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

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Abstract

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

Keywords

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

1. Introduction

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

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

Understanding biodiversity change associated with anthropogenic impacts is crucial to ecologists, and it will be essential for the future success of conservation decisions. Biodiversity, however, can be expressed in multiple ways. Several diversity studies have used taxonomic approaches based on species occurrence, abundance or biomass. Such taxonomic diversity measures treat taxa as being equally distinct from one other and disregard the fact that communities are composed of species with different evolutionary histories and a diverse array of ecological functions (Cardoso et al., 2014). Phylogenetic diversity provides interpretation of the evolutionary relationships among members of a community based on their evolutionary history (Cadotte et al., 2010). Recently, quantitative diversity measures have been developed that use functional traits because they are likely to provide more information about the biodiversity-ecosystem function relationships (Gagic et al., 2015). Additionally, communities in two regions can differ taxonomically but still be similar functionally; thus, functional diversity can be more geographically robust and transferable. Functional traits are measurable characteristics of the organism which define the ecological roles of the species, and functional diversity quantifies the variability or diversity of these functional traits in a community (Schmera et al., 2017). In other words, functional diversity includes those components of biodiversity that influence how an ecosystem operates or functions (Tilman, 1997). Although functional diversity is a promising concept in understanding the functional aspect of biodiversity, functional trait-based approaches are still relatively infrequently applied in comparison to the traditional taxonomic diversity measures (Weigel et al., 2015; Alahuhta et al., in press).This pattern is also the same in the urbanization-related studies. In sum, we can distinguish taxonomic, functional and phylogenetic facets of biodiversity, all of which should be addressed in urban biodiversity studies.

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

Macroinvertebrates (i.e. invertebrate animals > 0.25 mm in length; Rosenberg & Resh, 1993) play an important role in freshwater ecosystems by feeding on various food resources (e.g. algae, coarse detritus or fine particulate organic matter), by ecosystem engineering (Mermillod-Blondin, 2011), as well as by providing food for higher trophic levels (Covich et al., 1999; Nery and Schmera, 2016). Therefore, macroinvertebrates contribute to several ecosystem services as herbivores, predators or detritivores. Freshwater macroinvertebrate communities are widely used in biomonitoring and bioassessment because they show predictable responses to water quality (e.g. Alvarez-Mieles et al., 2013; Azrina et al., 2006; Gonzalo and Camargo, 2013), hydro-morphological and riparian habitat degradation (e.g. Beavan et al., 2001; Davies et al., 2010; Rios and Bailey, 2006), in terms of the structural and functional parameters of macroinvertebrate communities (Bonada et al., 2006; Li et al., 2019). Many studies have demonstrated that aquatic insects like mayflies (Ephemereoptera), stoneflies (Plecoptera) and caddisflies (Trichoptera) (EPT) are good biological indicators due their high sensitivity to anthropogenic stressors (Hauer and Lamberti, 2007). Some families of beetles (Coleoptera) and true bugs (Hemiptera), especially those using plastrons or bubbles for breathing, are also sensitive to water pollution and habitat degradation, whereas most true flies and midges (Diptera) are opportunists and also colonize polluted water (Tchakonté et al., 2015). In general, narrative reviews and individual case studies suggest that urbanization results in a reduction of richness and abundance of intolerant taxa, and that urban areas are characterized by species-poor assemblages composed of disturbance-tolerant taxa (Allan, 2004; Cuffney et al., 2010; Walsh et al., 2005). All of these studies emphasize the importance of the diversity-environment relationship in developing macroinvertebrate indicators in the urban realm. However, we did not find any systematic overview on whether urbanization influences the diversity of freshwater macroinvertebrates, and which facets (taxonomic, functional or phylogenetic) and components (alpha or beta) are generally impacted.

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

2. Methods

2.1 Literature search

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

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

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 taxon 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

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

3.2. Effect of urbanization on freshwater macroinvertebrate diversity

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

3.3 Considering publication bias

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

4. Discussion

Understanding the effects of urbanization on the diversity of freshwater macroinvertebrates is an important topic of biodiversity research that can serve as the basis for developing macroinvertebrate-based indicators and that has considerable conservation relevance. The absence of evidence-based systematic overview on this topic motivated us to perform meta-analyses and to synthetize the present state of knowledge. We found that urbanization had an overall negative effect on the diversity of freshwater macroinvertebrates. This finding is in compliance with the "urban stream syndrome" described by Meyer et al., (2005) and is in agreement with the majority of the published case studies. Compared to individual case studies, however, the present paper is the first that reports a statistical-based synthesis on this topic.

