N (mg kg^{-1} soil)	9.78 ± 3.4	6.96 ± 1.8	22.7 ± 9.1	21.4 ± 12.1	ns	ns
Available P (mg kg^{-1} soil)	2.08 ± 1.06	1.05 ± 0.55	1.84 ± 0.75	0.93 ± 0.48	ns	ns
Total C (g kg^{-1} soil)	270 ± 65	87 ± 59	223 ± 65	56 ± 8	ns	0.001
Total N (g kg^{-1} soil)	14.8 ± 3.4	14.2 ± 3.3	3.31 ± 0.5	6.10 ± 1.9	0.017	ns
Dissolved organic C (g kg^{-1} soil)	0.17 ± 0.02	0.18 ± 0.05	0.20 ± 0.02	0.15 ± 0.02	ns	ns
Total dissolved N (g kg^{-1} soil)	0.04 ± 0.01	0.04 ± 0.01	0.07 ± 0.01	0.05 ± 0.02	ns	ns
Microbial biomass C (g kg^{-1} soil)	1.90 ± 0.55	1.54 ± 0.77	2.49 ± 0.31	1.19 ± 0.20	ns	ns
Microbial biomass N (g kg^{-1} soil)	0.18 ± 0.04	0.30 ± 0.10	0.38 ± 0.06	0.31 ± 0.11	ns	ns

EC, electrical conductivity; ND, not determined.

Table S6. Controlling factors affecting the performance of the ecosystem services selected in this study, accompanied by unmeasured factors that mostly likely influence the behaviour of riparian areas in accomplishing ecosystem functioning.

Ecosystem service	Habitat physicochemical property found	Process likely to occur in riparian areas affecting the delivery of the ecosystem services
Phosphorus and simazine sorption	Organic matter Moisture content Bulk density Available forms of N and P Microbial biomass[1] C content	Erosion processes Rapid uptake by macrophytes Fluxes of organic matter from upland and streams creating 'hot moments' Changes in moisture content and pH controlling pollutant solubility
Simazine degradation	Microbial competition and specialisation pH Total carbon	Changes in pH and redox potential which control pesticide hydrolysis and bioavailability
Denitrification activity	High spatial variation Bulk density pH	Carbon and nitrogen sources provided by the stream Oscillation of anoxic and oxic conditions due to hydrographic regime
Pathogen survival	-	More exposure to animal waste events due to livestock attraction to watercourses
Shade provision	Habitat type canopy	Land change use

