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For: Ecological Indicators

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8 The effect of urbanization on freshwater macroinvertebrates - Knowledge gaps and future research
9 directions

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21

22 **Abstract**

23

24 Understanding the effects of urbanization on the diversity of freshwater macroinvertebrates is an
25 important topic of biodiversity research and has direct conservation relevance. The absence of
26 evidence-based systematic overviews on this topic motivated us to perform meta-analyses and to
27 synthesize the present state of knowledge. We observed significant heterogeneity among individual
28 case studies, reporting negative, neutral and positive effects. As expected, urbanization had an
29 overall negative effect on the diversity of freshwater macroinvertebrates. These results are based
30 mainly on the study of lotic (stream and river) ecosystems because there are insufficient data
31 available for lentic (pond and lake) ecosystems. Compared to individual case studies, the present
32 review reports an evidence-based synthesis for the first time. We identified knowledge gaps
33 regarding case studies reporting the effects of urbanization on pond and lake ecosystems, case
34 studies examining the phylogenetic and functional facets of biodiversity, as well case studies
35 investigating the effect of urbanization on the beta diversity component of macroinvertebrate
36 communities. The identification of these knowledge gaps allowed us to make recommendations for
37 future research: (1) report results on specific taxonomic groups and not only the entire
38 macroinvertebrate community, (2) study the impacts of urbanization on macroinvertebrate diversity
39 in different habitat types and understudied continents, (3) focus on the functional and phylogenetic
40 facets of diversity and (4) examine the influence of spatial scale on biodiversity (e.g. beta diversity) in
41 urban freshwater ecosystems. Our results also suggested that the analysis of diversity- environment
42 relationships is crucial for developing macroinvertebrate indicators especially in the increasingly
43 urbanized world.

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46 **Keywords**

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48 aquatic invertebrates, biodiversity, effect of urbanization, freshwater ecosystems, systematic review

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51 **1. Introduction**

52

53 Sixty-eight percent of the global population is expected to live in cities by 2050, and the most
54 urbanized regions are North America (with 82% of its population living in urban areas in 2018), Latin
55 America and the Caribbean (81%), and Europe (74%). At the same time, individual cities are also
56 growing in the developing world, resulting in new megacities (UNDESA, 2018). The proliferation of
57 densely-settled areas from the coastal zone to the upstream regions, including mega-cities, means
58 that many rivers are highly threatened over virtually their entire length (Vörösmarty et al., 2010).
59 These freshwater systems have been modified throughout human history to serve humankind,
60 including land cover change, urbanization and industrial purposes. In addition, we have been tireless
61 advocates for expanding the access to the water for many uses and services. Because of the varied
62 economic benefits of the water, it is a challenge to balance between societal and ecological needs
63 (Geist and Hawkins, 2016).

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65 Urbanization alters the physical and chemical environment of rivers, streams (Allan, 2004), lakes and
66 ponds (Heino et al., 2017). The increased impervious cover changes hydrology with frequent and
67 large flow events, while runoffs increase the concentration of sediments, nutrients and chemical
68 pollutants in lotic ecosystems. Such modifications can alter channel morphology and stability,
69 resulting in an altered sediment supply and flow regime. The combination of these changes creates
70 the “urban stream syndrome”, leading to low biotic diversity and altered community structure
71 (Meyer et al., 2005; Paul and Meyer, 2001; Walsh et al., 2005). Similar responses may be found in
72 urban ponds, which are systems that harbor high-levels of biodiversity, despite being small and
73 scattered in the landscape. Whereas previous works indicated biotic homogenization and an overall
74 decline in biological richness of urban ponds and lakes by reason of nutrient enrichment, habitat
75 modification (McGoff et al., 2013) and shoreline development (Brauns et al., 2007), recent findings do
76 not follow the same patterns and provide some contrast with these results in the case of ponds
77 (Hassall and Anderson, 2015; Hill et al., 2016a). Moreover, the effect of the local physical or chemical
78 factors and the degree of connectivity show stronger influence upon lentic systems’ biological
79 diversity than the land use gradients (Hill et al., 2016b; Thornhill et al., 2018). Finally, wetlands might
80 also be severely impacted by urbanization. The knowledge of this effect might guide both local
81 management of wetlands and conservation strategies at the watershed or regional scale to benefit
82 biodiversity of wetlands (Bried et al., 2016; Meyer et al., 2015).

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84 Understanding biodiversity change associated with anthropogenic impacts is crucial to ecologists,
85 and it will be essential for the future success of conservation decisions. Biodiversity, however, can be
86 expressed in multiple ways. Several diversity studies have used taxonomic approaches based on
87 species occurrence, abundance or biomass. Such taxonomic diversity measures treat taxa as being
88 equally distinct from one other and disregard the fact that communities are composed of species
89 with different evolutionary histories and a diverse array of ecological functions (Cardoso et al., 2014).
90 Phylogenetic diversity provides interpretation of the evolutionary relationships among members of a
91 community based on their evolutionary history (Cadotte et al., 2010). Recently, quantitative diversity
92 measures have been developed that use functional traits because they are likely to provide more
93 information about the biodiversity-ecosystem function relationships (Gagic et al., 2015). Additionally,
94 communities in two regions can differ taxonomically but still be similar functionally; thus, functional

95 diversity can be more geographically robust and transferable. Functional traits are measurable
96 characteristics of the organism which define the ecological roles of the species, and functional
97 diversity quantifies the variability or diversity of these functional traits in a community (Schmera et
98 al., 2017). In other words, functional diversity includes those components of biodiversity that
99 influence how an ecosystem operates or functions (Tilman, 1997). Although functional diversity is a
100 promising concept in understanding the functional aspect of biodiversity, functional trait-based
101 approaches are still relatively infrequently applied in comparison to the traditional taxonomic
102 diversity measures (Weigel et al., 2015; Alahuhta et al., in press). This pattern is also the same in the
103 urbanization-related studies. In sum, we can distinguish taxonomic, functional and phylogenetic
104 facets of biodiversity, all of which should be addressed in urban biodiversity studies.