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

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

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

Our systematic review showed that the identified negative effect of urbanization was based exclusively on the taxonomic facet of macroinvertebrate diversity and, thus, functional and phylogenetic aspects were totally ignored. We did not identify any case study which takes functional or phylogenetic diversity into consideration. Obviously, the use of the taxonomic facet alone has considerable limitation for the comprehensive assessment of the response of biodiversity to urbanization (Tanaka and Sato, 2015). This finding highlights a notable deficiency that needs to be addressed urgently in the future, since human impacts are assumed to affect the functional trait composition of macroinvertebrate assemblages (Flynn et al., 2009; Schmera et al., 2017; Vandewalle et al., 2010). Thus, such information might also be essential for conservation practice (Perronne, 2014), especially due to the possible mismatch of these diversity facets (Devictor et al., 2010; Heino and Tolonen, 2017).

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

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

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

A meta-analysis can yield a mathematically accurate synthesis of the case studies included in the analysis. However, if these studies are a biased sample of all relevant studies, then the mean effect computed by the meta-analysis will reflect this bias (Borenstein et al., 2009). We considered publication bias using two independent approaches and found that our conclusions are robust enough. However, our systematic review identified knowledge gaps regarding the studied habitat types (lentic systems), the reported facets (functional and phylogenetic) and components (beta) of diversity.

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

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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

