



26 use and environmental change might influence future patterns of biodiversity and  
27 ecosystem function.

28 **Key words:** anthropogenic land use, aquatic subsidies, diet, human population, stable  
29 isotopes, terrestrial predators

### 30 **Introduction**

31 Decades of research have demonstrated and quantified the tight linkages between aquatic  
32 and terrestrial ecosystems (Fisher and Likens 1973, Bartels et al. 2012). Cross-boundary  
33 fluxes connecting ecosystems, usually termed “subsidies” (Polis et al. 1997b), can be  
34 organisms, energy, or nutrients. Terrestrial-to-aquatic subsidies in the form of litter and  
35 organic matter are essential for aquatic ecosystem function (reviewed by Tank et al. 2010)  
36 and terrestrial prey subsidies also have important effects on riverine food-webs (Polis and  
37 Hurd 1996, Nakano and Murakami 2001, Erős et al. 2012, Gustafsson et al. 2014). More  
38 recently, research has focused on reciprocal subsidies between aquatic and terrestrial  
39 ecosystems (Baxter et al. 2005, Schindler and Smits 2017). Although the amount of  
40 terrestrial-to-aquatic prey subsidies often is greater than the reverse, their overall  
41 contribution to the carbon budget of predators is similar (Bartels et al. 2012). Thus, the  
42 most recent picture to emerge is that of tightly-coupled, roughly reciprocal aquatic-  
43 terrestrial ecosystems, at least at the local scale at which most studies have taken place.  
44 One of the remaining key challenges for understanding the ecology of cross-boundary  
45 fluxes is to determine at which scales and to what extent the structure of the surrounding  
46 terrestrial landscape affects the magnitude and the importance of aquatic-to-terrestrial  
47 subsidies (Marcarelli et al. 2011). At the local scale (100 m buffer), landscape structure

48 has an impact on predator diet by facilitating or preventing subsidies from entering  
49 recipient ecosystems (Greenwood 2014, Muehlbauer et al. 2014). At the landscape  
50 (catchment) scale, ecosystem size (McHugh et al. 2010, Jackson and Sullivan 2017) and  
51 land use (Stenroth et al. 2015, Carlson et al. 2016) have recently more attention. Studies  
52 focusing on the effect of ecosystem size and land use on riparian ecosystem food webs,  
53 however, remain scarce (e.g. Marczak et al., 2007; Schindler and Smits, 2017). Land use,  
54 at local and landscape scales, influences the composition and biomass of both aquatic  
55 insect communities (via water quality, terrestrial subsidies and canopy cover: Dolédec et  
56 al., 2006; Schindler and Smits, 2017; and predator communities: Hendrickx et al., 2007;  
57 Lafage et al., 2015). On the other hand, ecosystem size, by integrating the effects of  
58 spatial heterogeneity, disturbance and productivity, is a strong predictor of food chain  
59 length (Sabo et al. 2010). To gain a better understanding of broader-scale ecological  
60 processes, comparative studies of aquatic-terrestrial ecosystems at the catchment scale  
61 are needed.

62 In this study, we conducted a worldwide meta-analysis of studies that have assessed  
63 aquatic-to-terrestrial subsidies using stable isotopes. We quantified the effects of  
64 ecosystem size, stream morphology and land use on aquatic subsidies to terrestrial  
65 predators. First, we estimated the overall proportion of aquatic subsidies in the diet of  
66 several groups of terrestrial predators, and tested whether the proportion of these prey  
67 was significantly higher than that of terrestrial prey. We hypothesised that the proportion  
68 of aquatic subsidies varied between taxonomic groups of predators, hydrological system  
69 type (hydro-ecoregion) and year. Next, we assessed the relative importance of biotic and  
70 abiotic variables at local- and landscape- scales (100 m buffers and catchments,

71 respectively) for the proportion of aquatic subsidies in the diet of spider and carabid  
72 beetle predators. We hypothesised that landscape-scale variables related to anthropogenic  
73 land use would be of at least equal importance in explaining predators' diets as  
74 commonly-assessed local-scale variables.

## 75 Methods

76 Our meta-analysis focused on the use of aquatic subsidies by terrestrial predators. We  
77 restricted the subsidies to aquatic organisms actively crossing the boundary between  
78 aquatic and terrestrial ecosystems (i.e. macro-invertebrates). All predators consuming  
79 aquatic macro-invertebrates were included. In order to get a more accurate estimation of  
80 the proportion of aquatic subsidies in the diet of predators, we restricted our meta-  
81 analysis to studies using stable isotopes, which integrate the use of prey types over a  
82 longer period of time than do stomach content analyses (Tieszen et al. 1983).

### 83 Data retrieval

84 We searched the Web of Science and Google Scholar for studies focusing on riparian  
85 habitats and using stable isotopes as a tool to infer the contribution of aquatic prey to the  
86 diet of terrestrial predators. The keywords used were “aquatic subsidies” AND “stable  
87 isotope” AND “diet”, which gave 69 results. From these 69 articles we refined the  
88 selection in several steps. First, a selection was made based on words in the title and a  
89 second one on words in the abstract. We then screened the bibliography of the selected  
90 studies to find new references and iterated this search procedure until we did not find any  
91 new documents. This procedure reduced the 69 papers to 47. At last, a selection of  
92 studies was based on the number of sampling sites and replicates in the different studies,  
93 i.e. we kept studies with at least two sampling sites or studies with repeated  
94 measurements in time and studies including sampling of two predator species.