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106 Many studies investigating biodiversity change have been conducted at relatively small spatial scales,
107 generally considered at the local scale (Thompson et al., 2018). However, the spatial patterns of
108 species diversity observed at the local scale may be different from the regional and landscape scales
109 (Heino, 2011). The important effect of spatial scale on biodiversity variation has long been identified
110 (Beever et al., 2006). Taking this into consideration, we can distinguish diversity that occurs within
111 observation unit (α -diversity), among observation units (β -diversity) and total diversity components
112 (γ -diversity) (Whittaker, 1960). Alpha diversity represents the average amount of diversity among
113 samples, indicating the finest scale of sampling. Gamma diversity is the total species diversity of
114 observation units as the set of samples from a single habitat, landscape or region. Finally, beta
115 diversity can be defined as the variation in assemblage composition among sampling units or the
116 extent of change in assemblage composition along gradients (Anderson et al., 2011) and can be
117 calculated as the difference between the gamma and alpha diversity components (Crist and Veech,
118 2006) (Table 1). Despite the important influence of spatial scale on biodiversity (i.e. alpha, beta,
119 gamma components), it has only recently begun to gain broader interest in ecological studies (Crist et
120 al., 2003; Heino, 2011). Thus, it can also be assumed that urbanization influences both within-site
121 (alpha), regional (gamma) and among-sites (beta) diversity components.

122

123 Macroinvertebrates (i.e. invertebrate animals > 0.25 mm in length; Rosenberg & Resh, 1993) play an
124 important role in freshwater ecosystems by feeding on various food resources (e.g. algae, coarse
125 detritus or fine particulate organic matter), by ecosystem engineering (Mermillod-Blondin, 2011), as
126 well as by providing food for higher trophic levels (Covich et al., 1999; Nery and Schmera, 2016).
127 Therefore, macroinvertebrates contribute to several ecosystem services as herbivores, predators or
128 detritivores. Freshwater macroinvertebrate communities are widely used in biomonitoring and
129 bioassessment because they show predictable responses to water quality (e.g. Alvarez-Mieles et al.,
130 2013; Azrina et al., 2006; Gonzalo and Camargo, 2013), hydro-morphological and riparian habitat
131 degradation (e.g. Beavan et al., 2001; Davies et al., 2010; Rios and Bailey, 2006), in terms of the
132 structural and functional parameters of macroinvertebrate communities (Bonada et al., 2006; Li et
133 al., 2019). Many studies have demonstrated that aquatic insects like mayflies (Ephemeroptera),
134 stoneflies (Plecoptera) and caddisflies (Trichoptera) (EPT) are good biological indicators due their
135 high sensitivity to anthropogenic stressors (Hauer and Lamberti, 2007). Some families of beetles
136 (Coleoptera) and true bugs (Hemiptera), especially those using plastrons or bubbles for breathing,
137 are also sensitive to water pollution and habitat degradation, whereas most true flies and midges
138 (Diptera) are opportunists and also colonize polluted water (Tchakonté et al., 2015). In general,
139 narrative reviews and individual case studies suggest that urbanization results in a reduction of
140 richness and abundance of intolerant taxa, and that urban areas are characterized by species-poor
141 assemblages composed of disturbance-tolerant taxa (Allan, 2004; Cuffney et al., 2010; Walsh et al.,
142 2005). All of these studies emphasize the importance of the diversity-environment relationship in

143 developing macroinvertebrate indicators in the urban realm. However, we did not find any
144 systematic overview on whether urbanization influences the diversity of freshwater
145 macroinvertebrates, and which facets (taxonomic, functional or phylogenetic) and components
146 (alpha or beta) are generally impacted.

147

148 The objective of the present study was to assess the effect of urbanization on freshwater
149 macroinvertebrate diversity. To address this issue, we performed a systematic review along with a
150 meta-analysis. The present review focuses on the following questions: (i) Which taxonomic groups
151 have been examined when studying the effect of urbanization on macroinvertebrate diversity? (ii)
152 How is diversity conceptualized (i.e. which diversity facets and components are the foci in a study)
153 and measured in these studies? (iii) Which habitat types are examined? (iv) Does urbanization
154 influence, in general, the diversity of freshwater macroinvertebrates?