[1]Controlling factor only identified for P adsorption

Aerial photographs sample points

1. Aerial photograph of sample point nº 1 within the broadleaf woodland habitat type.

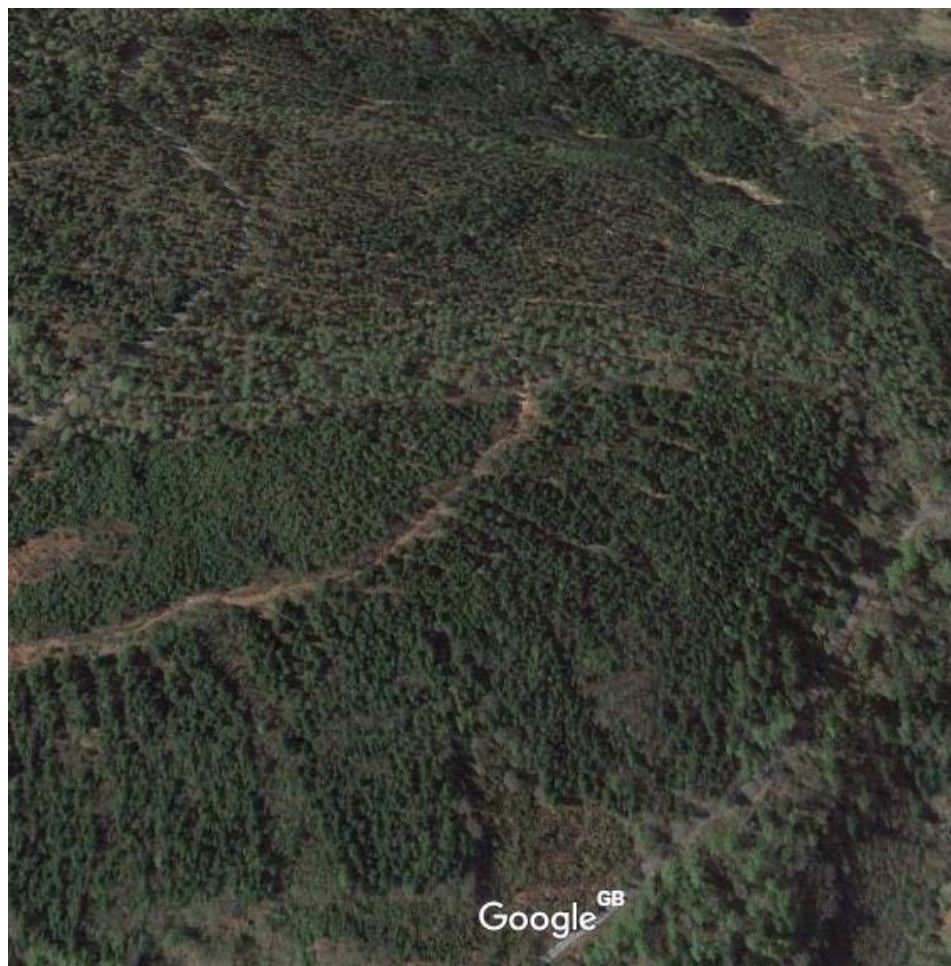2. Aerial