95 As studies using experimental manipulation of subsidies (and using stable isotopes) were  
96 very rare, descriptive studies were also included. Studies on predators’ diet based on

97 stable isotopes include a great variety of techniques used to partition the diet between  
98 aquatic and terrestrial prey (mainly linear mixing models vs Bayesian mixing models),  
99 and great differences in the assumed isotope fractionation between trophic levels. To  
100 overcome this issue we (re)-calculated the percentage of aquatic prey in the diet of  
101 predators using the same Bayesian mixing model and fractionation values. Using the  
102 same fractionation values for all studies was essential as Bayesian mixing models may be  
103 highly sensitive to the value used (Bond and Diamond 2011). Consequently, we rejected  
104 studies in which the mean and standard deviation of  $\delta^{13}\text{C}$  and  $\delta^{15}\text{N}$  for consumers and  
105 prey per sampling site could not be extracted. The final data set consisted of 21 studies  
106 (Table 1). Data were retrieved from tables, supplementary material, figures (using  
107 WebPlotDigitizer) or by contacting the authors.

#### 108 Response variable

109 The proportion of aquatic subsidies in predators' diet was inferred using two-source  
110 Bayesian mixing models. Inputs to the models were means and standard deviations for  
111  $\delta^{13}\text{C}$  and  $\delta^{15}\text{N}$  of aquatic and terrestrial preys with fractionation values recommended  
112 by McCutchan et al. (2003). In some studies,  $\delta^{13}\text{C}$  and  $\delta^{15}\text{N}$  values were only available  
113 for basal sources (algae and terrestrial litter). In these cases, trophic fractionation was  
114 estimated using the per trophic step fractionation multiplied by the estimated number of  
115 trophic transfers between the consumer and basal resources. This number was estimated  
116 as the difference between the consumer  $\delta^{15}\text{N}$  and mean basal resource  $\delta^{15}\text{N}$  divided by  
117 3.4‰ (McHugh et al. 2010, Jackson and Sullivan 2017). When raw data for stable  
118 isotope were available for consumers, we used the `simmr` package (Parnell et al. 2013,  
119 Parnell 2016) to infer the proportion of aquatic vs terrestrial subsidies in diet. When only

120 means and standard errors were available we used a modified version of the JAGS  
121 models used by Parnell et al. (2013) to include standard error of the consumer isotope  
122 values as a prior of the model. Source aggregation (terrestrial vs aquatic) was made *a*  
123 *priori* as the number of sources included in models was variable between studies, which  
124 is problematic for *a posteriori* aggregations if one wants to compare diets (Stock et al.  
125 2018). We chose not to give any prior to the proportion of aquatic preys in diet  
126 (generalist diets) which means that all possible combinations of proportions of aquatic  
127 and terrestrial preys were likely *a priori* (Stock et al. 2018).

## 128 Predictors

129 The catchment draining to each sampling location was delineated using QGIS 2.18.18  
130 (Quantum GIS Development Team 2017) and GRASS (GRASS Development Team  
131 2017) plugin r.watershed from a 30 m resolution digital elevation model (Shuttle Radar  
132 Topography Mission (SRTM) 1 Arc-Second Global, LP DAAC). Predictors were  
133 extracted at local (100 m buffer) and landscape (catchment) scales. At the landscape  
134 scale, the predictors were catchment perimeter-to-area (a function of size, shape, and  
135 fractal irregularity or folding of the edge: Polis et al., 1997a); percentage cover of  
136 agriculture, forests, non-forested natural habitats (bare ground, herbaceous, shrubs), open  
137 waters (lakes and meadows) and urban areas; mean percent tree cover (a measure of  
138 canopy cover); and mean human population. At the local scale, the predictors were river  
139 width; meandering ratio over 1 km upstream; land use class; mean percent tree cover; and  
140 mean human population.

141 Land use data were extracted from GLCNMO v3 (Tateishi et al. 2014). Percent tree cover  
142 was extracted from PTC V2 (Geospatial Information Authority of Japan, Chiba  
143 University and collaborating organizations). Mean human population was extracted from  
144 Gridded Population of the World, Version 4 (Center for International Earth Science  
145 Information Network, 2016). River width and meandering ratio were extracted under GIS  
146 using google maps satellite imagery. To take into account the possible influence of  
147 climate, location and local biodiversity, each sampling site was assigned to a freshwater  
148 ecoregion according (Abell et al. 2008).

#### 149 Statistical analysis

150 We used the proportion of aquatic subsidies in the diet minus 0.5 as an effect-size to test  
151 for differences between proportion of aquatic and terrestrial subsidies in the diet of the  
152 terrestrial predators. Freshwater ecoregion, sampling year and taxonomic group of the  
153 predators were included in the model as fixed factors. We used the metafor package  
154 (Viechtbauer 2010) with restricted maximum-likelihood estimator to test the effect-size.

155 The selection of landscape and local variables best explaining the proportion of aquatic  
156 subsidies in predators' diet was done using partial least square regression (PLS) on mean  
157 % of aquatic subsidies in the diet per sampling site. Given the low number of studies  
158 available for some groups (Table 1), the PLS were only performed for spiders and  
159 carabid beetles. Freshwater ecoregion and sampling year were also included in the model  
160 as moderators. PLS regression extracts orthogonal components (latent variables  
161 maximizing the explained variance in the dependent variables) from a set of variables  
162 (Eriksson et al. 2006) and are particularly useful when dealing with correlated predictors  
163 (Carrascal et al. 2009), which is often the case for land use variables. The number of



164 components to be kept was determined based on  $Q^2$  value with a M-fold cross-validation  
165 approach. Eriksson et al. (2006) recommend a ‘variable importance on the projection’  
166 (VIP) greater than 1 for identifying the most important predictors. Predictors with  
167  $0.8 < VIP < 1$  explain only some variation in the model and predictors with  $0.8 < VIP$  are  
168 considered non-explicative. Weights of the variables (loading values) describe the  
169 direction and strength of the relationship between predictor and dependent variables. The  
170 PLS were performed using mixOmics package for R (Le Cao et al. 2017). As we  
171 expected different scale effects according to taxonomic group, the PLS were performed  
172 separately for each group.