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157 **2. Methods**

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159 *2.1 Literature search*

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161 On 16th of November 2017, we performed a literature search in ISI Science Citation Index Expanded
162 database from 1975 to 2016 with the following combination of relevant keywords: ("*diversity*" OR
163 "*richness*") AND ("*macroinvertebrate*" OR "*aquatic invertebrate*") AND ("*urbanization*" OR
164 "*urbanisation*"). This search resulted in 197 papers. Each paper was read carefully to search for
165 outcomes on how urbanization influences the diversity of freshwater macroinvertebrate
166 assemblages. We searched for studies (a piece of scientific work for a particular purpose) reporting
167 contrast between the diversity of macroinvertebrates under natural and urban areas (contrast
168 outcomes), and for studies quantifying the direction and strength of association between
169 urbanization and macroinvertebrate diversity (correlative outcomes). We thus distinguished two
170 outcome types: contrast and correlative ones. We considered an outcome as a contrast outcome
171 when the mean value, the variation (expressed as standard error, standard deviation or confidence
172 interval), as well as the sample size were provided (in a form of text, figure, table or appendix). We
173 considered an outcome as a correlative outcome when both the correlation coefficient and the
174 sample size were given. We recorded taxonomic group (e.g. Decapoda, aquatic insects or
175 macroinvertebrates), habitat (e.g. stream, pond or lake), the facet (taxonomic, functional or
176 phylogenetic) and component (alpha or beta) of diversity from the studies. This search resulted in 27
177 publications, 31 studies and 74 outcomes.

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179 We excluded records when outcomes originated from non-independent observations (i.e. standard
180 error of pairwise beta diversity was quantified based on permutation-based methodology instead of
181 independent observations see Gimenez et al., 2015), or when the variation was obviously
182 inadequately assessed (zero standard error for none-zero mean at sample size 3, see Zhang *et al.*,
183 2012). Furthermore, we deleted records on subgroups if outcomes on entire (or an extended)
184 assemblage was also reported. This means that outcomes for EPT richness were not considered if
185 outcomes on the richness of the entire macroinvertebrate assemblages were also reported. In sum,
186 our search resulted in 27 publications (Electronic Supplementary Material 1), 31 studies and 61
187 outcomes. Using this eligibility dataset, we examined the studied taxonomic groups as well as the
188 methodology used for macroinvertebrate diversity assessment.

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2.2 Data synthesis

Some studies reported multiple outcomes (e.g. both taxa richness and Shannon diversity were given). In order to ensure the independence of outcomes within the same study, we kept only the most frequently-used measure (if both taxa richness and Shannon diversity was provided then we kept only taxa richness). When multiple seasons were studied then we selected only a single one (with the assumed highest diversity). This resulted in 27 papers, 31 studies and 32 outcomes (a single study reported both alpha and beta diversities, which we considered to be independent, see Chao et al., 2012 for more details). Using this final dataset, we examined the influence of urbanization on the diversity of freshwater macroinvertebrates in the meta-analyses.

We calculated Hedges' g (Hedges, 1981) as a measure of effect size for contrast outcomes, while we used Pearson correlation for correlative outcomes. To get an overall result, Pearson correlations were transformed to Hedges' g following (Borenstein et al., 2009). We found significant heterogeneity among studies (see Results section), and thus we fitted random effect models. Our data set did not allow us to test how habitat (only a single outcome reported on ponds while the rest focused on streams) or diversity component (only a single outcome reported on beta diversity while the rest on alpha diversity) influence the effect of urbanization on freshwater macroinvertebrate diversity. We therefore examined only the effect of output type (contrast vs. correlative outcomes) in three steps. First, we applied a random effect model where all outcomes were considered together. In the second step, contrast and correlative outcomes were examined separately in random effect models. Finally, in the third step, we fitted a random effect model containing a moderator (output type, i.e. contrast outcome or correlative outcome) called as mixed effect model (Batáry et al., 2011; Borenstein et al., 2009).

2.3 Assessing publication bias

Studies finding significant effect are more likely to be published than studies finding no effect. This issue is generally known as publication bias. Unfortunately, publication bias might influence the outcome of meta-analyses. To consider publication bias we applied two independent approaches: (1) the Rosenthal method, and (2) the trim and fill methods. The Rosenthal method (Rosenthal, 1979) calculates the number of non-significant studies that need to be added to a summary analysis in order to change the results from significant to non-significant (Batáry et al., 2011). The observed patterns are robust if the number of non-significant studies is greater than $5n+10$, where n is the original number of studies (Rosenthal, 1991). The trim and fill method (Duval and Tweedie, 2000a, 2000b) augments the observed data so that the effect of potentially missing outcomes (provided by the methodology) are incorporated. Then, the method recalculates the summary statistic. If the output agrees with the original conclusion then the inclusion of potentially missing outcomes would not influence our conclusion. All analyses were performed using *R* (R Core Team, 2017) using the package *metafor* (Viechtbauer, 2010).

3. Results

3.1 Methodology of diversity measurement

238 Macroinvertebrates were mostly represented as an entire group, while exclusively a subset of them
239 is only sporadically used in our eligibility dataset (Fig. 1). Regarding habitats, most findings were
240 based on studying the diversity of stream communities (55 of 61, 90.2%). The diversity of pond
241 communities was rarely studied (6 of 61, 9.8%) and that of lake communities were completely
242 ignored (0.0%). The selected outcomes focused exclusively (61 of 61) on the taxonomic facet of
243 macroinvertebrate diversity and, thus, functional and phylogenetic aspects were totally ignored.
244 Most of the outcomes focused on alpha diversity (95.0%, 58 outcomes) and only a relatively small
245 proportion examined beta diversity (3 outcomes). Taxon diversity was the most frequently used
246 measure of alpha diversity (Fig. 2), while Jaccard dissimilarity was the exclusive measure of beta
247 diversity. Finally, we found that most outcomes originate from North America, South America and
248 Europe, while Australia, Asia as well as Africa were less well represented (Fig. 3).

