

## Soil bioengineering techniques enhance riparian habitat quality and multi-taxonomic diversity in the foothills of the Alps and Jura Mountains

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- 1 Soil bioengineering techniques enhance riparian habitat quality and multi-taxonomic
- 2 diversity in the foothills of the Alps and Jura Mountains
- 3
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#### 10 Abstract

11 Riparian zones have disproportional ecological importance relative to their size. For decades, the functionality of riparian zones has been altered, with detrimental consequences on 12 13 biodiversity. Recently, riparian zone restoration has become a major issue. When channel mobility cannot be restored and when erosion control is of primary concern, soil 14 bioengineering techniques are often viewed as a compromise solution. We studied 37 15 riverbanks, from civil engineering to soil bioengineering, plus natural willow stands, in the 16 17 foothills of the Alps and Jura Mountains. Using a principal component analysis, we first studied whether terrestrial and aquatic habitat variables varied among riverbank stabilization 18 19 structures and bank stabilization age and built a synthetic index of riparian habitat quality 20 reflecting the multivariate similarity of riverbank sites. Then, using a modelling approach, we 21 tested whether multi-taxonomic diversity responded to changes in habitat quality and to broad-scale environmental variables (i.e., climate, hydrology and land cover). Soil 22 23 bioengineering techniques, especially willow fascines and to lower extend vegetated crib wall, 24 enhanced riparian habitat quality by allowing for a greater richness and density of pioneer tree 25 species but also for a larger cover of high quality aquatic micro-habitats. This increase in riparian habitat quality induced an increase in both terrestrial and aquatic species diversity, 26 27 highlighting the added-value of soil bioengineering techniques to restore riparian biodiversity. 28 This may confirm that stabilization structures made of willow fascines are better suited than 29 stabilization structures made of artificial substrata to support riparian species. Also, beyond 30 the positive effect of soil bioengineering techniques for riparian biodiversity, we found that 31 climatic, hydrological and land cover variables strongly influenced diversity patterns. Thus, 32 multi-taxonomic diversity decreased along larger rivers and in landscapes dominated by urban 33 areas. This may indicate that the full added value of soil bioengineering techniques for 34 biodiversity will only become apparent if more attention is paid to mitigating the negative

- 35 impact of human activities in the vicinity of riparian zones and if larger scale environmental
- 36 parameters are taken into account as early as possible in restoration project. Therefore, we
- 37 strongly recommend that riverbank restoration projects, based on the active introduction of
- 38 native pioneer tree species, should be planned at the catchment scale.
- 39
- 40 Keywords: biodiversity patterns, riparian habitat quality, soil bioengineering techniques,
- 41 ecological restoration, riverbank stabilization

#### 42 **1. Introduction**

43 Riparian zones, i.e., ecosystems at the interface between terrestrial and freshwater habitats, are small natural features with disproportional ecological importance relative to their size 44 45 (González et al., 2017). Indeed, though riparian zones represent only 1 % of the European continental area (Weissteiner et al., 2016), they host a unique species pool and are of critical 46 47 concern for biodiversity conservation (Naiman and Decamps, 1997; Sabo et al., 2005). Moreover, riparian zones insure a wide range of functions (e.g., water regulation, soil 48 49 retention) which directly improve human well-being by providing many ecosystem services 50 (e.g., flood control, water quality) (Naiman and Decamps, 1997). In this context, restoring or 51 conserving the ecological integrity of riparian zones is of great importance (Strayer and 52 Dudgeon, 2010). 53 Worldwide, riparian zones have been severly impacted by human activities (Feld et al., 2011; 54 Nilsson et al., 2005). Civil engineering has been widely used to control floods and prevent 55 channel migration. These practices have profoundly altered the dynamics of rivers by 56 modifying flow and sediment regimes, and consequently, have seriously impacted both 57 aquatic and terrestrial biodiversity (Dudgeon et al., 2006; Poff et al., 2007). Moreover, the radical changes in the structure and composition of riparian vegetation (Naiman and 58 59 Decamps, 1997) have favored the establishment of alien species (Pyšek et al., 2010), leading 60 to an oversimplification of riparian biodiversity (Richardson et al., 2007). 61 In recent years, riparian zone restoration has become a major issue in European countries (Gumiero et al., 2013), and is being carried out both to improve biodiversity conservation and 62 63 water quality. Although the removal of flood prevention infrastructures is often the most 64 effective solution, in many cases, this is far from feasible (González et al., 2017). When 65 channel mobility cannot be restored and when erosion control is of primary concern, soil bioengineering techniques can be viewed as an alternative solution. Soil bioengineering 66

consists of copying naturally functioning riverbank models by using the physical proprieties
of living plants, i.e., root systems provide underground soil reinforcement while foliage and
stems provide surface protection from scouring (Li and Eddleman, 2002). These techniques
have long been used in Europe (Evette et al., 2009) and are still common in restoration
projects nowadays (González et al., 2015).

