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Original article

Invertebrate biodiversity is greater in urban sites with nature-based solutions than control sites: A cross-continental study

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ABSTRACT

Nature-based solutions have gained considerable policy and research attention for enhancing biodiversity and ecosystem services, which face intense pressure from urban development. Assessments of the biodiversity outcomes of nature-based solutions are, however, insufficient. A key unaddressed question is whether such biodiversity effects vary across the urban landscape. We assess whether insect biodiversity (richness, abundance and biomass) effects vary across a gradient of percentage impervious surface in three large cities (Barcelona, São Paulo, Buenos Aires). We focus on Sustainable Urban Drainage Systems (SUDS) which are key in alleviating flood risk and diffuse pollution associated with urban development. We examine habitats that mimic SUDS to investigate well-established habitats, to avoid the risk of underestimating biodiversity benefits arising from lags in the colonisation of recently established nature-based solutions, i.e. taking an observation approach. We do so using random stratification to select survey locations along the urbanisation intensity gradient in each city. Across all cities, NBS sites had greater invertebrate richness, biomass and abundance than control sites. Notably, biodiversity gain is greatest in more urbanised locations where biodiversity is lowest at the control sites. Our work has substantial implications for the cost-effective design and deployment of NBS interventions in urban locations in the Global North and South.

1. Introduction

Urbanisation creates intense pressure on ecological systems, reducing biodiversity and ecosystem service provision (IUCN, 2016) yet urban transformation and urban greenspaces can also enable potential responses to address these impacts (Capon, 2017; Egerer, 2024). Addressing the negative impacts of urbanisation on biodiversity is critical in responding to the 'triple crisis' of pollution, climate change, and loss of ecosystem services (Andersen, 2021; Kowarik et al., 2025).

By definition (sensu EC, 2020) nature-based solutions, such as sustainable urban drainage systems (SUDS), must demonstrably benefit biodiversity and restore ecosystem services. Accepted definitions of SUDS, as nature-based interventions, also incorporate biodiversity improvements (D'Arcy, 1998). However, biodiversity benefits are frequently assumed rather than demonstrated (Lepczyk et al., 2017; Filazzola et al., 2019) and there is surprisingly little research proving urban biodiversity improvements due to the implementation of nature-based solutions (Naumann, 2020). This paucity of evidence for the biodiversity impacts of urban ecosystem restoration previously received insufficient attention (Eggermont et al., 2015).

Interest is growing rapidly in biodiversity offsetting, and the potential role of private sector investment in efforts to restore ecosystems (Seddon et al., 2020; 2021). This is coupled with a growing call for better regulation of nature-based solutions programmes, projects and markets, amid concerns about the possibility of greenwashing, also having the potential to undermine credibility in terms of biodiversity. Efforts are underway to provide ratings to give confidence to investors, buyers, and sellers seeking to ensure that their nature restoration initiatives are effective whilst delivering returns on investment (FFN, 2025). Global standards and impact assessment frameworks have been developed (e.g. IUCN - Cohen-Shacham et al., 2019; EC - Dumitru and Wendling, 2021) but are still relatively new, with assessment standards remaining rather vague as regards ecological assessment; there is a lack of evidence on biodiversity benefits of interventions such as SUDS (Akoumianaki & Pakeman, 2023; Grilo et al., 2025). For both terrestrial and aquatic invertebrates, few investigations have compared urban blue-green interventions with traditional 'grey' infrastructures, or characterised the impacts or mechanisms through which the intensity of

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urbanisation (imperviousness) in which interventions are embedded influences their capacity to enhance biodiversity (Thaweepworadej & Evans, 2022; Nguyen et al., 2024; Bowler et al., 2025). Invertebrates are suffering from rapid decline, are understudied, and yet are central to the functioning of ecosystems (Eisenhauer et al., 2019; Goulson, 2019), further highlighting the importance of these knowledge gaps.

Key drivers of urban biodiversity and particularly for invertebrates may include topography, abiotic factors such as temperature and moisture with associated fluctuations, and anthropogenic drivers e.g. land use and constructed impervious surfaces (Lewthwaite et al., 2024). Other important pressures may include pollution, habitat fragmentation and the loss of vegetation or presence of exotic plant species (Fenoglio et al., 2023). Urbanisation has been found to negatively impact populations of invertebrates such as pollinators, with interacting effects between urban green spaces, agricultural land use and semi-natural areas (Desaegher et al., 2023); thus the overall picture is not straightforward and impact assessment strategies need to be sensitive to diverse factors.