photograph of sample point nº 2 within the broadleaf woodland habitat type.
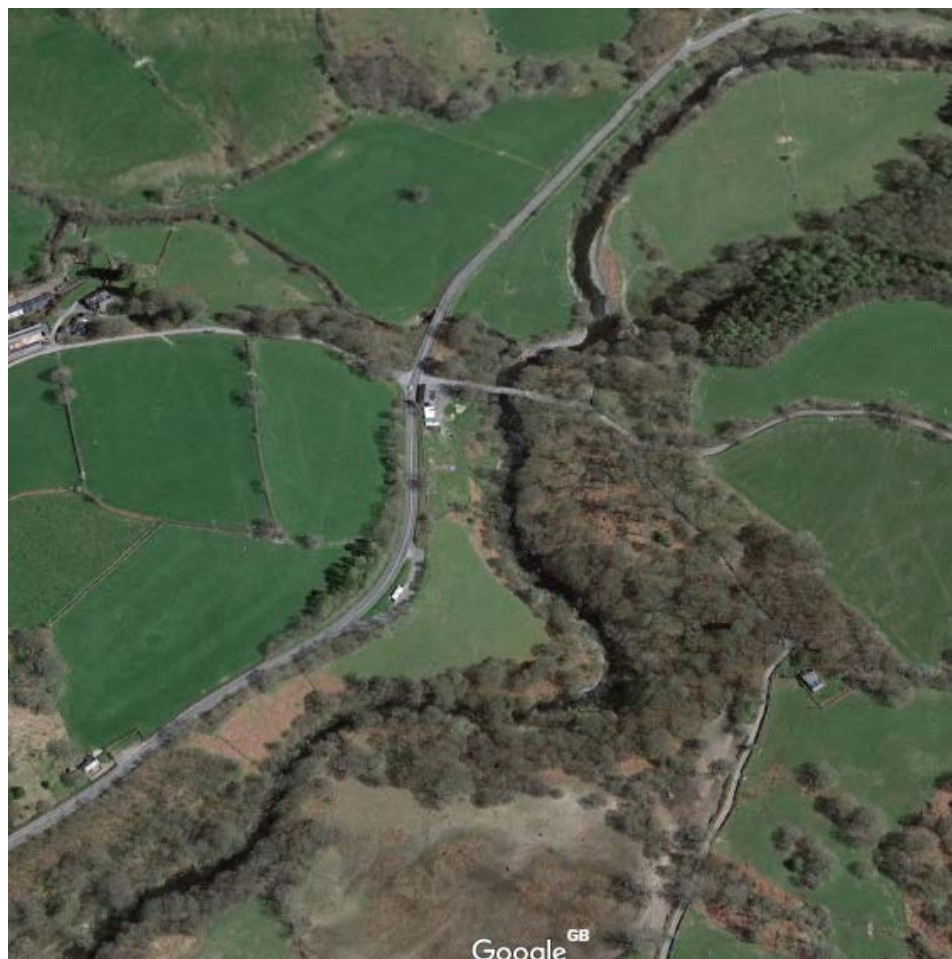
3. Aerial photograph of sample point nº 3 within the broadleaf woodland habitat type.