72 Beyond riverbank stabilization, soil bioengineering techniques are expected to have positive 73 impacts on riparian ecosystems, by increasing the quality of wildlife habitats and water 74 resources (Li and Eddleman, 2002). Through facilitation and amelioration processes, the 75 active introduction of plant species may improve growing conditions for other plant species 76 and increase local heterogeneity (Gurnell, 2014; Gurnell et al., 2012). This, in turn, is likely to benefit riparian biodiversity as a whole. Indeed, soil bioengineering techniques assume that 77 78 the use of early successional species can accelerate desirable successional trajectories by 79 forming a base structure for the desired ecosystem (Clements, 1916; Connell and Slatyer, 80 1977). Thus, soil bioengineering techniques are often viewed as a way to restore riparian 81 ecosystem by favoring the establishment of native species (e.g., Holl and Crone, 2004; 82 McClain et al., 2011) and more generally as a way to improve ecological conditions for riparian biodiversity on degraded riverbanks (e.g., Cavaillé et al., 2013; Li et al., 2006; 83 Sudduth and Meyer, 2006; Wu and Feng, 2006). 84 85 We aimed to study the effect of soil bioengineering techniques on terrestrial and aquatic

habitat quality and taxonomic diversity in the French and Swiss foothills of the Alps and Jura
Mountains. We considered four different riverbank stabilization structures – riprap protection,
mixed protection, vegetated crib wall and willow fascines – of different ages (i.e. occurring
between 3 and 9 years prior to the study), plus natural riparian willow stands as a reference.
Beyond the direct influence of riverbank protection techniques on biodiversity, we were
interested in understanding whether riverbank protection techniques and bank stabilization

92 age induced changes in riparian habitat quality and whether related changes influence multi-93 taxonomic diversity. Indeed, for all types of stabilization structures, vegetation cover is 94 expected to increase over time (Bariteau et al., 2013), which may increase the magnitude of 95 physical interactions with flows and sediments (Corenblit et al., 2007), thus enhancing habitat 96 heterogeneity (Naiman et al., 2005). As a rule of thumb, an increase in habitat quality over 97 time is often observed in restoration projects (e.g., Hasselquist et al., 2015; Lennox et al., 2011). To investigate the added value of soil bioengineering techniques for riparian 98 99 biodiversity, it is thus critical to, first, better define sound techniques that reconcile erosion 100 control and ecological restoration (Rey et al., 2019), second, move beyond categories of 101 individual riverbank stabilization structure by accounting for the effects of bank stabilization 102 age. 103 Specifically, we tested how selected terrestrial and aquatic habitat variables varied among 104 riverbank stabilization structures and bank stabilization age and how richness and abundance 105 of herbaceous plant, ground-beetle and benthic macro-invertebrate species, as well as multi-106 richness and multi-abundance, responded to changes in habitat quality and to broad-scale 107 environmental variables (i.e., climate, hydrology and land-cover) that are known to greatly 108 influence riparian biodiversity (e.g., Collier and Clements, 2011; Feld and Hering, 2007; Kail 109 and Wolter, 2013). The decision was made to focus on richness and abundance patterns, 110 because both metrics may highlight different aspects of ecosystem functioning, e.g., biotic 111 interactions for species richness versus biomass production for total abundance (Soliveres et 112 al., 2016). The three taxonomic groups were selected based on (i) their complementary 113 response in terms of dispersal ability and (ii) their habitat requirements. Since cross-taxon 114 congruence is of low consistency (Burrascano et al., 2018), the multi-taxonomic approach 115 represents an opportunity to enhance our understanding of the human impacts on biodiversity, 116 allowing us to better orient conservation and restoration strategies. Based on this approach,

we explored the three following hypotheses: (i) soil bioengineering techniques through the
active introduction of native tree species induce an increase in riparian habitat quality, which
is expected to furthermore increase over time; (ii) related changes in riparian habitat quality
induce an increase in the richness and abundance of terrestrial and aquatic species groups; (iii)
the added-value of soil bioengineering techniques for biodiversity is mediated by broad-scales
environmental factors (i.e., climate, hydrology and land-cover) which may greatly influence
the response of multi-taxonomic diversity to riparian habitat quality.

124

#### 125 **2.** Materials and Methods

#### 126

#### 2.1. Study area and sampling design

127 The study was carried out in the French and Swiss foothills of the Alps and Jura Mountains (Fig. 1), which are characterized by a limestone substratum and a temperate climate. We 128 129 selected a large array of rivers (n = 23), between the Drôme River ( $44^{\circ}43$ 'N;  $4^{\circ}58$ 'E) and the Doubs River (47°21'N; 7°10'E), at elevations ranging from 200 to 700 m asl. The watershed 130 area at the sampling sites ranges from 5 to  $5,700 \text{ km}^2$  and land cover is characterized by 131 132 forested areas (48 %), agricultural areas (39 %), sparsely vegetated areas (8 %) and urban 133 areas (5 %). All of the rivers studied belong to the same vegetation zone, i.e., "lower and mid-134 mountain: collinear and mountain vegetation belts" (Ozenda and Borel, 2000) and to the same 135 major group of stream types in Europe, i.e., mountain streams (Sandin and Verdonschot, 136 2006). 137 In 2011, we sampled a total of 37 riverbank sites: 29 were engineered for erosion control and 8 were young natural riparian willow stands (Table A.1.). Among the engineered riverbank 138 139 sites, four different stabilization structures were investigated: riprap protection (n = 8), mixed

140 protection, i.e., riprap at lower part combined with soil bioengineering at the upper part of the

bank (n = 7), vegetated crib wall (n= 6) and willow fascines (n= 8). All streambank protection
occurred between 3 and 9 years prior to the study.

143 **2.2. Bio** 

#### 2.2. Biodiversity assessment

Along each riverbank site, two terrestrial species groups – vascular plants and ground-beetles
- and one aquatic species group - benthic invertebrates - were sampled.

146 Plant species were surveyed following the Point Contact Method from May to July 2011.

147 Plant species diversity and frequency were estimated using a 2m-long stick. Measurements

148 were taken every meter along three 25 m transects located parallel to the riverbank (transect 1

149 near the water line, transect 2 in the middle of the bank and transect 3 at the top of the

150 riverbank). All species were identified by the authors using floras.