173 Dataset and code are available on the Open Science Framework repository  
174 (DOI 10.17605/OSF.IO/T6EYP).

## 175 **Results**

### 176 Dataset description

177 The final dataset resulted in 21 studies representing 159 sampling sites and 400 diets  
178 (Table 1). This corresponds to almost half of the studies initially selected. Twenty-six  
179 studies could not be used, mainly because they did not report data in a suitable format  
180 and quality for analysis of diet partitioning. Among these 21 studies, two were not used  
181 in the PLS because we could not locate the sampling sites with enough accuracy. Spiders  
182 and carabid beetles were the two most studied groups whose diets were estimated in  
183 51.3% and 41.6% of the studies, respectively. The studies were mainly located in the  
184 northern hemisphere with cold or temperate climates (Fig. 1 and Table 1).

185 Study site locations were strongly biased toward small forested catchments with very low  
186 human population density and urbanization extent and located mainly in the northern  
187 hemisphere (Fig. 2 and 3). Conversely, a few studies were also located in rivers with very  
188 large catchments or/and high human population.

#### 189 Predator's reliance on aquatic subsidies

190 The contribution of aquatic subsidies was significantly higher than 50% (effect size =  
191 0.07, CI 95%: 0.013 – 0.13: fig. 4). Our model accounted for 95.3% of the heterogeneity  
192 in diet ( $R^2=95.3$ ,  $Q= 207.5$ ,  $df = 19$ ,  $p < 0.001$ ) with a significant overall effect of  
193 moderators ( $Q_M = 272.7$ ,  $df = 23$ ,  $p < 0.001$ ). Sampling year and freshwater ecoregion  
194 both had a significant effect ( $Q_M = 76.4$ ,  $df = 4$ ,  $p < 0.001$  for year and  $Q_M = 168.8$ ,  $df =$   
195  $15$ ,  $p < 0.001$  for ecoregion). The predator taxonomic group effect was not significant  
196 ( $Q_M = 7.88$ ,  $df = 4$ ,  $p = 0.096$ ), whereas the test for residual heterogeneity was significant  
197 ( $Q_E = 63.5$ ,  $df = 3$ ,  $p < 0.0001$ ), and most of the unaccounted variance is due to residual  
198 heterogeneity ( $I^2= 95.3\%$ ).

#### 199 Predictors of aquatic subsidies contribution

200 In the PLS regression model for spiders (two components:  $R^2 = 0.394$  and  $R^2 = 0.460$ ),  
201 the mean human population at both local scale and landscape scale as well as the  
202 percentage of agriculture at the landscape scale were the most important variables related  
203 to a high proportion of aquatic prey. In contrast, the percentage of non-forested natural  
204 habitats and open waters were related to low percentage of aquatic prey (fig 5). Despite  
205 high loading value, the percentage of open waters was weakly correlated to the percent of  
206 aquatic prey in the diet.

207 In the PLS regression model for carabid beetles (two components:  $R^2 = 0.112$  and  $R^2 =$   
208  $0.041$ ), percent tree cover, forests, and water bodies at the landscape scale were the most  
209 important variables for low proportion of aquatic prey. The percentage of non-forested  
210 natural habitats, urban areas and agriculture at the landscape scale and the river width of  
211 the local scale were most important variables for high proportion of aquatic prey (fig. 6).

## 212 **Discussion**

213 Our study extends recent findings that demonstrate high levels of aquatic-to-terrestrial  
214 subsidies in riparian ecosystems (Bartels et al. 2012), improving both the resolution of  
215 subsidy quantification, and allowing inferences at broader ecological scales. Our meta-  
216 analysis also provides some of the strongest evidence to date of widespread effects of  
217 anthropogenic land use on the riparian food webs. These effects seem to be prevalent at  
218 the landscape scale, probably the most relevant scale for understanding the role of  
219 aquatic-terrestrial linkages for land management practices, such as proposed land use  
220 conversion or biodiversity conservation (Carpenter and Biggs 2009). Despite the general  
221 pattern of high aquatic subsidies use by terrestrial predators, we also documented  
222 significant inter-annual and geographic variations in these subsidies, largely driven by  
223 hydrologic cycles and ecoregion, respectively.

224 We found the diet of riparian predators to be highly dependent on aquatic subsidies ( $>$   
225  $50\%$ , overall effect size =  $0.07$ ). Since we re-computed diet partitioning from raw data to  
226 reduce mixing-model and discrimination-factor biases (Bond and Diamond 2011), our  
227 estimate is likely the most robust to date. This suggests that, in general, the proportion of  
228 aquatic subsidies in predator diets may be even higher than the  $40\%$  reported in Bartels et

229 al.'s (2012) meta-analysis. We could not find any significant effect of predator taxonomic  
230 group, which might be due to the small number of studies dealing with groups other than  
231 carabid beetles and spiders. Given the wide geographic spread of our analysis and the  
232 pattern of high proportion of aquatic-derived carbon across the study sites, it seems likely  
233 that most predator taxa in riparian systems rely on these subsidies for more than 50% of  
234 their diet.