A key concern is that the lack of biodiversity evidence, in cities in particular, may limit wide scale application as well as future investment, as global standards and accreditation schemes increasingly seek to 'prove' biodiversity benefits as a condition for financing (TNFD, 2023; FFN, 2025). This is important because ecosystem services are often most lacking in urban areas (Kowarik et al., 2025), where great scope exists to restore ecosystems (Wild et al., 2024). In part, the lack of data on urban biodiversity impacts may stem from the relatively small number of cases implemented in a sufficiently wide range of settings, despite significant investment in research and innovation (EC, 2020). These evidence gaps provided the focus for our work, which sought to address three key research questions (RQs 1–3):

RQ1. Do nature-based solutions and comparable habitats impart biodiversity benefits in cities?

RQ2. Do such impacts vary across the urban - peri-urban - rural gradient?

RQ3. Can such biodiversity impacts be measured, quantified, and predicted?

2. Methods and materials

This research investigated how nature-based solutions alter urban biodiversity, focusing on invertebrates. An observational approach was taken, examining established habitats to avoid the risk of underestimating biodiversity benefits due to lags in colonisation of recent interventions. Fieldwork was undertaken in three large cities (Barcelona - Spain, São Paulo - Brazil, and Buenos Aires - Argentina) between 2022 and 2023. The research quantified invertebrate biodiversity values for existing habitats relative to randomly selected points, to assess potential biodiversity gains. Cities were selected based on their large size (population and land cover), and where researchers had access to expert staff including local authorities with capacity to support the study.

2.1. Site selection

To assess relative benefits in urban, peri-urban and rural landscapes, the research sought to establish how impacts might vary with position along urbanisation gradients (imperviousness). Impervious Surface Area (ISA) has provided a useful descriptor of urban settlement areas and is closely related to urbanisation intensity (Weng, 2012; Xu and Fina, 2023). It is one of the most widely used metrics for assessing urbanisation intensity (Moll et al. 2019). Previous research has established that biodiversity varies across urban-rural gradients (Korányi et al., 2022; Svenningsen et al., 2022; 2024) but additional research is required to address both urbanisation intensity and potential impacts of nature-based interventions. Random stratification was undertaken to identify sampling grid cells along the urbanisation gradient. Invertebrate biodiversity responses were investigated through ecological surveying (richness, abundance and biomass) of SUDS and comparable habitats as surrogates for nature-based solutions ('NBS'). Other important forms of NBS e.g. green roofs, river daylighting and tree-planting were beyond the scope of the study.

Mapping and random stratification was undertaken between 2021 and 2023, to target sampling locations. Satellite imagery was used to map impervious ground surfaces using the most recent 10 m Sentinel-2A data available at the time (Fig.1). Best available images were selected according to date (latest), cloud cover (minimum) and vegetation cover (greenest).

Mosaic images from the same day and time were compiled using images from 11am to 1pm to minimise shadows. Supervised image classification was undertaken in ArcGIS v.10.7. To resolve known problems with reflectivity from surface water (Hu et al., 2024), waterbodies were extracted using modified Normalised Difference Water Index (Xu, 2006) and official rivers datasets. Post-processing, filtering, boundary cleaning, and verification was undertaken using high resolution RGB image basemaps from ArcGIS and Google Earth. The classification model was re-trained using verified data, to produce ISA surface layers for each city.

Following production of the impervious surface layer for the whole region, city and surrounding area maps were graticulated into 1 km x 1 km grid cells, calculating the ISA for each cell (Fig.2a&2b). The limit of the urban area was defined as the contiguous connected grid cells with more than 25% of ISA, following Bonnington et al. (2014). After defining urbanisation limits, ISA% was calculated for each 1 km x 1 km grid cell (Fig.3). To identify sampling locations across all urban-rural gradients, five cells were selected randomly in each decile category (50 1 km x 1 km cells were selected in each city).