151 Ground-beetles were collected on a subset of 34 riverbank sites in June 2011. At each of the

sites, two pitfall traps were buried into the ground, 20 m away from each other. The traps had

a diameter of 7 cm and were filled with a mixture of 50 % propylene glycol and 50 % water

and detergent to kill and preserve the insects. All the ground-beetles trapped were identified to

the species level by independent experts.

156 A surber sampler was used to sample benthic invertebrates – aquatic insects, shellfish,

157 mollusks and worms (i.e., Platyhelminthes, Annelida and Nematoda) – in September and

158 October 2011. To explore a representative range of habitats at each site, five surber samples

159 (500  $\mu$ m mesh size, sampling area of 1/20 m<sup>2</sup>) were taken in the five most qualitative

submerged habitats (for details, see Cavaillé et al., 2018). The material collected was fixed in

161 70 % ethanol and sorted in laboratory. Benthic invertebrates were identified by independent

162 experts to the lowest practical taxonomic level.

163

#### 2.3. Riparian habitat variables

164 To assess the added value of soil bioengineering for riparian biodiversity, we used three165 terrestrial and three aquatic habitat variables that were expected to highlight differences in

habitat quality among the four different stabilization structures plus natural willow stands and 166 167 between recent and old structures. For the terrestrial part of riverbanks, we considered the 168 richness and density of pioneer tree species (i.e., Salix spp., Populus spp. and Alnus spp.) as 169 well as the density of others tree species recorded along the three transects. The focus was 170 made on pioneer tree species because they are naturally widespread along mountain stream 171 riverbanks, they are considered as river system engineers (Gurnell, 2014) and they support a 172 wide range of animal species (Kennedy and Southwood, 1984; Newsholme, 1992). For the 173 aquatic part of the bank, we considered the proportion of the submerged bank covered by the 174 two dominant substrates, i.e., slabs (mean cover = 33.6%) and pebbles (mean cover = 41.8%) 175 which respectively represent low and high potential habitats for macroinvertebrate species, 176 along with an index of substrate quality, combining the proportion of each aquaticmicrohabitat with a "habitability" note, ranking substrates in relation to their ability to host 177 178 organisms. This was done by considering the following 12 substrate types: slab (Hab = 0), 179 algae (Hab = 1), sand/silt (Hab = 2), mud (Hab = 3), helophyte (Hab = 4), gravel (Hab = 5), 180 block (Hab = 6), pebble (Hab = 7), root (Hab = 8), litter (Hab = 9), hydrophyte (Hab = 10) 181 and bryophyte (Hab = 11). Aquatic-microhabitats were considered because they are known to 182 greatly influence benthic invertebrates (Cogerino et al., 1995; Verdonschot et al., 2016) but 183 are rarely considered in restoration projects using soil bioengineering techniques, which 184 mainly focus on the upper part of the riverbank. 185 Based on these six habitat quality variables, we built a synthetic index of riparian habitat 186 quality using the first axis of a principal component analysis (PCA) that reflects the 187 multivariate similarity of riverbank sites (Fig. A.2.). Decision was made to focus on the first 188 PCA axis only because it best represented variations among stabilization structures (i.e., PCA

second axis best represented variations within stabilization structures).

190 **2.4.** Broad-scale environmental variables

191 To validate our sampling design, we used spatial, topographical, climatic, hydrological and 192 land cover variables (Table 1 and Fig. A.3.), in addition to the above-mentioned riparian 193 habitat variables. Spatial – i.e., latitude, longitude – and topographical variables – i.e., 194 altitude, slope gradient and sunlight exposure - were measured directly on each riverbank site 195 using GPS, compass and inclinometer and were used to control for possible bias in sites 196 distribution throughout the study area. Climatic, hydrological and land cover variables were 197 used in models to account for important parameters that may structure biodiversity patterns 198 but also to consider possible factors underlying the effect of riparian habitat variables. 199 Climatic variables – i.e., mean annual air temperature and total annual precipitation – were 200 derived from the WorldClim climatic model (Hijmans et al., 2005) and adjusted for the effect 201 of altitude following Zimmermann and Kienast (1999). Hydrological variables – i.e., stream 202 width and watershed area – were respectively assessed using a laser rangefinder and Digital 203 Elevation Model analysis, by calculating flow direction grids and then delimiting individual 204 drainage areas for each site along river networks with the GRASS GIS software (GRASS 205 Development Team, 2017). Land cover variables – i.e., forest area proportion and urban area 206 proportion in the surrounding landscape – were measured within a 500-m-radius around each 207 riverbank site with the QGIS Geographic Information System (QGIS Development Team, 2015). 208

#### 209 **2.5.** Statistical analyses

Analyses were performed with the R version 3.3.2 software (R Core Team, 2018).

Independent continuous variables with a skewness >1 were log- or log+1-transformed to
reach an approximately normal distribution, while for proportional data, logit transformation
was applied. As independent factors we considered "riverbank protection techniques", a five
levels factor distinguishing among riprap, mixed, crib wall, fascine and natural riverbank

sites, and "age of the structure", a three levels factor distinguishing among recent structures
(i.e., 3 to 5 years old), old structures (i.e., 6 to 9 years old) and natural riverbanks.

To determine whether terrestrial and aquatic habitat variables, as well as the index of riparian
habitat quality, varied among "techniques" and "age" factors, we used one-way ANOVAs
with Tukey's HSD post hoc tests.