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

FIGURE LEGENDS

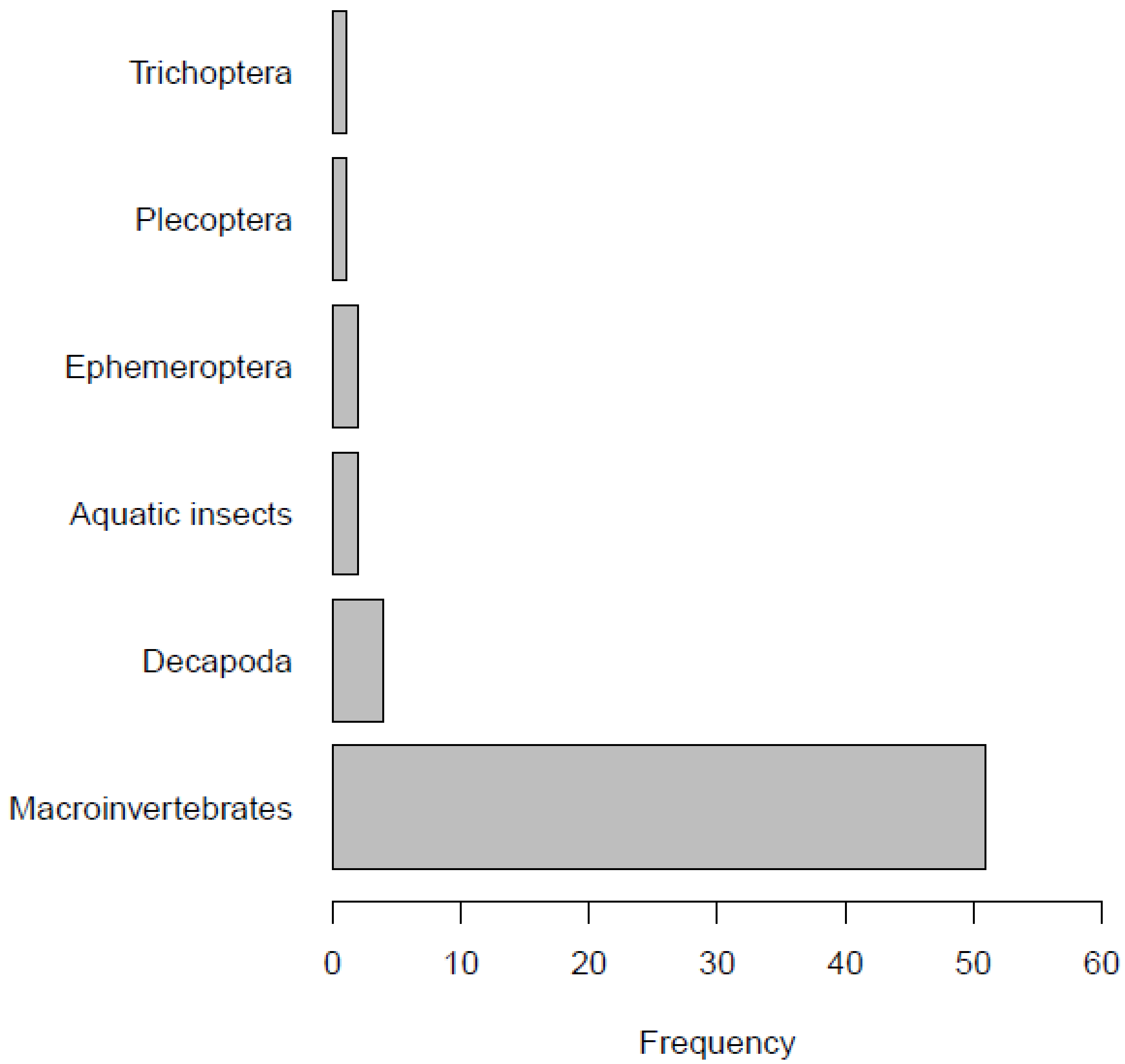

Fig. 1: Frequency distribution of taxonomic groups used to study the effect of urbanization on macroinvertebrate diversity