Within each of the 50 cells identified randomly in each city, and for NBS locations, pairs of sampling points were established within each 1 km x 1 km selected cell. The central point of each randomly selected cell (centroid) was taken as a control site location. NBS sites were then identified within the same grid cell but at least 150 m away from the control point, to reduce the effect of any spill over from the SUDS-like habitat influencing biodiversity at the control point. SUDS and comparable habitats were established as suitable monitoring sites by identifying ephemeral waterbodies, i.e. depressions in the land receiving runoff and featuring temporary standing or running water but not being watercourses, as well as designed systems expressly constructed as NBS (Supplement A). These ephemerally-wetted habitats were identified consistently across the cities and checked by multiple team members, to ensure they were actual SUDS or appropriate surrogates. Locations were identified using ArcGIS and Google Earth, and this process, along with fieldwork, was undertaken with the support of local partners from university and local authority institutions.

2.2. Fieldwork methods

Invertebrate sampling was carried out using sweep netting, beat sampling and pantrapping, all established methodologies widely applied in the assessment of biodiversity (Spafford and Lortie, 2013; Wheater and Cook, 2015). These are commonly used methods to sample invertebrates associated with low-lying vegetation across habitat types (Roulston et al., 2007). Sweep netting is considered a powerful tool for rapid sampling (Yi et al., 2012) avoiding bias towards population density and trapping susceptibility (Mazon and Bordera, 2008). Sampling was undertaken in Buenos Aires between 3rd –27th November 2022, in São Paulo between 2nd-25th March 2023, and in Barcelona between 7th-25th June 2023, each site being monitored once. Sweep netting was undertaken along 20 m linear habitats within each identified location (paired control point and NBS site), using a single 30 cm wide triangular sweepnet. Beat sampling took place at 10 points, spaced 2 m apart. Beating was undertaken using a metal pole (30 cm long, 3 cm diameter) to beat woody and tall herbaceous vegetation, collecting samples using a 50 cm x 50 cm tray consisting of white fabric. Yellow pantraps were

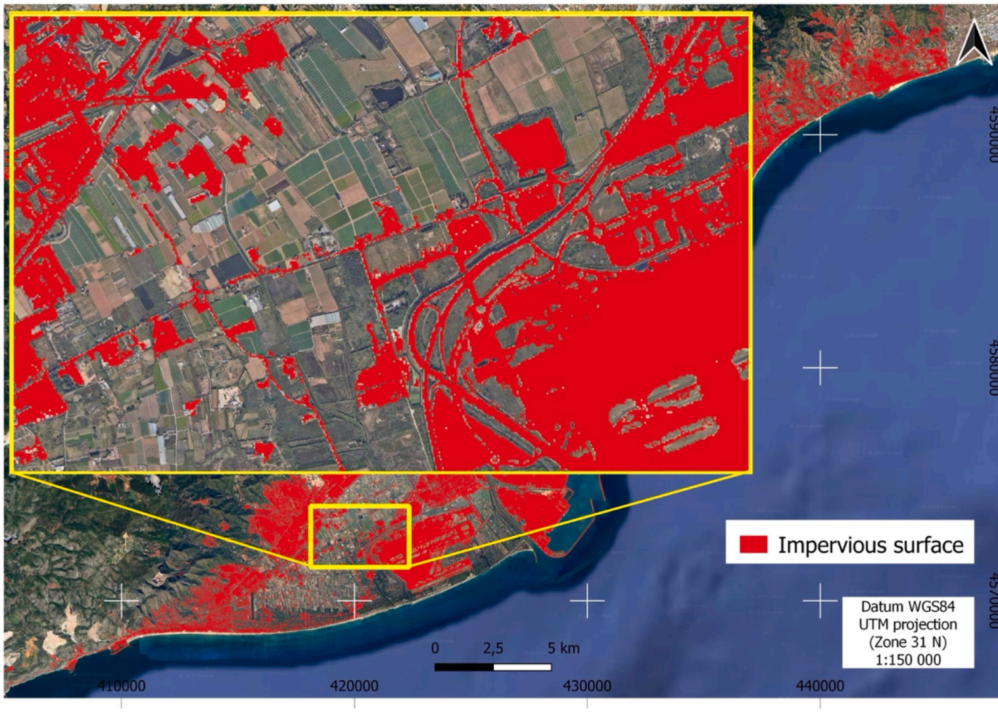

Fig.1. Mapping of impervious surface area using Sentinal-2A data (Barcelona, Spain).