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251 3.2. Effect of urbanization on freshwater macroinvertebrate diversity

252

253 We identified 29 contrast and 3 correlative outcomes in our final data set. When all outcomes were
254 considered together, urbanization had a significant negative effect on macroinvertebrate diversity
255 (*Hedges' g* = -1.643, *s.e.* = 0.429, *z* = -3.33, *P* < 0.001, lower bound of the confidence interval [*ci.lb*] = -
256 2.483, upper bound of the confidence interval [*ci.ub*] = -0.803, Fig. 3). When only contrast outcomes
257 were considered, the effect of urbanization was significantly negative (estimate *Hedges' g* = -1.636,
258 *s.e.* = 0.416, *z* = -3.926, *P* < 0.001, *ci.lb* = -2.453, *ci.ub* = -0.819, Fig. 3), and when only correlative
259 outcomes, the effect was negative but not significant (estimate *Hedges' g* = -1.518, *s.e.* = 2.403, *z* = -
260 0.632, *P* = 0.528, *ci.lb* = -6.229, *ci.ub* = 3.192, Fig. 3). This non-significantly negative effect was caused
261 by two outcomes reporting significantly negative, and one outcome reporting significantly positive
262 effect of urbanization (Fig. 3). Finally, when outcome type was considered as a moderator (mixed
263 effect model), then the intercept of the statistical model (that coincides with contrast outcome type)
264 was significantly negative (*Hedges' g* = -1.661, *s.e.* = 0.461, *z* = -3.599, *P* < 0.001, *ci.lb* = -2.565, *ci.up* =
265 -0.756), and there was no significant difference between outcome types (*Hedges' g* = 0.134, *s.e.* =
266 1.430, *z* = 0.094, *P* = 0.925, *ci.lb* = -2.668, *ci.up* = 2.937 for correlative outcome type), suggesting that
267 there was no difference in the effect of urbanization due to outcome type.

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269

270 3.3 Considering publication bias

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272 The Rosenthal method indicated that 6758 outcomes should be incorporated into our analyses in
273 order to change our significant results to non-significant. This value is much higher than the
274 threshold value (170) suggesting that the conclusion drawn is robust enough. The trim and fill
275 method showed that even when 3 missing outcomes would be added to our data set, the effect of
276 urbanization on macroinvertebrate diversity would still be significantly negative (*Hedges' g* = -2.001,
277 *s.e.* = 0.445, *z* = -4.509, *P* < 0.001, *ci.lb* = -2.877, *ci.ub* = -1.134; Electronic Supplementary Material 2).

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279

280 4. Discussion

281

282 Understanding the effects of urbanization on the diversity of freshwater macroinvertebrates is an
283 important topic of biodiversity research that can serve as the basis for developing
284 macroinvertebrate-based indicators and that has considerable conservation relevance. The absence
285 of evidence-based systematic overview on this topic motivated us to perform meta-analyses and to

286 synthesize the present state of knowledge. We found that urbanization had an overall negative effect
287 on the diversity of freshwater macroinvertebrates. This finding is in compliance with the “urban
288 stream syndrome” described by Meyer et al., (2005) and is in agreement with the majority of the
289 published case studies. Compared to individual case studies, however, the present paper is the first
290 that reports a statistical-based synthesis on this topic.

291
292 The majority of the case studies in our eligibility data set investigated only entire macroinvertebrate
293 communities, some examined both entire communities and specific taxonomic groups (e.g.
294 Ephemeroptera, Plecoptera and Trichoptera), and finally a limited number of case studies focused
295 only on specific taxonomic groups. The consequence of these differences is that we can synthesize
296 information only on entire macroinvertebrate communities, but our synthetic knowledge on how
297 urbanization influences the diversity of individual taxonomic groups is missing. Such information
298 would obviously be important not only for the specialists of particular taxonomic groups, but also for
299 a deeper understanding of the response of entire macroinvertebrate community. Literature evidence
300 suggests that different taxonomic groups (e.g. Ephemeroptera, Plecoptera, Trichoptera, Coleoptera
301 or Hemiptera) respond differently to the effect of urbanization (Compin and Céréghino, 2007;
302 Sánchez-Fernández et al., 2006; Tchakonté et al., 2015) and thus further studies are clearly required.

303
304 Regarding the habitats studied, most outcomes reported case studies on lotic systems and
305 sporadically on ponds, while lakes were completely ignored. These findings suggest that our general
306 conclusion is heavily based on stream studies, and there is a knowledge gap on how urbanization
307 influences macroinvertebrate diversity in pond and lake habitats. We cannot provide a clear
308 explanation for the overrepresentation of stream studies, but a similar bias was found in functional
309 diversity research (Schmera et al., 2017). A possible explanation might be that the comparison of lake
310 communities under clear natural and urban conditions could be challenging (e.g. because of the lack
311 of adequate sampling sites). Despite the conservation importance of urban ponds (Oertli et al.,
312 2005), this habitat type has been mostly ignored by freshwater ecologists (Céréghino et al., 2008)
313 until recently (Heino et al., 2017; Hill et al., 2017). It should also be noted that we did not find any
314 study of wetlands, despite the fact wetlands are ecologically important systems and increasingly
315 threatened by urbanization. Based on our results, well-documented case studies are needed in lake,
316 pond and wetland habitats for the comprehensive interpretation of the effect of urbanization on
317 freshwater macroinvertebrate diversity.

318
319 Regarding the continents, most of the outcomes in our eligibility data set were originated from
320 America (both from North and South America), whereas Africa, Asia and Australia are clearly
321 underrepresented (Fig. 3). This virtual lack of studies might bias our synthesis and should give an
322 incentive to research the effect of urbanization on freshwater macroinvertebrate diversity on the
323 little-studied continents.