220 To determine whether changes in riparian habitat quality among riverbank stabilization structures and bank stabilization age and in broad-scale environmental variables influence 221 222 biodiversity patterns we used Linear Mixed Models (LMMs) and General Linear Mixed 223 Models (GLMMs). As dependent variables, we used the pooled species richness or abundance 224 of herbaceous plants, ground-beetles and aquatic macro-invertebrates, divided into: aquatic 225 insects, shellfish, mollusks and worms, as well as an index of multi-taxonomic diversity 226 (Allan et al., 2014). This diversity metric was calculated as the average scaled species 227 richness or abundance per taxonomic group (i.e., relative to the maximum observed number 228 of species or individuals from each group across all sites). The multi-diversity metric ranges 229 between 0 and 1, with a value of 1 meaning that a site hosts all species contained in the study 230 area species pool. It has the advantage of equally balancing the different taxa and of being comparable across sites, whatever the sampling effort (Allan et al., 2014). We developed 13 a 231 232 priori biologically plausible candidate linear models, plus the null model, to verify hypothesis 233 statements (Table A.4.) and used mixed-effect models with site proximity as a random effect. 234 Site proximity, i.e., sites that belong to the same main river catchment within a 20-km radius, 235 was used to account for spatial autocorrelation between the closest sites and because the study 236 sites on the same riverbank were not real replicated. We fitted normal distribution LMMs for 237 multi-richness and multi-abundance, Poisson distribution GLMMs for richness of each 238 taxonomic group and Negative Binomial distribution GLMMs for abundance of each 239 taxonomic group. Independence of climatic, hydrological and land cover variables from the

index of riparian habitat quality were tested using linear models. The variance explained by 240 241 the models was estimated with the marginal coefficient of determination for fixed effect 242 parameters alone (Nakagawa and Schielzeth, 2013). In all candidate models, the variance 243 inflation factor was below three, indicating a lack of collinearity issues (Dormann et al., 244 2013). To identify the most parsimonious regression model, we used the Akaike information 245 criterion corrected for small sample sizes (Burnham and Anderson, 2002). Model averaging 246 was used when the AICc weight of the top-ranking model was < 0.95. Average parameter 247 estimates and associated unconditional standard errors were calculated from the subset of top-248 ranking models for which the sum of AICc weights reached > 0.95.

How riparian habitat quality varied among riverbank stabilization structures

249

250 **3. Results** 

3.1.

## 251 252

#### and bank stabilization age

253 One-way ANOVA revealed that the index of riparian habitat quality varied significantly 254 among riverbank protection techniques ( $p \le 0.001$ ) and among age classes (p = 0.043) (Fig. 255 2). Tukey HSD tests showed that the index of riparian habitat quality was significantly more 256 important on crib wall, fascine and natural sites than on riprap and mixed sites. For the age 257 factor, Tukey HSD tests revealed no significant differences. For terrestrial habitat variables, 258 one-way ANOVAs revealed that the richness (p < 0.001) and density (p < 0.001) of pioneer 259 tree species as well as the density of others tree species (p = 0.005) varied significantly among 260 riverbank protection techniques but not among age classes (Fig. A.5.). Tukey HSD tests 261 showed that the richness and density of pioneer tree species was significantly more important 262 on mixed, crib wall, fascine and natural sites than on riprap sites and that richness only was 263 significantly more important on fascine sites than on mixed sites. Also, the density of others 264 tree species was significantly more important on mixed sites than on riprap, fascine and

265natural sites. For aquatic habitat variables, one-way ANOVAs revealed that the slab (p <2660.001) and pebble (p = 0.019) substrate proportions varied significantly among riverbank267protection techniques, while only the slab substrate proportion (p = 0.010) varied significantly268among age classes (Fig. A.5.). Tukey HSD tests showed that the proportion of slab was269significantly more important on riprap, mixed and crib wall sites than on fascine and natural270sites. Also, the proportion of slab was significantly more important on recent and old271structures than on natural sites.

3.2. How diversity patterns responded to riparian habitat quality and broad-scale
environmental variables

Overall, 64 ground-beetle species were recorded on the subset of 34 sites; while 189
herbaceous plant, 182 aquatic insect, 15 shellfish, 28 mollusk and 32 worm taxa were
recorded on all 37 sites, resulting in a dataset of 510 taxa.

Linear models showed that the variation in spatial, topographical, climatic, hydrological and
land cover variables was not related to the index of riparian habitat quality (Table A.6.). This
indicates that environmental factors varied consistently along the five types of riverbank sites
and that, in our study design, environmental factors were independent from the synthetic
index developed. Only riverbank slope gradient decreased significantly with an increase of
index values.

LMM results showed that multi-taxonomic diversity was best predicted by models accounting for the index of riparian habitat quality (Table 2). GLMM results showed that the richness of aquatic insect, mollusk and worm species and the abundance of plant species were also best predicted by models accounting for the index. Models accounting for climatic variables only best predicted the richness of shellfish species; models accounting for hydrological variables only best predicted the richness of ground-beetle species and the abundance of ground-beetle, shellfish and worm species; models accounting for land cover variables only best predicted

the richness of plant species and the abundance of aquatic-insect and mollusk species.

Although some of the models seemed quite robust for explaining biodiversity patterns, model
selection uncertainty still remains (Fig. 3 & Fig. A.7.). We therefore used model averaging,
from the 2 to the 12 best models, to draw inferences about how much the variables influenced
diversity patterns.