235 Perhaps unsurprisingly, we also found significant temporal (inter-annual) and spatial  
236 (ecoregion) variation in aquatic-to-terrestrial subsidies across the broad geographic scale  
237 of our study. Inter-annual climate-driven effects on stream hydrology (droughts vs  
238 floods) may have important impacts on aquatic and riparian communities (Power et al.  
239 2008, Lafage et al. 2015b, Lafage and Pétilion 2016), and on aquatic and terrestrial food  
240 webs (Marks et al. 2000, O'Callaghan et al. 2013). Thus, inter-annual variation in  
241 hydrologic conditions act as a filter on functional traits of species and determines e.g.  
242 functional length of the riparian food chains. The significant effect of ecoregion on  
243 aquatic-to-terrestrial subsidies is probably due to region-specific differences in species  
244 communities, driven by both physical and ecological processes (Abell et al. 2008). It has  
245 been suggested that aquatic subsidy composition (especially through changes in species  
246 traits) is a key factor for resource use in the recipient system (Stenroth et al., 2015). Also,  
247 changes in predator communities might result in changes in species richness and  
248 functional diversity affecting the ability of predators to capture aquatic preys (e.g. for  
249 birds: Philpott et al. 2009).

250 Numerous studies have demonstrated the importance of landscape-scale processes on  
251 ecological status (e.g. Allan, 2004) and macro-invertebrate communities (aquatic:

252 Lammert and Allan, 1999; Richards et al., 1996, terrestrial: Hendrickx et al., 2007;  
253 Lafage et al., 2015a). The relative importance of landscape- versus local-scale factors,  
254 however, is still under debate (Sandin and K. Johnson 2004, Stoll et al. 2016). In our  
255 study, the proportion of aquatic subsidies in terrestrial predator diets was almost  
256 exclusively related to landscape scale variables; the only significant local variable being  
257 human population. This was surprising, as many studies have highlighted the role of local  
258 vegetation (Tagwireyi and Sullivan, 2016), land use (Stenroth et al., 2015) and stream  
259 morphology (Iwata 2007, Muehlbauer et al. 2014). Our results could be related to the low  
260 resolution of our vegetation-related local variables, which were extracted from satellite  
261 data within a 100 m buffer. Nevertheless, variables related to stream morphology were  
262 not selected, although habitat geometry has been found to be the best predictor of trophic  
263 flow rate across habitat boundaries (Polis et al. 1997a).

264 At the landscape scale, ecosystem size did not explain the proportion of aquatic-terrestrial  
265 subsidies in predator diets. This may be due to the fact that the importance of ecosystem  
266 size and the direction of its relationship to predator diets can be system-specific, as  
267 conflicting relationships have been reported (Iwata, 2007, Stenroth et al. 2015, Jackson  
268 and Sullivan 2017). In our study, agricultural land use and urbanization, however, did  
269 have strong and consistent effects on terrestrial consumer diet which might be driven by  
270 either direct or indirect effects. First, by decreasing water quality, agriculture and  
271 urbanization usually directly affect the composition and quantity of aquatic subsidies  
272 (Carlson et al. 2016), shifting towards more and smaller species and resulting in better  
273 prey availability for smaller terrestrial predators (Stenroth et al. 2015). Second, land use  
274 changes may affect the amount and quality of terrestrial-to aquatic subsidies thereby

275 indirectly influencing reciprocal aquatic-to-terrestrial subsidies (Nakano et al. 1999, Krell  
276 et al. 2015).

277 Habitat openness had opposite effects on spider and carabid diets so that spiders relied  
278 more on aquatic subsidies in forested catchments whereas carabids did the opposite.

279 Riparian carabid beetles are usually small flattened winged species (O’Callaghan et al.  
280 2013) more likely to capture small preys favored by open habitat (Carlson et al. 2016).

281 Conversely, typical riparian spiders in forested catchment are large web-building spiders  
282 (e.g. *Tetragnatha* sp.) that are able to catch and consume large flying preys favored by  
283 forested habitats. Several studies have highlighted body size–trophic level linkages (e.g.  
284 (Cohen et al. 2003). A positive relationship between prey body-size and *Tetragnatha* use  
285 of aquatic subsidies has been previously demonstrated (Tagwireyi and Sullivan 2015).

286 Finally, both groups’ uses of aquatic subsidies were negatively related to the percentage  
287 of lakes at the landscape scale. Jonsson et al. (2018) recently found black fly larvae  
288 autochthony to be positively related to the lake proportion in river. In our case, it is most  
289 likely a geographical artefact. Sites located in Sweden presented the largest proportion of  
290 lakes and the smallest proportion of aquatic subsidies in predator’s diet.

291 The literature on insect emergence is heavily biased towards small streams (Muehlbauer  
292 et al. 2014, Schindler and Smits 2017). We found the same pattern, plus a geographical  
293 bias, for studies on predators’ diet using stable isotopes. Most of the studies we used were  
294 located in the northern hemisphere, in small-forested catchments with low proportions of  
295 agriculture or urbanization (except for studies specifically dealing with the impact of  
296 these land use related variables). As agriculture represents the main land use type in  
297 many developed catchments (Allan 2004) and urban land use exerts a disproportionately

298 large influence on aquatic systems (Paul and Meyer 2001) we call for the development of  
299 studies on large rivers, and on catchment impacted by agriculture and urbanization.  
300 Studies are also needed on southern hemisphere streams.