324
325 Our systematic review showed that the identified negative effect of urbanization was based
326 exclusively on the taxonomic facet of macroinvertebrate diversity and, thus, functional and
327 phylogenetic aspects were totally ignored. We did not identify any case study which takes functional
328 or phylogenetic diversity into consideration. Obviously, the use of the taxonomic facet alone has
329 considerable limitation for the comprehensive assessment of the response of biodiversity to
330 urbanization (Tanaka and Sato, 2015). This finding highlights a notable deficiency that needs to be
331 addressed urgently in the future, since human impacts are assumed to affect the functional trait
332 composition of macroinvertebrate assemblages (Flynn et al., 2009; Schmera et al., 2017; Vandewalle
333 et al., 2010). Thus, such information might also be essential for conservation practice (Perronne,

334 2014), especially due to the possible mismatch of these diversity facets (Devictor et al., 2010; Heino
335 and Tolonen, 2017).

336

337 We found that the detected negative effect of urbanization on macroinvertebrate diversity was
338 based almost exclusively on local (alpha) component, while among-sites (beta) component has been
339 virtually ignored. It is known, however, that human-impacted ecosystems might suffer beta-diversity
340 decline (Passy and Blanchet, 2007), and thus the investigation of the among-site spatial component
341 of diversity would be an urgent task in urban freshwater ecosystems. The examination of
342 urbanization's influence on beta diversity would be more important in headwater stream systems,
343 where alpha diversity is generally low, although the well-known high beta diversity could generate
344 high gamma diversity (Clarke et al., 2008; Heino et al., 2003). In contrast, in the case of urban ponds,
345 both the alpha and gamma diversities might be relatively high due the already degraded state of the
346 non-urban ponds and the management in the cities which may promote high diversity (Hill et al.,
347 2016a). Moreover, urbanization modifies aquatic habitats with different intensity, which increases
348 the heterogeneity of environmental conditions (Barboza et al., 2015), thereby influencing beta
349 diversity (Specziár et al., 2018). Therefore, the assessment of urbanization's influence on beta
350 diversity is beneficial for determining priority urban conservation areas and potentially degraded
351 sites (Barboza et al., 2015). Our results suggest that there is a need for a further exploration of the
352 urbanization-related mechanisms which might affect the diversity of freshwater macroinvertebrate
353 assemblages.

354

355 Our results clearly indicated some knowledge gaps on how urbanization impacts macroinvertebrate
356 diversity. To deal with these issues, we proposed some recommendations (Table 2). In short, our
357 research field would benefit from the study of the effect of urbanization on the individual taxonomic
358 groups. We identified that the investigation of lentic ecosystems (ponds, lakes) and wetlands are
359 marginal, and that some continents are extremely underrepresented in urban studies. Additionally,
360 our study revealed a serious deficiency on the investigation of functional and phylogenetic diversity
361 facets, as well as the study of among-site (beta) diversity component in urban freshwater
362 ecosystems. All of these findings suggest that information on the effect of urbanization on
363 macroinvertebrate diversity is superficial.

364

365 Our statistical models showed that the overall negative effect of urbanization was associated with a
366 significant heterogeneity (expressed as Q , see also Fig. 4), suggesting that effect sizes (Hedges' g)
367 were more heterogeneous than expected based on sampling error. Therefore, the mixed effect
368 model provided the most adequate synthesis of the examined case studies and heterogeneity should
369 deserve special attention. Interestingly, a single case study indicated a significant positive effect of
370 urbanization on macroinvertebrate diversity (Chadwick et al., 2012). In the study of Chadwick et al.
371 (2012), the examined coastal plain streams as a natural habitat typically have low biodiversity of
372 macroinvertebrates, especially lack of Ephemeroptera, Plecoptera and Trichoptera taxa. Moreover,
373 tidal influence causes lower dissolved oxygen and finer sediment as a natural stressor that masks
374 urbanization effects. Several studies showed that freshwater ecosystems, and especially streams, are
375 dynamic systems with remarkable environmental and biological heterogeneity (Palmer et al., 2010;
376 Vinson and Hawkins, 1998). We found that this heterogeneity can also be observed when the effect
377 of urbanization on macroinvertebrate diversity is estimated.

378

379 A meta-analysis can yield a mathematically accurate synthesis of the case studies included in the
380 analysis. However, if these studies are a biased sample of all relevant studies, then the mean effect
381 computed by the meta-analysis will reflect this bias (Borenstein et al., 2009). We considered

382 publication bias using two independent approaches and found that our conclusions are robust
383 enough. However, our systematic review identified knowledge gaps regarding the studied habitat
384 types (lentic systems), the reported facets (functional and phylogenetic) and components (beta) of
385 diversity.

386

387 To conclude, the present paper reports the first evidence-based synthesis on how urbanization
388 influences the diversity of freshwater macroinvertebrates. We found that urbanization had an overall
389 negative effect on macroinvertebrate diversity. Our systematic review also showed that the
390 knowledge on how urbanization impacts the diversity of freshwater macroinvertebrates is rather
391 deficient, and thus further studies are needed for a more comprehensive understanding of the topic.
392 As a contribution from our study, we made recommendations for the future research topics (Table
393 2).

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624
 625 Table 1: Different components of biodiversity and their interpretation.

Alpha diversity	Local diversity of a sample, a habitat or a site
Beta diversity	Variation in community composition among habitats or the extent of change in assemblage composition along gradients
Gamma diversity	Total species diversity of across single habitat, landscape or region

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628 Table 2: Recommendation for the future research.