295 Multi-richness, aquatic insect and worm richness and plant abundance increased significantly 296 with increasing values for the index of riparian habitat quality (Table 3). For climatic 297 variables, only mollusk richness increased significantly with increasing values for temperature 298 while multi-richness and mollusk richness decreased significantly with increasing values for 299 annual precipitation. For hydrological variables, multi-abundance, aquatic-insect, shellfish 300 and worm abundance decreased significantly with increasing values for stream width, while 301 ground-beetle richness increased significantly. Moreover, shellfish abundance decreased significantly with increasing values for watershed area, while ground-beetle and mollusk 302 303 richness increased significantly. For land cover variables, multi-richness and mollusk richness 304 and abundance decreased significantly with an increasing proportion of urban area in the 305 surrounding landscape, while plant richness and aquatic insect richness and abundance 306 increased significantly with an increasing proportion of forest area in the surrounding 307 landscape.

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#### 4. Discussion

Our results clearly show that multi-taxonomic diversity increased with increasing values of riparian habitat quality on riverbanks. Though the significance of the index developed varied among taxonomic groups, the effect on diversity patterns was consistent. These findings support the idea that (i) the active introduction of native tree species enhances riparian habitat quality over time for biodiversity, and that (ii) soil bioengineering techniques, especially

315 willow fascines and to lower extend vegetated crib wall, can be a good compromise solution 316 to support both erosion control and biodiversity conservation.

#### 317 4.1. Soil bioengineering techniques enhance riparian habitat quality for 318 biodiversity

319 Consistently with our first hypothesis, we found that soil bioengineering techniques enhanced 320 the habitat quality of both terrestrial and aquatic compartments. Thus, our results indicate that willow fascines and vegetated crib wall, i.e. soil bioengineering techniques, allowed for a 321 322 greater richness and density of pioneer tree species but also for a larger cover of high quality 323 aquatic micro-habitats, i.e., better able to host organisms. Reversely, riprap and mixed 324 protection, i.e. civil engineering techniques, were associated with an increase in non-pioneer 325 tree species richness, among which the shrubs species Buddleia davidii, Cornus sanguinea 326 and *Ligustrum vulgare* dominated, and with an oversimplification in aquatic-microhabitats, 327 which was due to an artificially increase in the proportion of slabs on the submerged part of 328 the bank. Globally, our results pointed out the multivariate similarity, on the one hand, 329 between stabilization structures made of willow fascines and natural riparian willow stands 330 and, on the other hand, between riprap structures and mixed protections (see also Fig. A.2.). This highlights the importance of including not only the upper part of the riverbank but also 331 332 the lower submerged part when designing restoration projects. Riverbank protection 333 techniques combining artificial riprap on the lower part of the bank and bioengineering 334 structures on the upper part of the bank appear to be poor restoration solutions for aquatic 335 biodiversity (Cavaillé et al., 2018), even though positive effects on terrestrial plant diversity 336 have been highlighted (Cavaillé et al., 2015). River managers wishing to restore both the 337 terrestrial and the aquatic biodiversity should therefore promote substrate heterogeneity along 338 the lower part of the riverbank and maintain or restore native tree cover along the upper part 339 of the riverbank. Overall, our results confirm that soil bioengineering techniques can increase

the habitat quality of degraded riverbanks (Li and Eddleman, 2002), first, by accelerating the 340 341 colonization and establishment of native species (e.g., Holl and Crone, 2004; McClain et al., 342 2011), second, by increasing the overall quality and diversity of wildlife habitats (e.g., Li et 343 al., 2006; Sudduth and Meyer, 2006). This effect may be related to the active introduction of 344 ecosystem engineer species (Gurnell, 2014), such as Salix viminalis, S. puprurea or S. 345 *triandra* in our study area, which are able to rapidly respond to the physical riparian 346 environment by modifying local ecological conditions (e.g., water temperature, sunshine or 347 flow conditions), by increasing physical interactions with sediment load (Corenblit et al., 348 2009) and thus by favouring greater environmental heterogeneity (Gurnell et al., 2012). 349 As expected, the increase in both terrestrial and aquatic habitat quality was related to an 350 increase in both terrestrial and aquatic species richness and abundance. Indeed, we found that 351 the synthetic index of riparian habitat quality developed had a consistent effect among taxa, 352 i.e., the richness and abundance of all the taxonomic groups increased positively with 353 increasing values of the index and significantly for several species groups and the multi-354 richness. Thus, in accordance with the few available studies (e.g., Cavaillé et al., 2013; Li et 355 al., 2006; Sudduth and Meyer, 2006), our results pointed out a positive relationship between 356 soil bioengineering techniques and riparian biodiversity. This result confirms that the active 357 introduction of pioneer tree species, may facilitate the establishment of terrestrial and aquatic 358 species and thus promote biodiversity along riparian zones (Gurnell, 2014). Moreover, the use 359 of willow species, i.e., native pioneer species characteristic of European riverbanks, in soil 360 bioengineering techniques such as vegetated crib wall and fascines may also promote positive 361 biotic interactions by providing shelter and resources for a wide range of species 362 (Newsholme, 1992). For example, it has been showed that *Populus* spp. and *Salix* spp. 363 enhanced fine sediment retention, which increased seed retention of hydrochorous species and 364 favored greater riparian plant diversity (Corenblit et al., 2016). Additionally, our results

showed that benthic macroinvertebrates were significantly influenced by the index of riparian
habitat quality. This confirms the importance of high quality substrates for aquatic
biodiversity (Cogerino et al., 1995; Verdonschot et al., 2016) and the fact that artificial
substrata resulting from civil engineering techniques can reduce the taxonomic and functional
diversity of benthic invertebrates (Feld and Hering, 2007). Overall, we infer that soil
bioengineering techniques may promote the self-organizing ability of riparian ecosystems,
leading toward the desired target of stable community development over time.