301 Our study is the first worldwide meta-analyses to use exclusively stable isotope studies in  
302 order to better integrate the temporal component of terrestrial predator diets. We  
303 demonstrated a high reliance (more than 50%) of terrestrial predators on aquatic subsidies  
304 across broad geographic regions, despite large geographic and inter-annual variations.  
305 We further demonstrated a large effect of anthropogenic land use at the catchment scale  
306 across geographic regions. Linking these two key findings suggests that more attention to  
307 broad-scale landscape patterns is warranted to improve our understanding of how these  
308 cross-boundary energy flows affects biodiversity and ecosystem function of tightly  
309 coupled aquatic-terrestrial systems.

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316

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Table 1. Characteristics of the studies used for the meta-analysis. Climate is extracted from Peel et al. (2007) and ecoregion from Abell et al. (2008)

<b>Reference</b>	<b>Code</b>	<b>Group</b>	<b>Nb diets</b>	<b>Country</b>	<b>Sampling year</b>	<b>Nb sites</b>	<b>Climate</b>	<b>Ecoregion</b>
Akamatsu et al. 2004 – Ecol. Res.	AKA04	Spiders	22	JPN	2001	3	Cold, Without dry season, Hot summer	Honshu - Shikoku - Kyushu
Akamatsu & Toda 2011 – Environ. Pollut.	AKA11	Spiders	28	JPN	2002	5	Cold, Dry winter, Hot summer	Honshu - Shikoku - Kyushu
Alberts et al. 2013 – Sci. Total Environ.	ALB13	Birds	8	USA	2011	11	Cold, Without dry season, Hot summer	Teays - Old Ohio
Mccaffery & Eby 2016 - Freshw. Biol.	CAF16	Spiders	4	USA	2005	4	Cold, Without dry season, Warm summer	Upper Missouri
Mccaffery & Eby 2016 - Freshw. Biol.	CAF16	Mouse	4	USA	2005	4	Cold, Without dry season, Warm summer	Upper Missouri
Collier et al. 2002 – Freshw. Biol.	COL02	Spiders	4	ZAF	1997	2	Temperate, Dry winter, Warm summer	Southern Temperate Highveld
Gergs et al. 2014 – Freshw. Biol.	GER14	Spiders	18	GER	2011	9	Temperate, Without dry season, Warm summer	Central & Western Europe
Jackson et al. 2015 – Freshwater. Sci.	JAC15	Spiders	11	USA	2011	11	Temperate, Dry summer, Warm summer	Sacramento - San Joaquin
Jackson et al. 2016 – Ecol. Evol.	JAC16	Spiders	12	ZAF	2014	6	Temperate, Dry winter, Warm summer	Southern Temperate Highveld
Jackson et al. 2017 - Can. J. Fish. Aquat. Sci.	JAC17	Spiders	36	USA	2011	30	Temperate, Dry summer, Hot summer	Sacramento - San Joaquin

Kato et al. 2004 – Ecol. Res.	KAT04	Spiders	13 JPN	2001	Cold, Without dry season, 2 Warm summer	Sakhalin, Hokkaido, & Sikhote - Alin Coast
Kelly et al. 2015 – PeerJ	KEL15	Spiders	17 PRI	2012	1 Tropical rainforest	Puerto Rico - Virgin Islands
Krell et al. 2015 – Limnologica	KRE15	Spiders	6 DEU	2012	3 Temperate, Without dry season, Warm summer	Central & Western Europe
Leigh et al. 2013 – Int. Waters	LEI13	Spiders	4 AUS	2010	4 Arid steppe, Hot	Arafura - Carpentaria
O’Callaghan et al. 2013 – PloS One	OCA13	Beetles	174 GBR	2012	14 Temperate, Without dry season, Warm summer	Central & Western Europe
Paetzold et al. 2005 – Ecosystems	PAE05	Spiders	4 ITA	2002	1 Temperate, Without dry season, Warm summer	Gulf of Venice Drainages
Paetzold et al. 2005 – Ecosystems	PAE05	Beetles	4 ITA	2002	1 Temperate, Without dry season, Warm summer	Gulf of Venice Drainages
Paetzold et al. 2005 – Ecosystems	PAE05	Ants	1 ITA	2002	1 Temperate, Without dry season, Warm summer	Gulf of Venice Drainages
Recalade et al. 2016 – J. Anim. Ecol.	REC16	Spiders	4 BRA	2012	1 Temperate, Dry winter, Hot summer	Upper Parana
Sanzone et al. 2003 – Oecologia	SAN03	Spiders	5 USA	2003	1 Arid steppe, Hot	Gila
Stenroth et al. 2015 – Freshw. Biol.	STE15	Spiders	19 SWE	2011	10 Cold, Without dry season, Cold summer	Northern Baltic Drainages
Stenroth et al. 2015 – Freshw. Biol.	STE15	Beetles	10 SWE	2011	10 Cold, Without dry season, Cold summer	Northern Baltic Drainages
Terui et al. 2017 – Ecol. Res.	TER17	Spiders	1 JPN	2014	4 Cold, Without dry season, Warm summer	Sakhalin, Hokkaido, & Sikhote - Alin Coast

Terui et al. 2017 – Ecol. Res.	TER17 Beetles	5 JPN	2014	Cold, Without dry season, 4 Warm summer	Sakhalin, Hokkaido, & Sikhote - Alin Coast
Walters et al. 2008 – Ecol. Appl.	WAL08 Spiders	12 USA	2008	Temperate, Without dry 4 season, Hot summer	Appalachian Piedmont
Walters et al. 2008 – Ecol. Appl.	WAL08 Herptile	4 USA	2008	Temperate, Without dry 4 season, Hot summer	Appalachian Piedmont
Yuen et al. 2016 Biotropica	YUE16 Spiders	18 HKG	2013	Temperate, Dry winter, Hot 3 summer	Xi Yiang
Yuen et al. 2016 Biotropica	YUE16 Odonata	6 HKG	2013	Temperate, Dry winter, Hot 3 summer	Xi Yiang
Yuen et al. 2016 Biotropica	YUE16 Hemiptera	10 HKG	2013	Temperate, Dry winter, Hot 3 summer	Xi Yiang

## Figure captions

Figure 1: Map of the selected studies. White symbols are studies that were rejected on data quality grounds (see text for details).