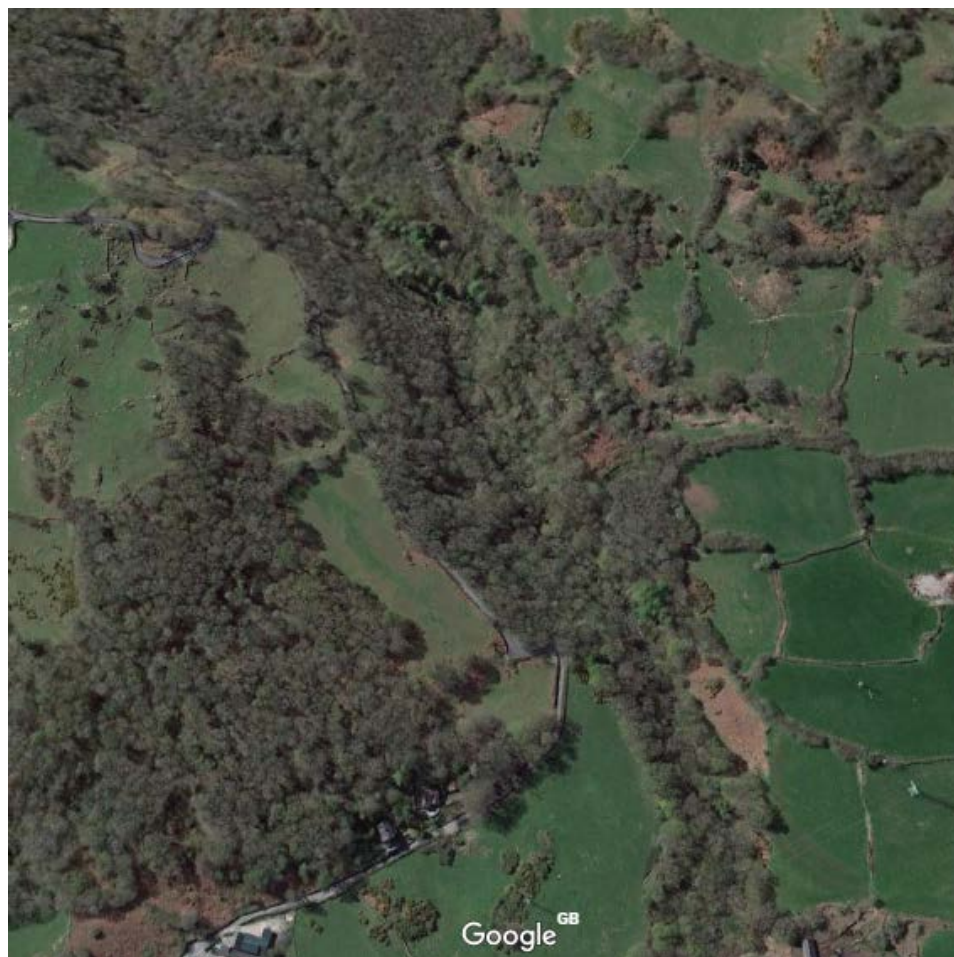
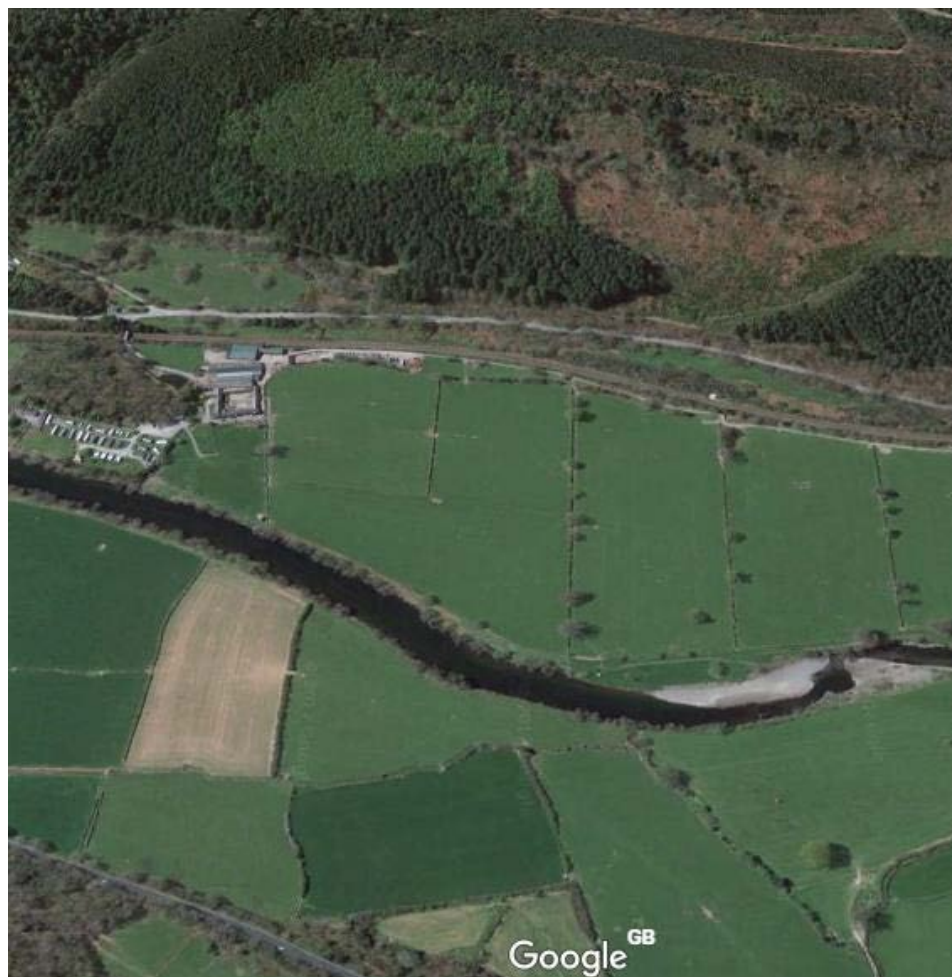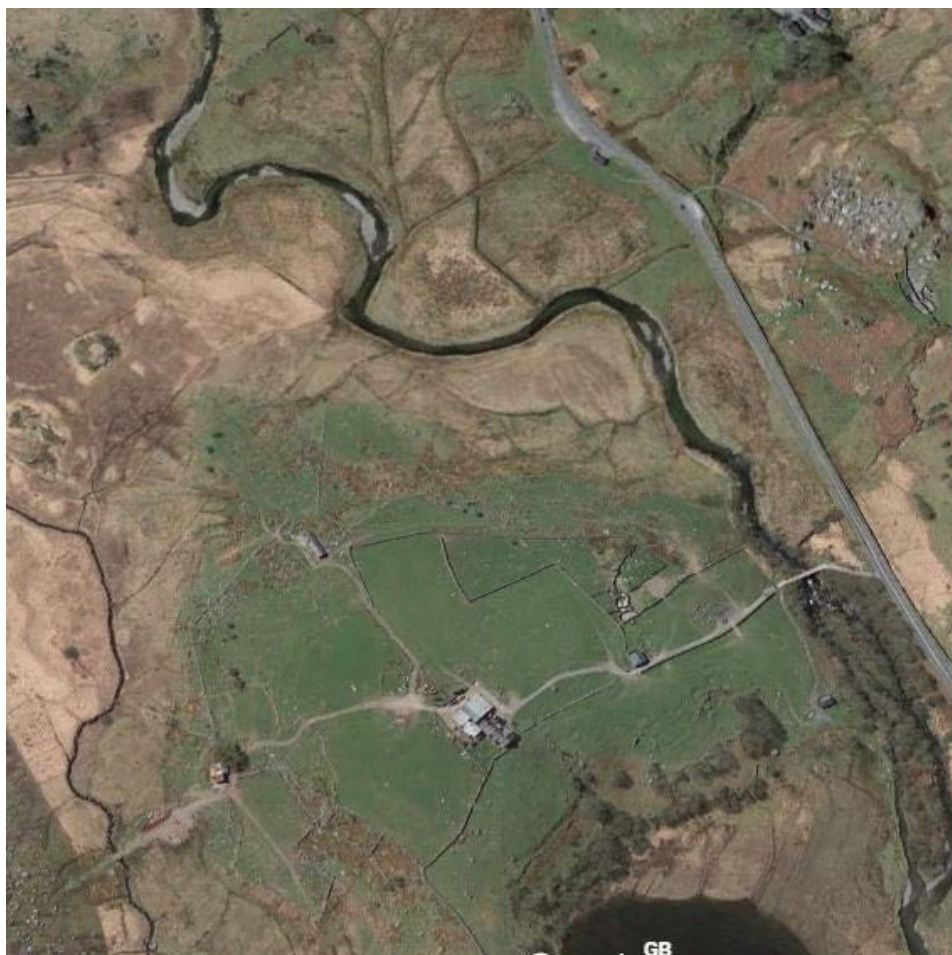4. Aerial photograph of sample point nº 4 within the broadleaf woodland habitat type.