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# 4.2. Broad-scale environmental variables mediate the positive effects that soil bioengineering techniques have on riparian biodiversity

374 Beside the effect of riparian habitat quality on biodiversity, our results showed that diversity 375 patterns were obviously influenced by environmental factors at larger scales, which mediated 376 the added-value of soil bioengineering techniques for biodiversity. For climatic variables, the 377 negative effect that annual precipitation had on multi-richness may be related to an increase in 378 water flow velocity for streams located at higher elevations (altitude/precipitation, r = 0.86). 379 Indeed, water flow velocity may induce temporal variations in the diversity of benthic 380 organisms, but also in that of terrestrial arthropods through occasional flooding (e.g., Lafage 381 et al., 2015). These deleterious effects may be exacerbated in mountainous areas by the 382 influence of stream slope. Moreover, since precipitation and temperature are closely related (r 383 = -0.72), this effect may also mask the influence that colder environments have on stream 384 invertebrate assemblages, i.e., a decrease in diversity with altitude (Jacobsen et al., 1997). For 385 hydrological variables, negative effects on both the richness and abundance of several 386 taxonomic groups were found. Specifically, multi-abundance decreased significantly with 387 increasing values for stream width. Given the sensibility of benthic macro-invertebrate 388 species to aquatic habitat quality (Cogerino et al., 1995; Verdonschot et al., 2016), we inferred 389 that the negative effect of hydrological factors reported herein may be attributed to changes in

390 water physical proprieties (e.g., turbidity, dissolved oxygen) and/or in terrestrial and aquatic-391 microhabitats (e.g., sedimentation, homogenization) with increasing stream width and 392 watershed area. Indeed, in our study area, the largest rivers (e.g., the Rhône, Isère and Arve 393 Rivers) are located in industrialized valleys or in intensive farming areas and are more likely 394 to be subject to channeling. The negative effects of human activities on riparian biodiversity 395 are even more confirmed by the fact that multi-richness decreased with an increase in the 396 proportion of urban areas in the surrounding landscape. This negative effect of human 397 occupation on riparian biodiversity has already been well documented for multiple taxa (Feld 398 and Hering, 2007; Moore and Palmer, 2005; Paul and Meyer, 2001) and our results confirm 399 these previous findings. Specifically, this showed that stabilization structures made of willow 400 fascines or vegetated crib wall cannot solve environmental degradations due to human activity 401 in the vicinity of riparian zones. Reversely, we found that herbaceous plant richness increased 402 concomitantly with an increase in the proportion of forest areas in the surrounding landscape, 403 which is consistent with the positive overstory-understory relationship that had been 404 previously reported in forest ecosystems (e.g., Ingerpuu et al., 2003; Mölder et al., 2008). 405 Also, we found that the richness and abundance of aquatic-insects, among which 406 Ephemeroptera, Plecoptera and Trichoptera represented 44 % of the total richness and 54 % 407 of the total abundance, significantly increased with forest cover in the surrounding landscape. 408 Given the sensitivity of these taxa to water quality, this result may confirm the effectiveness 409 of riparian forest buffer filters in improving water quality (e.g., Lowrance et al., 1997; 410 Osborne and Kovacic, 1993). Overall, our findings highlight the importance of considering 411 multiple-scale environmental factors when analyzing diversity patterns in riparian zones 412 (Collier and Clements, 2011; Feld and Hering, 2007; Kail and Wolter, 2013). This is all the 413 more true for studies encompassing a wide range of ecological conditions as it was the case 414 for our study in the French and Swiss foothill of the Alps and Jura Mountains.

416

#### 5. Conclusions

417 We found that soil bioengineering techniques are a good compromise between erosion control 418 and biodiversity conservation. However, given that the erosion control performance of these 419 techniques have been rarely investigated (e.g., Fernandes and Guiomar, 2016), we call for 420 more research to better understand the optimal balance between erosion and conservation in 421 riparian zones (Stokes et al., 2014). Also, it should be reminded that riverbank stabilization 422 impedes bank erosion, which is a natural geomorphic process that promotes dynamic habitats 423 for riparian biodiversity (Florsheim et al., 2008). River managers should therefore firstly 424 allow bank erosion processes to operate, secondly promote soil bioengineering techniques if 425 important human stakes are threatened by erosion. Beyond the influence of local habitat 426 variables, we found that broad-scale environmental variables shape diversity patterns. This 427 indicates that the full added value of soil bioengineering techniques for biodiversity will only 428 become apparent if more attention is paid to mitigating the negative impact of human-induced 429 environmental changes and if larger scale environmental parameters are taken into account as 430 early as possible in restoration project (Rey et al., 2019). Finally, the strong variation in the 431 response of taxonomic groups to environmental variables that we found suggest that 432 restoration success should not be evaluated based on the interpretation of a single taxonomic 433 group (e.g., Johnson et al., 2006). Integrated measurements of biodiversity seem best suited 434 because they identify conditions that simultaneously maximize the diversity of multiple 435 taxonomic groups.

436

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7° E Bern Switzerland France eneva Lyon Protection techniques: O Riprap ∧ Mixed St. Vegetated crib wall Willow fascines V Natural river bank 45° **Rivers studied Borders** 25 50 100 km

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- **Fig. 1.** Study area and distribution of rivers and sites sampled for riverbank protection
- techniques in the foothills of the Alps and Jura Mountains (France and Switzerland).