Figure 2: Plot of the percentage cover of each land use class in catchments per study.

Figure 3: Histogram of catchment area and mean human population size in the catchment.

Figure 4: Forest plot showing the overall effect-size (observed proportion of aquatic prey in diet minus 0.5). Squares and bars denote means and 95% confidence intervals of the effect sizes, while the size of the squares reflects the weight of each study. Single studies are coded according to Table 1.

Figure 5: The variable weights of the first component in the PLS models for proportion of aquatic prey in spider diet. Positive weights indicate a positive relationship between the predictor and response variables and vice versa. Variables with white bars are non-significant ( $VIP < 0.7$ ). Variables with grey bars are significant with low explicative power ( $0.8 < VIP < 1$ ). Variables in black are significant and are the most contributing variables ( $VIP > 1$ ).

Figure 6: The variable weights of the first component in the PLS models for proportion of aquatic prey in the carabid beetle diet. Positive weights indicate a positive relationship between the predictor and response variables and vice versa. Variables with white bars are non-significant ( $VIP < 0.7$ ). Variables with grey bars are significant with low explicative power ( $0.8 < VIP < 1$ ). Variables in black are significant and are the most contributing variables ( $VIP > 1$ ).

Figure 1

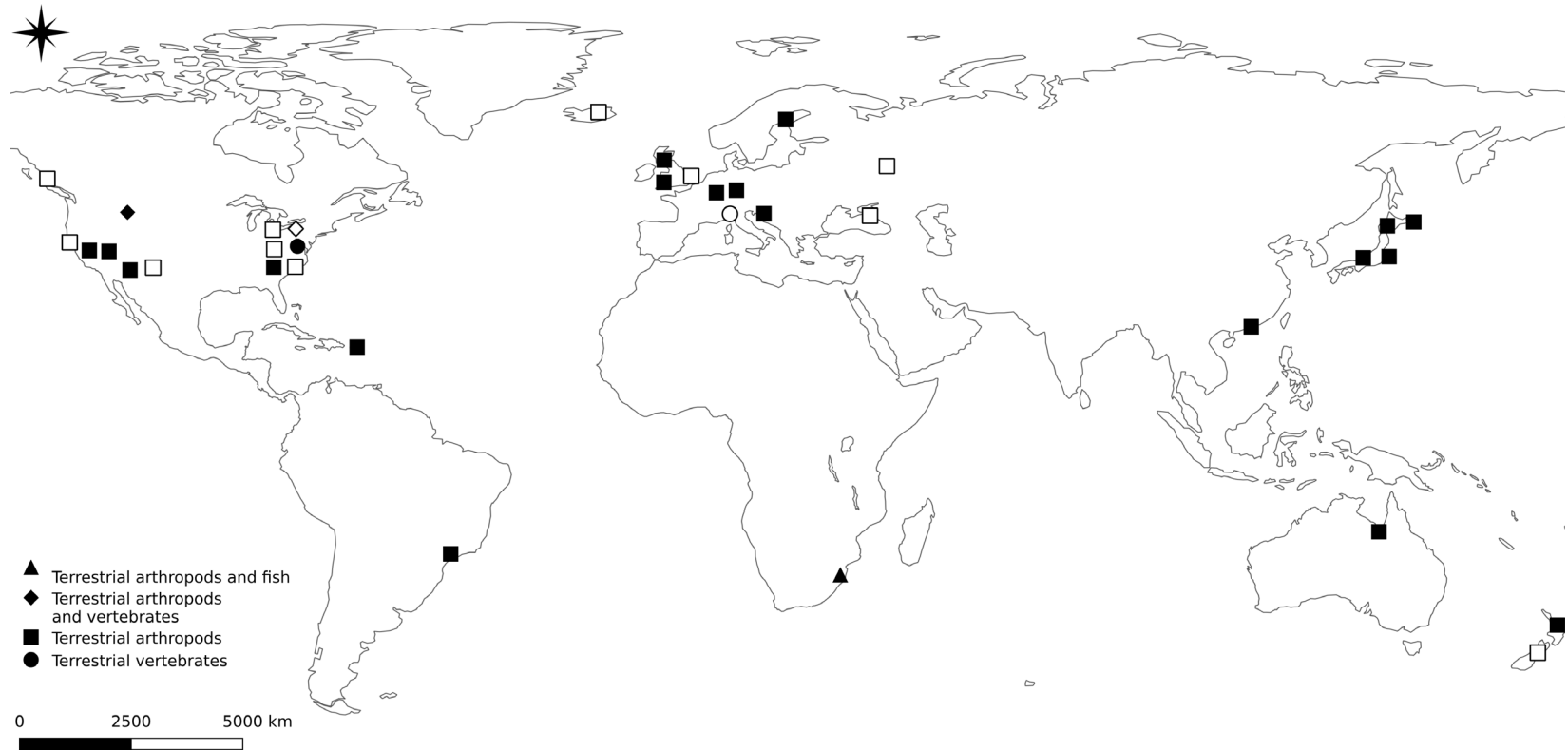




Figure 2

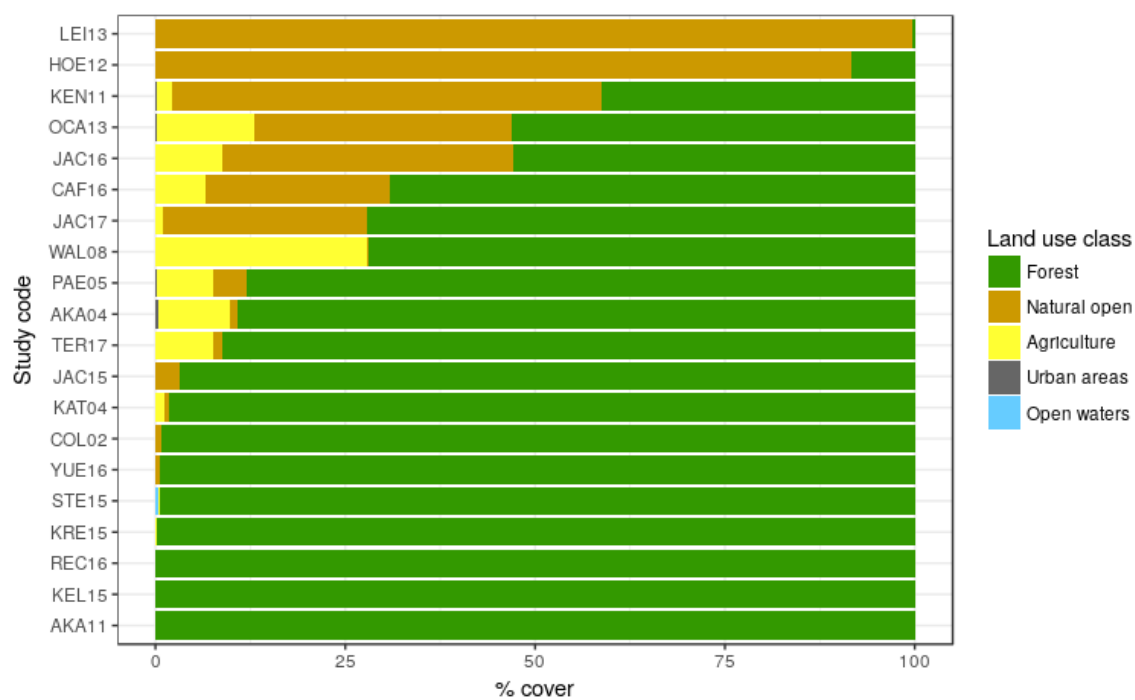


Figure 3

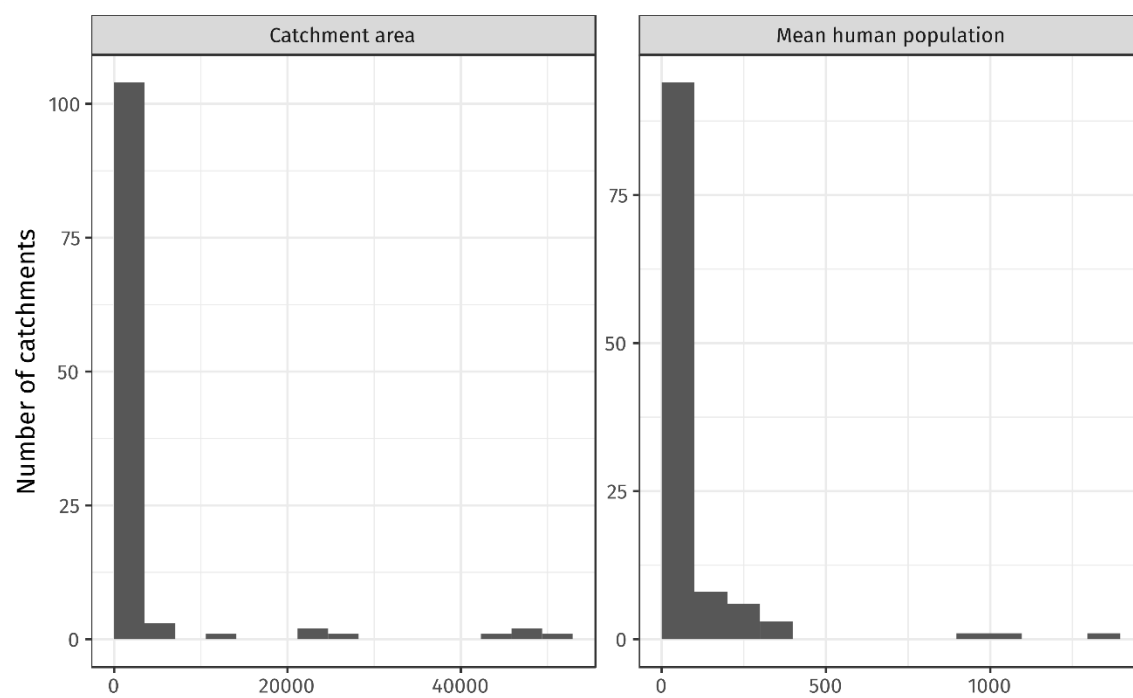


Figure 4

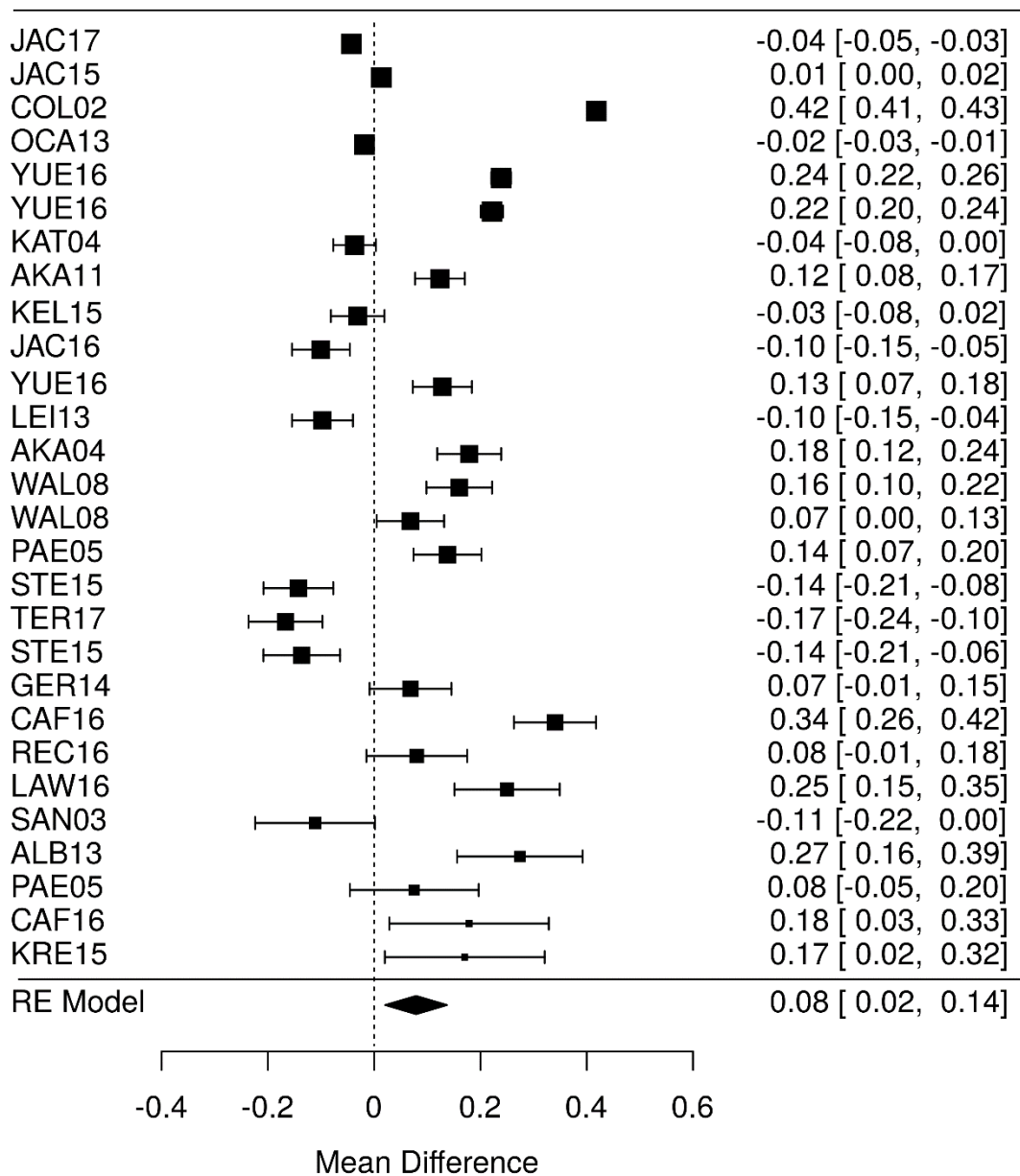


Figure 5

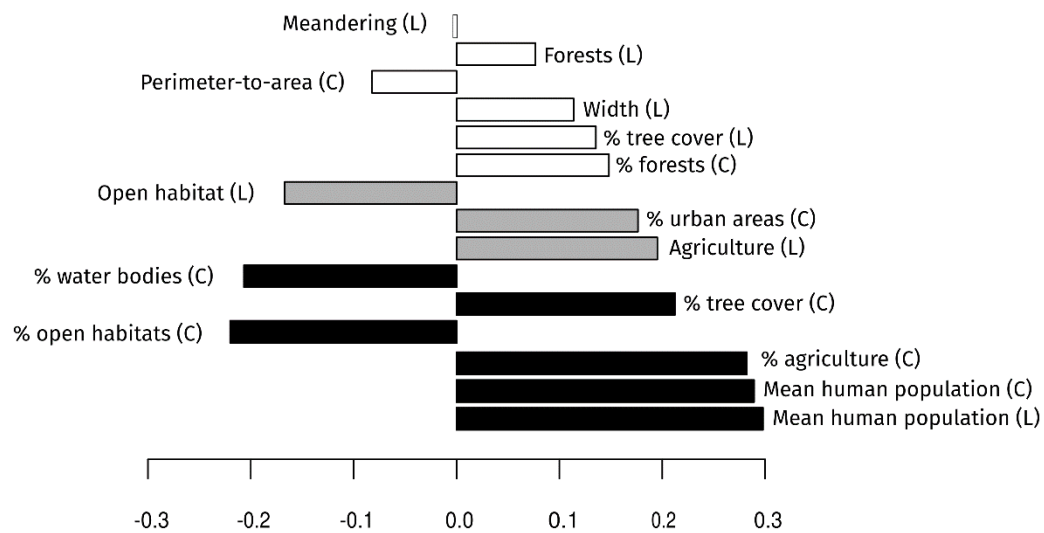


Figure 6