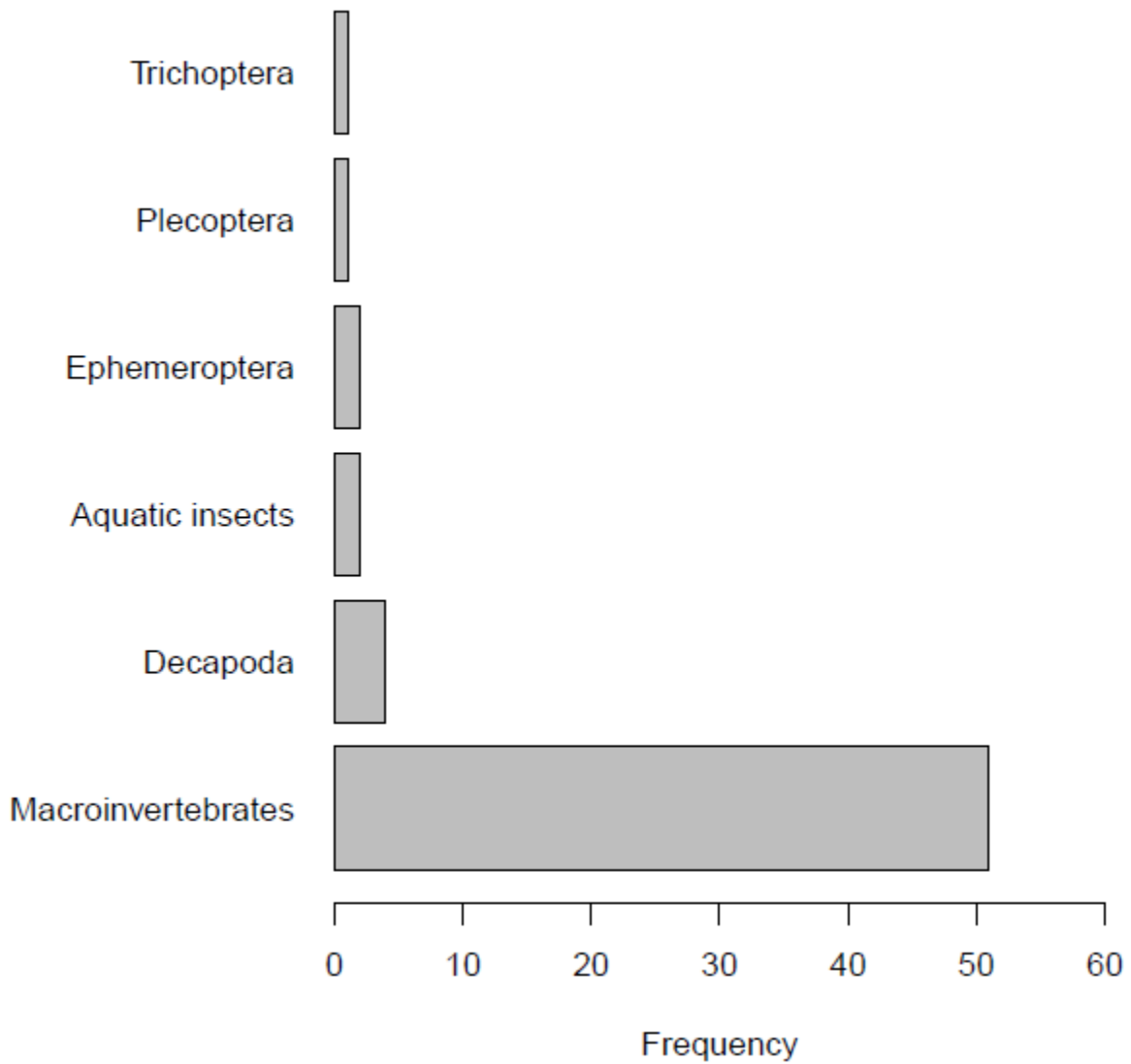
ID	Recommendation
1.	Report results on specific taxonomic group for a deeper understanding of the entire macroinvertebrate community
2.	Study the impacts of urbanization on macroinvertebrate diversity in understudied continents and different habitat types (especially wetlands, ponds and lakes)
3.	Complement taxonomic diversity measures by measures focusing on functional and