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Fig. 2. Variation of the index of riparian habitat quality, i.e., PCA first axis combining three
terrestrial and three aquatic habitat variables, in relation to riverbank protection techniques
factor and the age of the stabilization structure factor. Bold letters indicate significance
differences between factor levels based on Tukey's HSD post hoc test.



Fig. 3. Top-ranked LMMs predicting multi-richness and multi-abundance along different
riverbank stabilization structures of different ages, plus natural riparian willow stands, along a
gradient of increasing riparian habitat quality in the foothills of the Alps and Jura Mountains
(France and Switzerland).



- **Table 1.** Description of environmental variables used to model multi-taxonomic diversity
- 679 patterns along different riverbank stabilization structures of different ages, plus natural
- 680 riparian willow stands, in the foothills of the Alps and Jura Mountains (France and
- 681 Switzerland).

Variables	Description	mean (±SD)	Range						
Spatial and topographical variables									
Lati	Latitude in decimal degrees	45.98 (±0.77)	44.73 - 47.41						
Long	Longitude in decimal degrees	6.08 (±0.71)	4.95 - 7.39						
Alti	Altitude in meters	396 (±116)	161 - 700						
Slope	Slope gradient in percentage	58.4 (±33.8)	12 - 152						
Expo	Sunlight exposure in degrees	137.43 (±97.76)	0 - 315						
Riverbank habitat va	riables								
Habitat_Quality	Index of riparian habitat quality	0.00 (±1.71)	-3.32 - 2.52						
Terrestrial part of	Terrestrial part of the bank:								
Rich_Pioneer	Richness of pioneer tree species	4.14 (±2.25)	0 - 9						
Dens_Pioneer	Density of pioneer tree species	87.43 (±69.67)	0 - 282						
Dens_Others	Density of others tree species	18.95 (±24.10)	0 - 97						
Aquatic part of the	e bank:								
Subst_Quality	Substrate quality index	131.27 (±170.87)	15 - 780						
Prop_Slab	Slab microhabitat proportion (%)	33.62 (±38.80)	0 - 96						
Prop_Pebble	Pebble microhabitat proportion (%)	41.84 (±37.24)	0 - 96						
Broad-scale environr	nental variables								
Climatic variables									
Temp	Mean annual temperature (°C)	10.32 (±0.87)	8.95 - 12.41						
Precip	Sum annual precipitation (mm)	1802 (±87)	1580 - 1961						

Hydrological variables:

Stream_W	Stream width (m)	20.29 (±23.12)	3.00 - 107.00			
Watershed	Watershed area (km <sup>2</sup> )	512 (±1106)	4.4 - 5761.2			
Land cover variables						
Prop_Forest	Forest proportion (%, 500 m radius)	25.64 (±13.63)	0.00 - 54.85			
Prop_Urban	Urban area proportion (%, 500 m radius)	17.83 (±20.87)	0.17 - 62.94			

683	<b>Table 2.</b> Top-ranking models among 14 a priori models predicting terrestrial and aquatic
684	species richness and abundance along different riverbank stabilization structures of different
685	ages, plus natural riparian willow stands, in the foothills of the Alps and Jura Mountains
686	(France and Switzerland), as assessed with the Akaike information criterion corrected for
687	small sample size (AICc). Number of estimated parameters including the intercept and
688	random effect (k), AICc, AICc weight (W) and marginal coefficient of determination for fixed
689	effect ( $R^{2}_{GLMM}$ ) are provided (Independent variables are defined in Table 1).

Dependent variable	Model (fixed-effects)	k	AICc	W	R <sup>2</sup> <sub>GLMM</sub>
Multi-richness	Habitat_Quality + Prop_Urban	5	-61.0	0.387	0.304
Mutli-abundance	Habitat_Quality + Stream_W	5	-63.4	0.485	0.282
Plant richness	Prop_Forest	3	257.5	0.488	0.143
Plant abundance	Habitat_Quality	4	414.9	0.316	0.175
Ground-beetle richness	Watershed	3	167.9	0.604	0.228
Ground-beetle abundance	Watershed	4	226.0	0.150	0.096
Aquatic-insect richness	Habitat_Quality + Prop_Forest	4	298.2	0.94	0.357
Aquatic-insect abundance	Prop_Forest	4	568.5	0.516	0.228
Shellfish richness	Precip	3	134.3	0.202	0.081
Shellfish abundance	Stream_W	4	579.4	0.672	0.422
Mollusk richness	Habitat_Quality + Precip	4	154.9	0.299	0.262
Mollusk abundance	Prop_Urban	4	390.6	0.268	0.167
Worm richness	Habitat_Quality	3	177.6	0.296	0.174
Worm abundance	Stream_W	4	479.5	0.325	0.124

**Table 3.** Average coefficients (AC, mean ±SD) and confidence intervals (95% CI) for each variable predicting terrestrial and aquatic species richness and abundance along different riverbank stabilization structures of different ages, plus natural riparian willow stands, in the foothills of the Alps and Jura Mountains (France and Switzerland). The 95% confidence interval of coefficients in bold excluded 0.