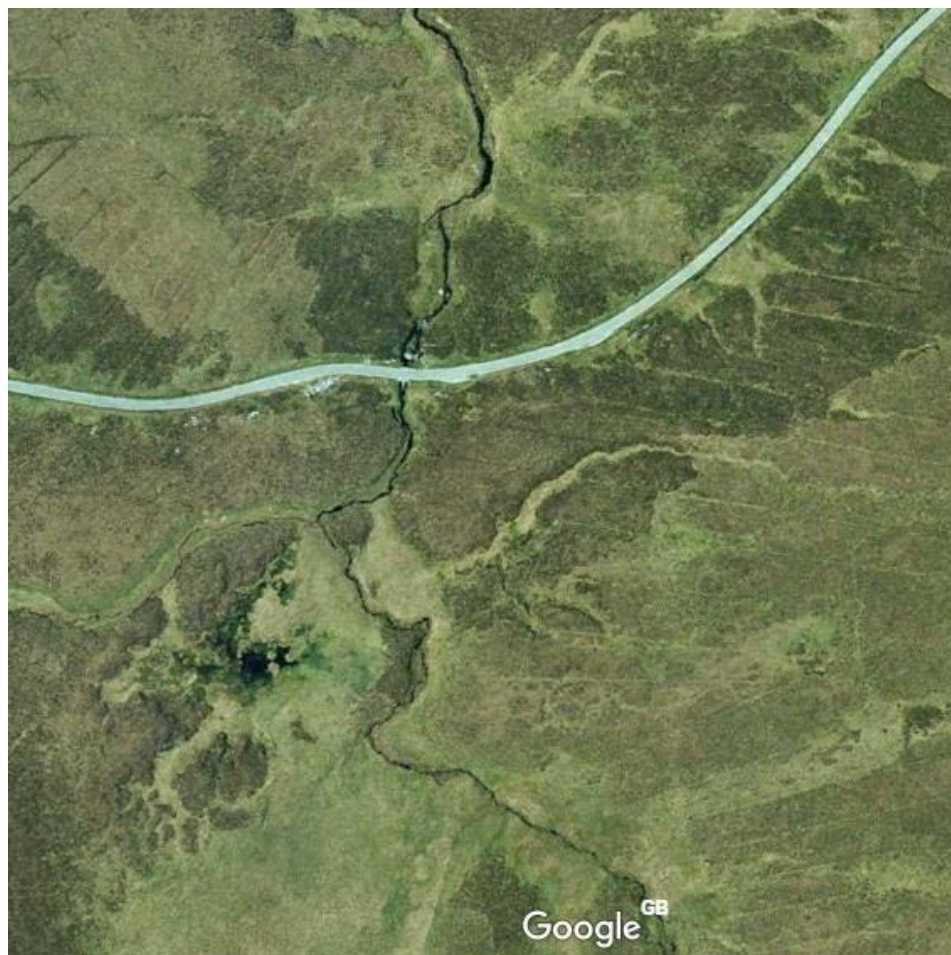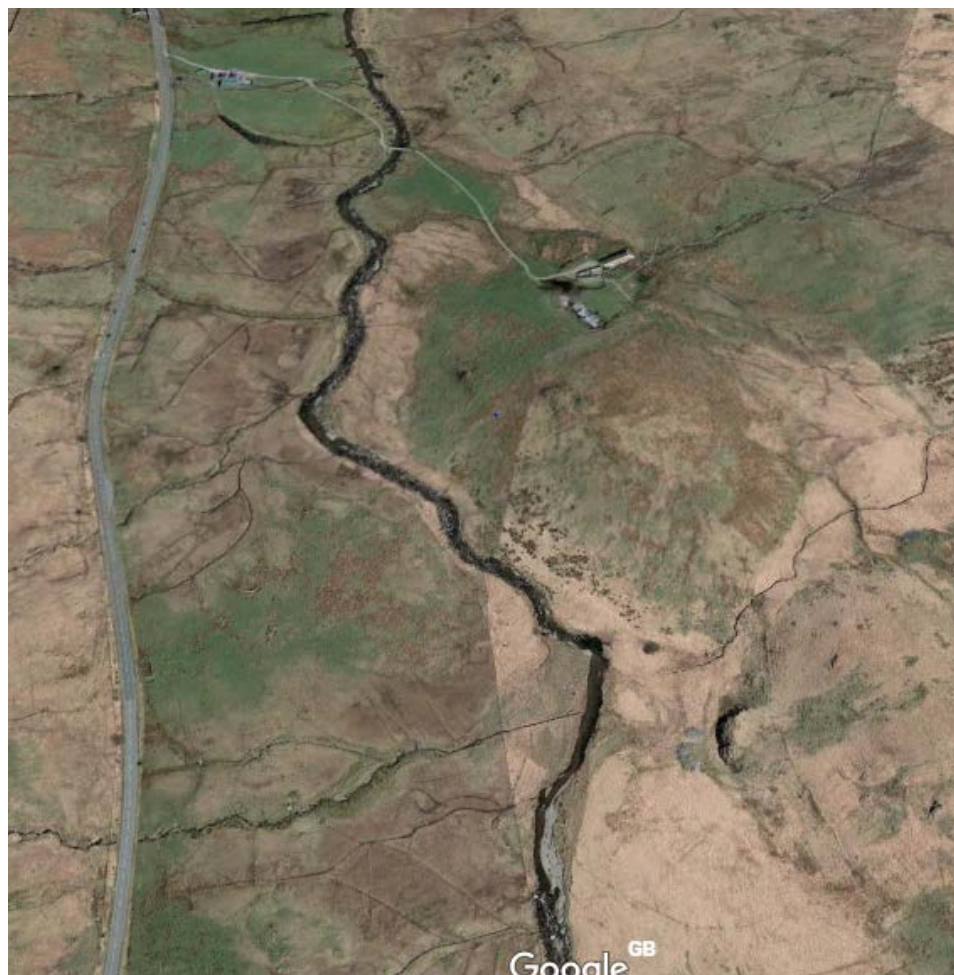5. Aerial photograph of sample point nº 5 within the broadleaf woodland habitat type.