phylogenetic facets of the diversity

4 Study the influence of spatial scale on biodiversity, e.g., beta diversity

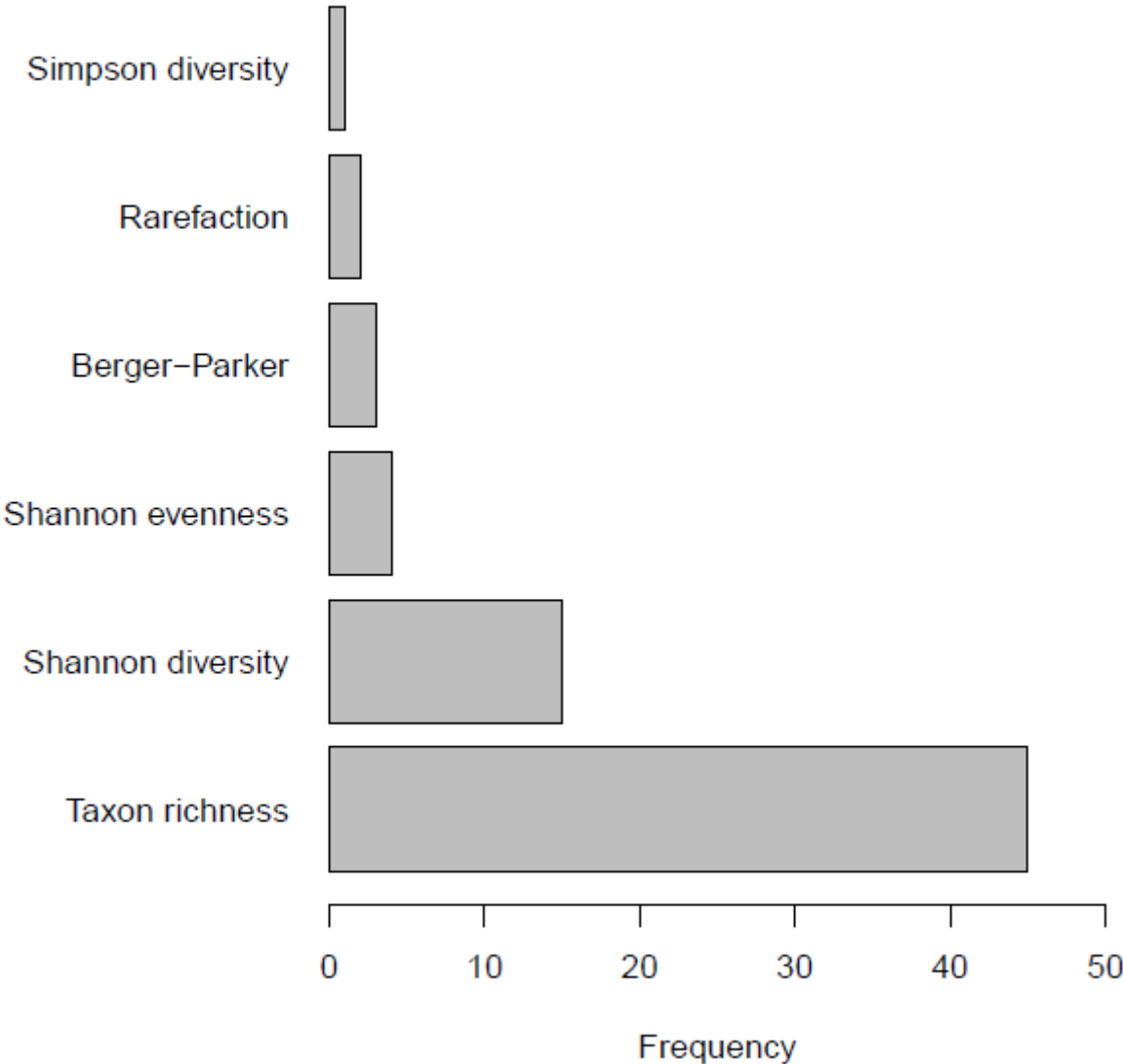
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FIGURE LEGENDS