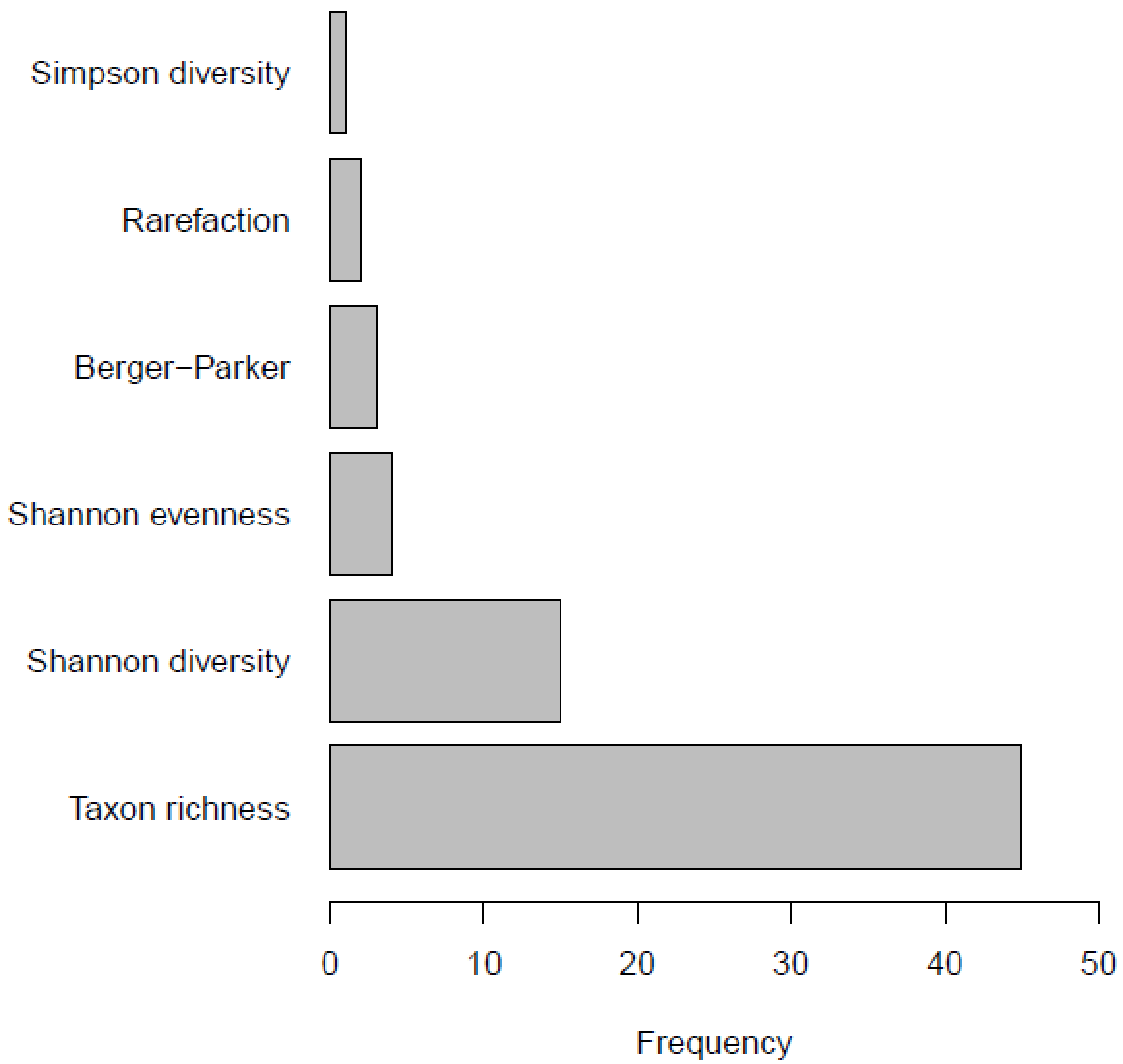

Fig. 2: Frequency distribution of measures used to study the effect of urbanization on macroinvertebrate diversity