1. Aerial photograph of sample point n° 1 within the coniferous woodland habitat type.

2. Aerial photograph of sample point n° 2 within the coniferous woodland habitat type.

3. Aerial photograph of sample point nº 3 within the coniferous woodland habitat type.

4. Aerial photograph of sample point n° 4 within the coniferous woodland habitat type.

5. Aerial photograph of sample point nº 5 within the coniferous woodland habitat type.

1. Aerial photograph of sample point n° 1 within the improved grassland habitat type.

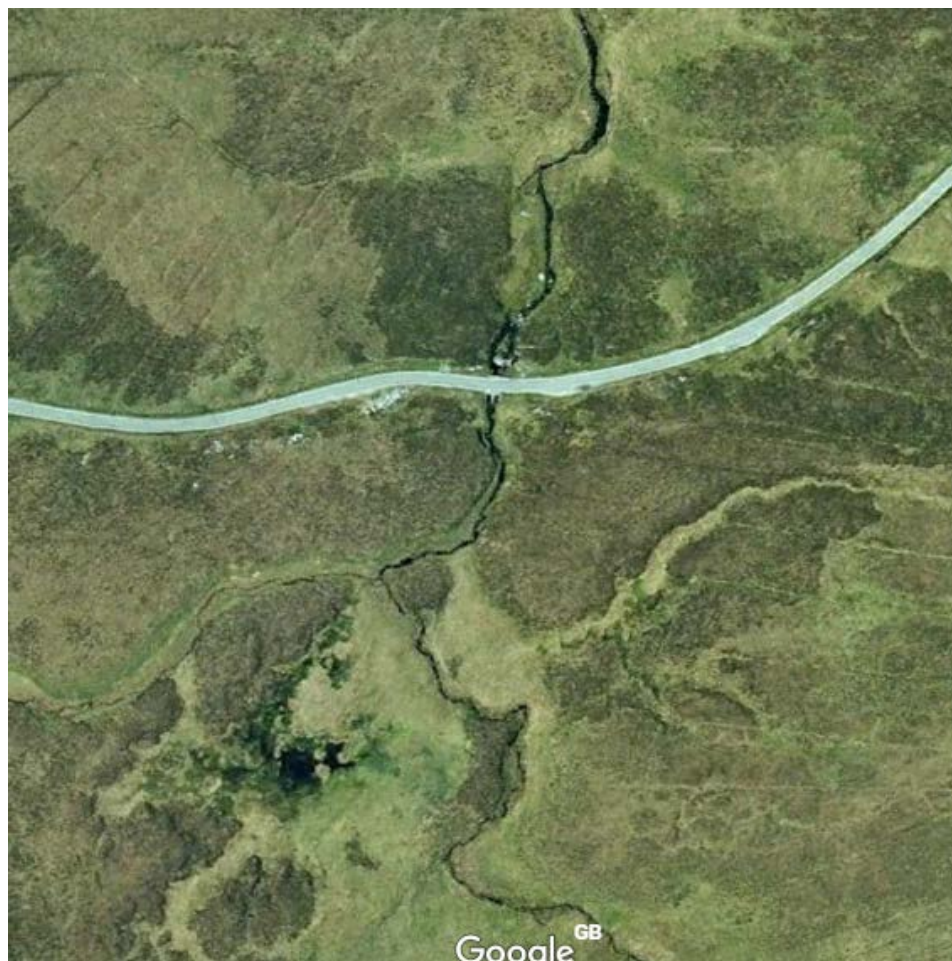2. Aerial photograph of sample point nº 2 within the improved grassland habitat type.