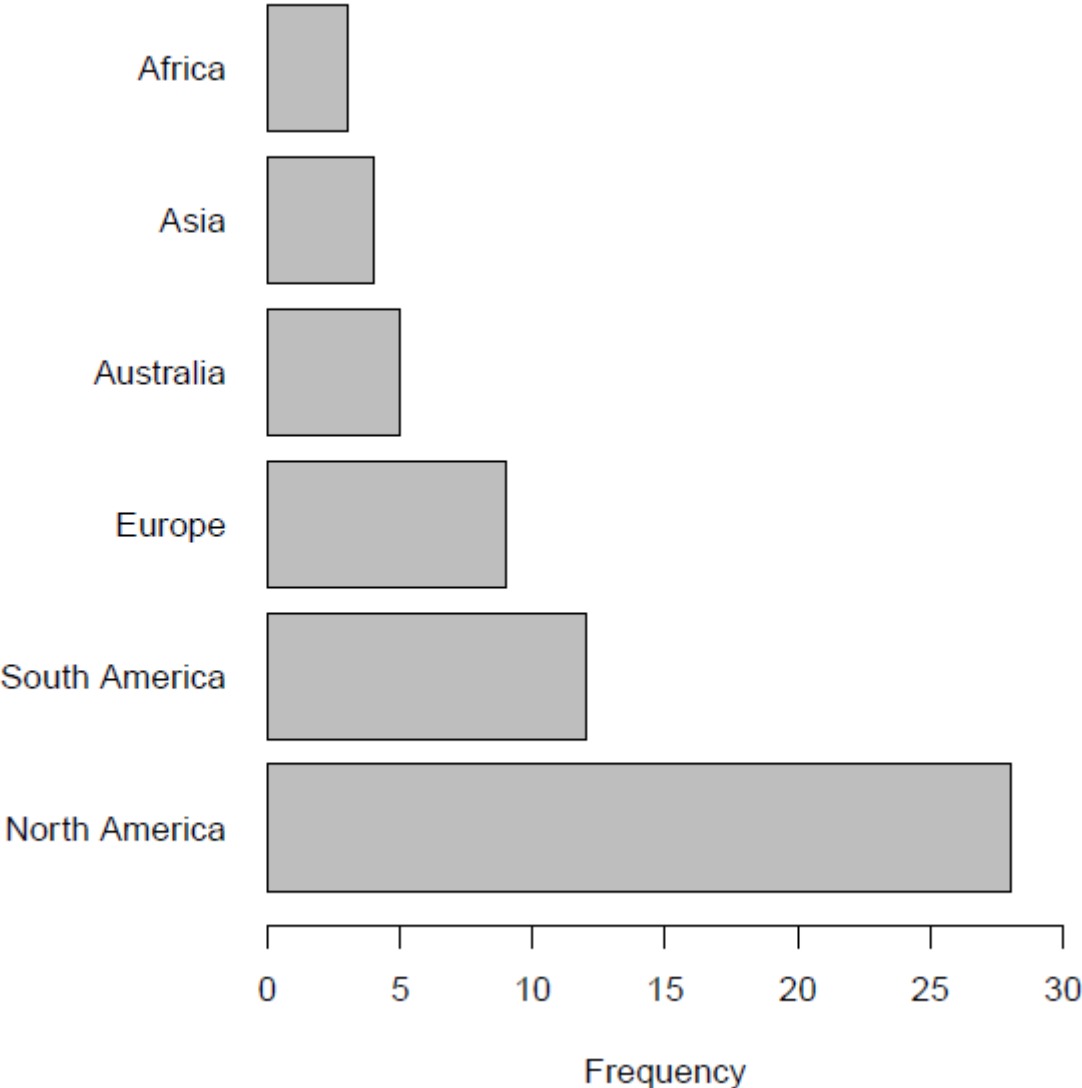


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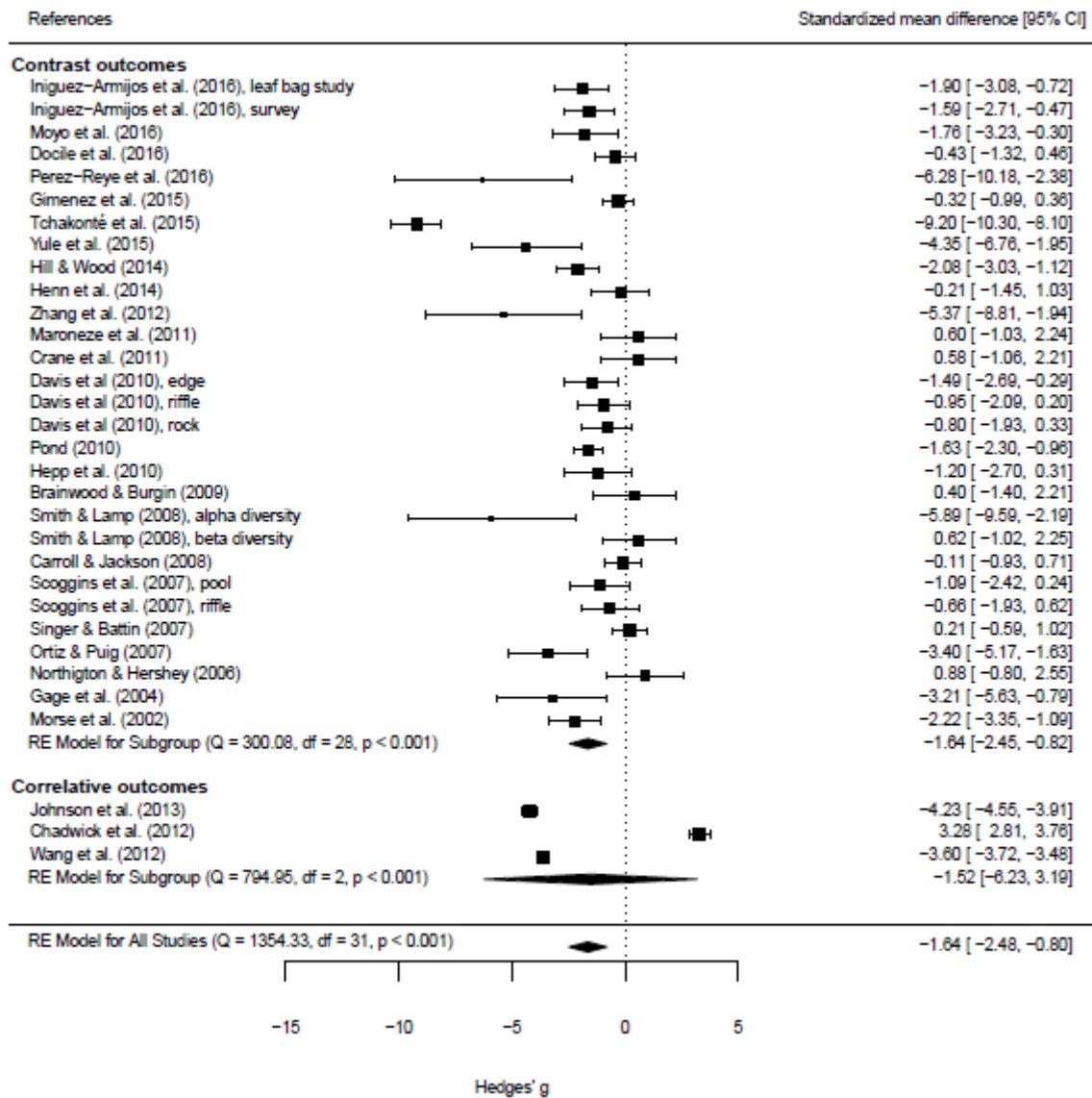
Fig. 1: Frequency distribution of taxonomic groups used to study the effect of urbanization on macroinvertebrate diversity



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639 Fig. 2: Frequency distribution of measures used to study the effect of urbanization on
640 macroinvertebrate diversity
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643 Fig. 3: Frequency distribution of the outcomes in different continents
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Fig. 4: Forest plot of effect sizes (Hedges' g) measuring the effect of urbanization on macroinvertebrate diversity