	Habitat_Quality		Temp		Precip		Stream_W	
Dependent variable	AC	95% CI	AC	95% CI	AC	95% CI	AC	95% CI
Multi-richness	0.021 (±0.010)	(0.001; 0.040)	0.036 (±0.019)	(-0.001; 0.073)	-0.041 (±0.018)	(-0.077; -0.006)	NA	NA
Multi-abundance	0.016 (±0.009)	(-0.002; 0.033)	0.025 (±0.020)	(-0.015; 0.064)	NA	NA	-0.047 (±0.015)	(-0.077; -0.017)
Plant richness	0.004 (±0.026)	(-0.046; 0.054)	-0.087 (±0.058)	(-0.200; 0.026)	0.056 (±0.053)	(-0.048; 0.159)	-0.008 (±0.043)	(-0.092; 0.076)
Plant abundance	0.112 (±0.043)	(0.029; 0.196)	0.026 (±0.081)	(-0.133; 0.186)	-0.066 (±0.070)	(-0.203; 0.070)	0.000 (±0.071)	(-0.139; 0.139)
Ground-beetle richness	0.030 (±0.058)	(-0.083; 0.143)	NA	NA	NA	NA	0.297 (±0.101)	(0.099; 0.494)
Ground-beetle abundance	0.121 (±0.105)	(-0.086; 0.327)	0.207 (±0.211)	(-0.206; 0.620)	-0.180 (±0.147)	(-0.468; 0.107)	0.063 (±0.207)	(-0.342; 0.469)
Aquatic-insect richness	0.051 (±0.019)	(0.014; 0.089)	NA	NA	NA	NA	NA	NA
Aquatic-insect	0.010 (10.070)		0.10((+0.102)		0.071 (10.160)	(0.284, 0.242)	0.045 (+0.150)	
abundance	$0.012 (\pm 0.079)$	(-0.143; 0.167)	0.106 (±0.182)	(-0.250; 0.462)	-0.071 (±0.160)	(-0.384; 0.243)	-0.265 (±0.152)	(-0.564; 0.034)
Shellfish richness	0.051 (±0.061)	(-0.067; 0.170)	0.160 (±0.113)	(-0.062; 0.382)	-0.169 (±0.097)	(-0.359; 0.021)	0.031 (±0.098)	(-0.161; 0.223)
Shellfish abundance	0.080 (±0.108)	(-0.131; 0.291)	NA	NA	NA	NA	-0.975 (±0.216)	(-1.397; -0.552)

Mollusk richness	0.105 (±0.065)	(-0.024; 0.233)	0.424 (±0.202)	(0.027; 0.820)	-0.412 (±0.153)	(-0.711; -0.113)	NA	NA
Mollusk abundance	0.196 (±0.189)	(-0.175; 0.567)	0.927 (±0.607)	(-0.262; 2.117)	-0.927 (±0.580)	(-2.063; 0.210)	-0.114 (±0.468)	(-1.031; 0.803)
Worm richness	0.106 (±0.043)	(0.022; 0.191)	0.056 (±0.079)	(-0.098; 0.210)	-0.039 (±0.068)	(-0.172; 0.095)	-0.018 (±0.071)	(-0.158; 0.122)
Worm abundance	0.037 (±0.081)	(-0.122; 0.196)	-0.055 (±0.159)	(-0.367; 0.258)	0.142 (±0.116)	(-0.086; 0.370)	-0.272 (±0.129)	(-0.525; -0.018)

	Watershed		Prop_	Urban	Prop_Forest		
Dependent variable	AC	95% CI	AC	95% CI	AC	95% CI	
Multi-richness	NA	NA	-0.025 (±0.009)	(-0.043; -0.007)	0.015 (±0.022)	(-0.028; 0.058)	
Multi-abundance	-0.007 (±0.009)	(-0.025; 0.010)	-0.012 (±0.009)	(-0.031; 0.007)	NA	NA	
Plant richness	-0.020 (±0.023)	(-0.064; 0.024)	0.006 (±0.028)	(-0.049; 0.061)	0.132 (±0.056)	(0.022; 0.242)	
Plant abundance	0.020 (±0.039)	(-0.057; 0.096)	0.005 (±0.039)	(-0.072; 0.081)	0.029 (±0.095)	(-0.157; 0.215)	
Ground-beetle richness	0.175 (±0.051)	(0.075; 0.275)	NA	NA	NA	NA	
Ground-beetle abundance	0.173 (±0.106)	(-0.035; 0.381)	-0.144 (±0.100)	(-0.339; 0.052)	-0.199 (±0.314)	(-0.815; 0.416)	
Aquatic-insect richness	NA	NA	NA	NA	0.287 (±0.056)	(0.176; 0.397)	
Aquatic-insect abundance	-0.060 (±0.086)	(-0.228; 0.107)	-0.053 (±0.095)	(-0.240; 0.134)	0.548 (±0.200)	(0.156; 0.940)	
Shellfish richness	0.036 (±0.054)	(-0.070; 0.142)	-0.078 (±0.055)	(-0.186; 0.030)	-0.047 (±0.123)	(-0.289; 0.194)	
Shellfish abundance	-0.479 (±0.133)	(-0.739; -0.218)	NA	NA	NA	NA	
Mollusk richness	0.138 (±0.064)	(0.013; 0.263)	-0.191 (±0.089)	(-0.365; -0.016)	NA	NA	
Mollusk abundance	0.175 (±0.224)	(-0.264; 0.613)	-0.505 (±0.253)	(-1.001; -0.010)	0.542 (±0.580)	(-0.594; 1.678)	
Worm richness	0.015 (±0.038)	(-0.060; 0.091)	-0.038 (±0.040)	(-0.117; 0.040)	-0.008 (±0.088)	(-0.182; 0.165)	

Worm abundance