3. Aerial photograph of sample point n° 3 within the improved grassland habitat type.

4. Aerial photograph of sample point n° 4 within the improved grassland habitat type.

5. Aerial photograph of sample point nº 5 within the improved grassland habitat type.

1. Aerial photograph of sample point nº 1 within the mountain, heath and bog habitat type.

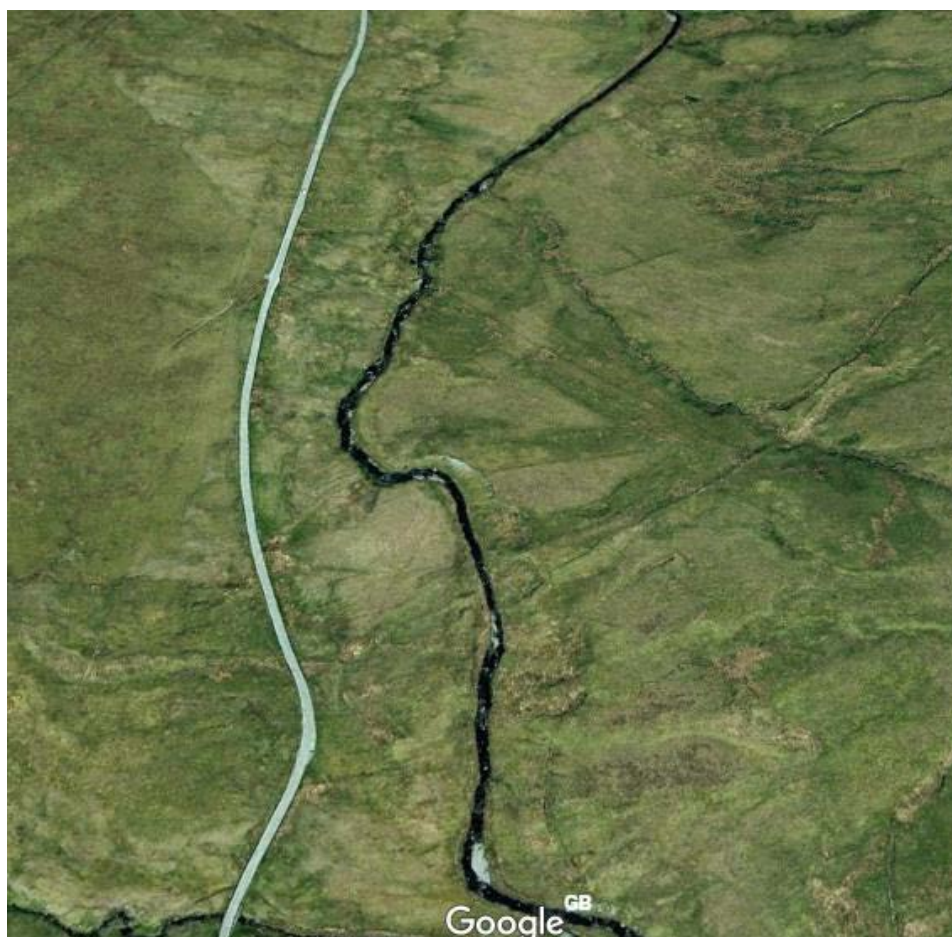2. Aerial photograph of sample point nº 2 within the mountain, heath and bog habitat type.

3. Aerial photograph of sample point nº 3 within the mountain, heath and bog habitat type.

4. Aerial photograph of sample point n° 4 within the mountain, heath and bog habitat type.
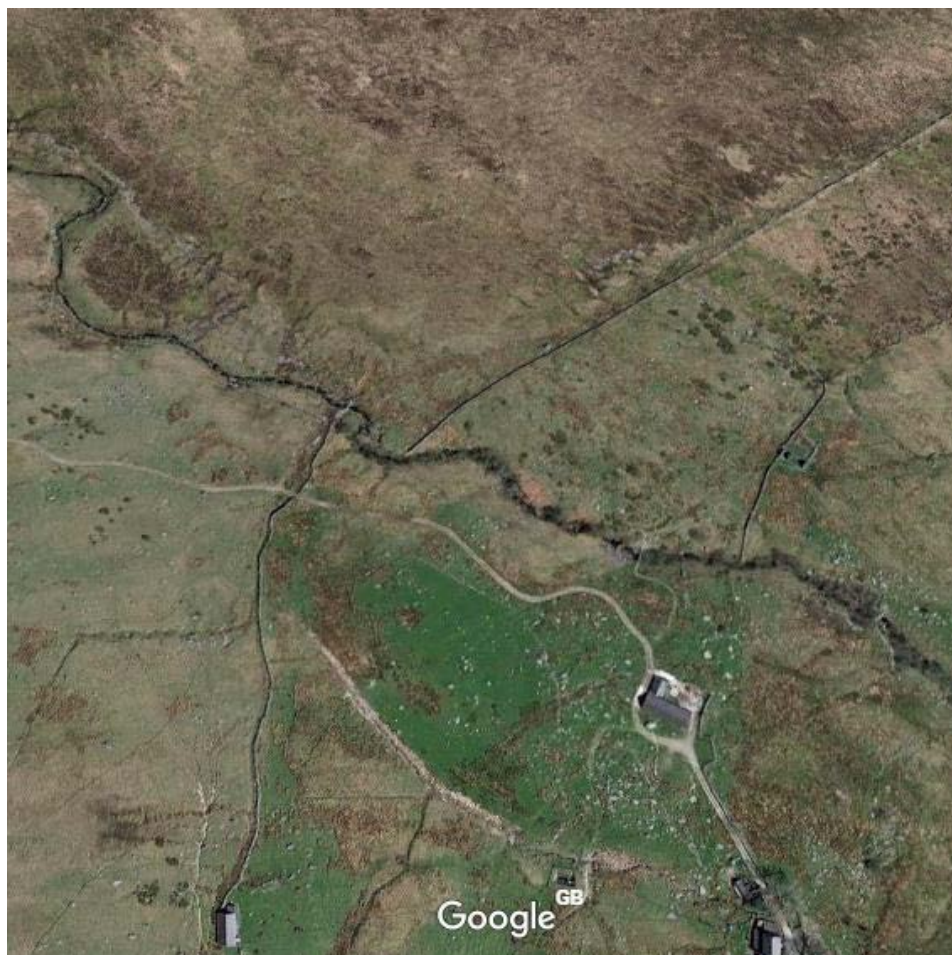
5. Aerial photograph of sample point nº 5 within the mountain, heath and bog habitat type.