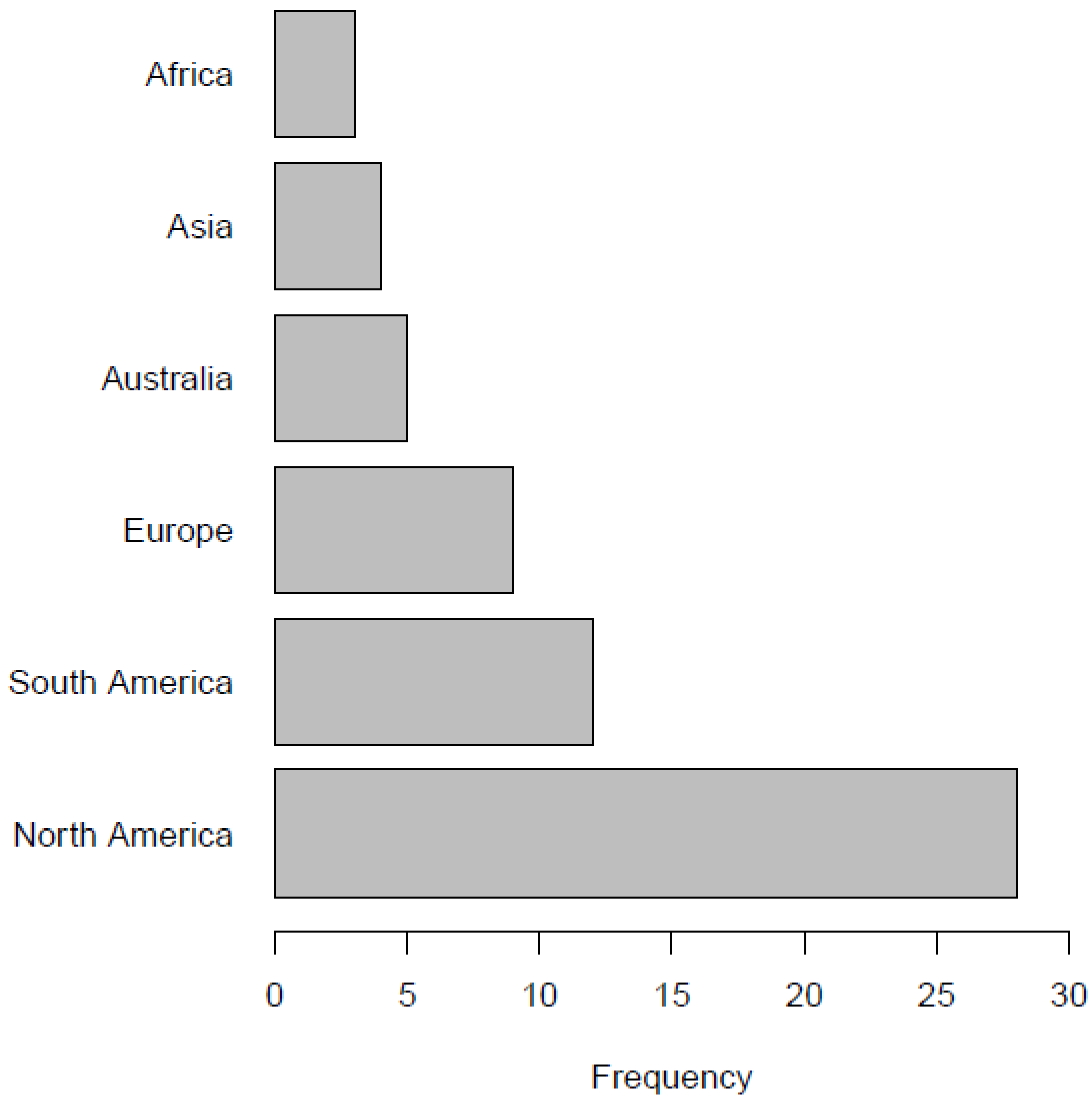

Fig. 3: Frequency distribution of the outcomes in different continents

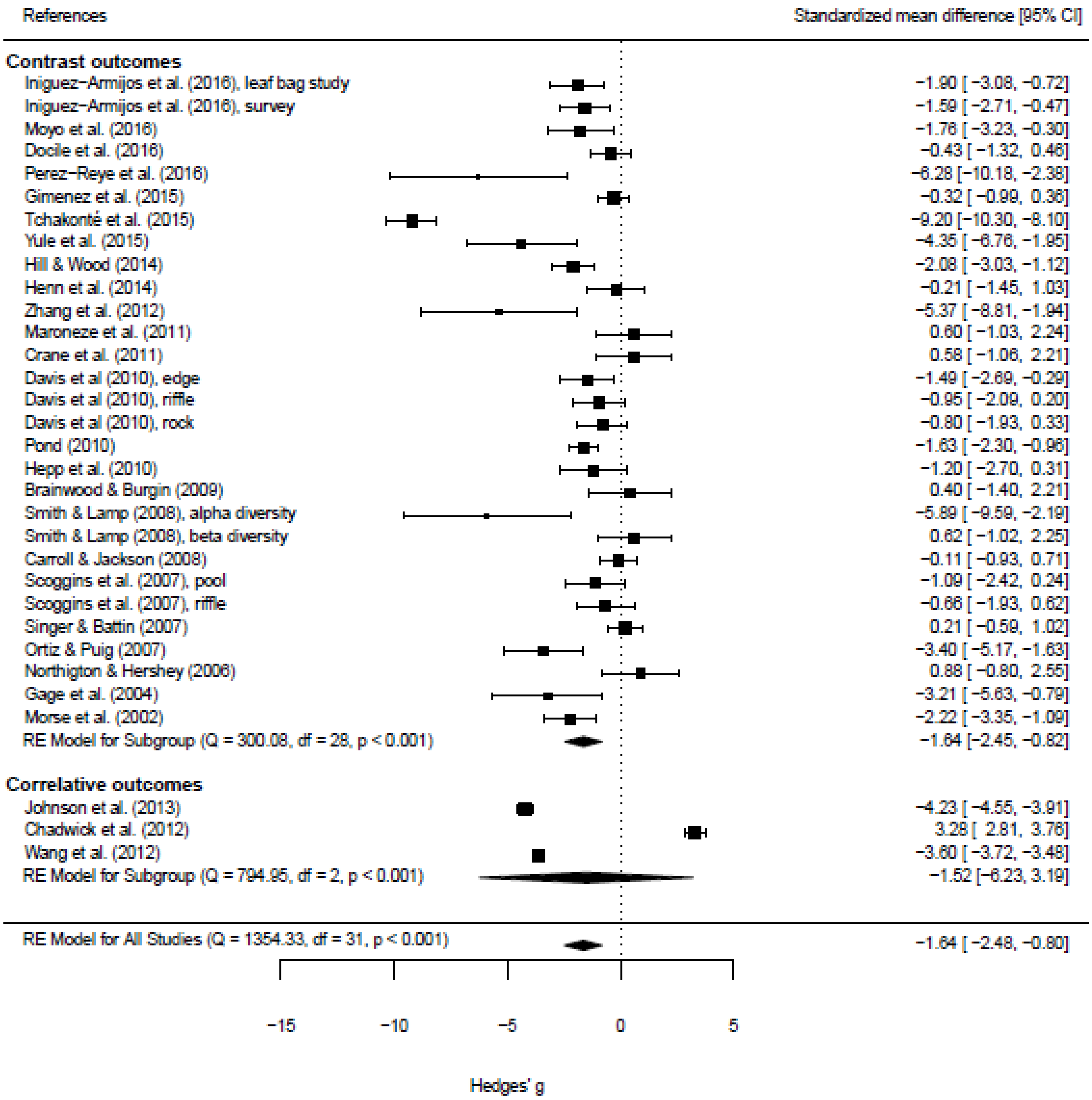

Fig. 4: Forest plot of effect sizes (Hedges' g) measuring the effect of urbanization on macroinvertebrate diversity