1. Aerial photograph of sample point nº 1 within the semi-natural grassland habitat type.
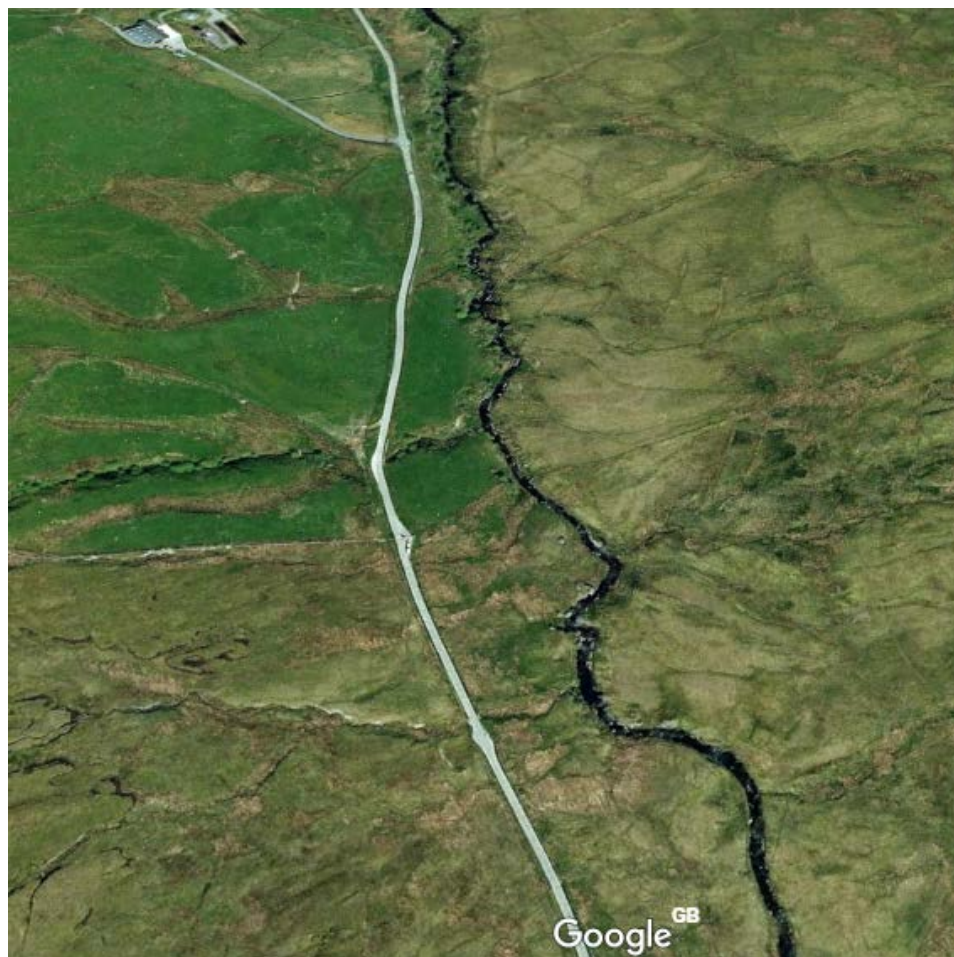
2. Aerial photograph of sample point nº 2 within the semi-natural grassland habitat type.

3. Aerial photograph of sample point n° 3 within the semi-natural grassland habitat type.

4. Aerial photograph of sample point n° 4 within the semi-natural grassland habitat type.

5. Aerial photograph of sample point n° 5 within the semi-natural grassland habitat type.