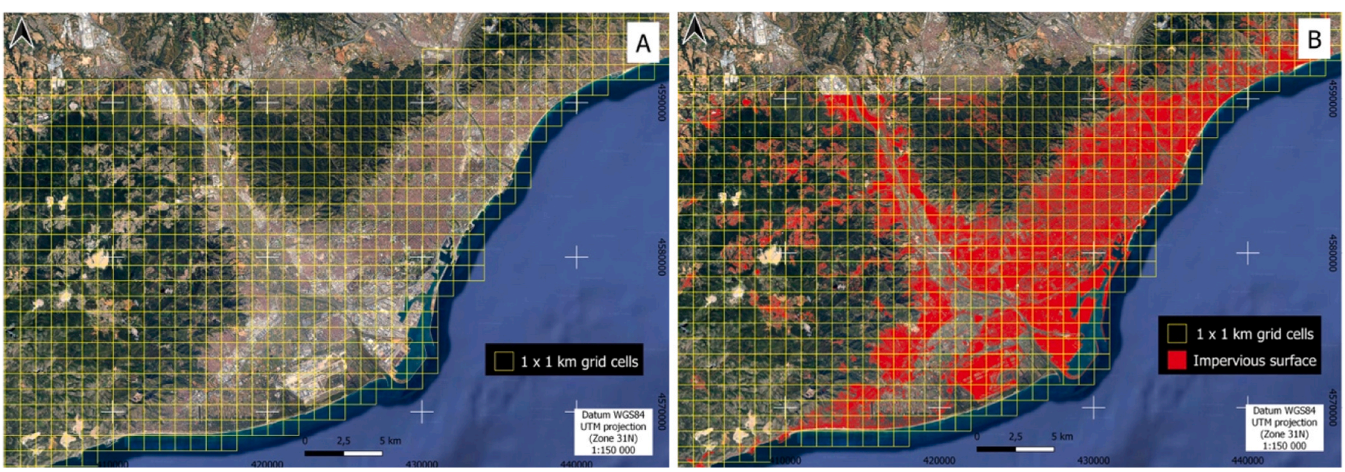

Fig.2. a. 1 km x 1 km grid cells (Barcelona, Spain). Fig.2b. Impervious surface area data associated with grid cells.

deployed for a continuous 24-hour period, containing water and odourless detergent i.e. unscented washing up liquid (see Laubertie et al., 2006; Saunders and Luck, 2013). Pantraps consisted of plastic bowls (15 cm diameter, 4 cm height) containing 200 ml of liquid solution. Pantraps were secured in the ground using bamboo skewers.

Invertebrates were identified to order level and then separated into morphotypes with the support of local experts (see Acknowledgements section), who also helped with storage of specimens. Morphotyping focused only on adult stages and for each group was based on key traits typically used for identification of that group e.g. patterns of wing venation for flying insects (Triplehorn and Johnson, 2005). A standardised measure of invertebrate biomass was obtained by weighing specimens using digital scales with 3 decimal digits (Multicomp Pro BAL1, Chicago, USA) following air drying on absorbent paper for a 10-minute time period (trials indicated this resulted in no further change in mass). Individuals were weighed several times until mass variation stabilised. Specimens were subsequently stored in individual plastic tubes with ethanol (70%). Specimens were sorted and classified in morphotypes according to the taxonomic order (e.g. Lepidoptera 1, Lepidoptera 2, etc). Specimens were labelled, noting sampling locations and dates. Collections were stored at: Argentina: University of Buenos Aires - Department of Planning and Landscape Design - Av. San Martín 4453 - C1417DSE - Buenos Aires; Brazil: Center for Nuclear Energy in Agriculture-University of São Paulo), Av. Centenário, 303 - CEP: 13416–000 – Piracicaba; Spain: Barcelona Regional – Carrer 60, 25 – 08040 – Barcelona. All sampling was undertaken by the same ecologist.

Invertebrate biodiversity was assessed by calculating abundance, richness and biomass of combined samples for each site. Surveying in several locations proved problematic, especially in the deployment and collection of pantraps, many of which were disturbed, turned over or removed. For these reasons, pantraps data were excluded from subsequent analyses. Sampling results were processed for 34 pairs of NBS and control sites in Barcelona, 36 pairs for São Paulo and 48 pairs for Buenos Aires.

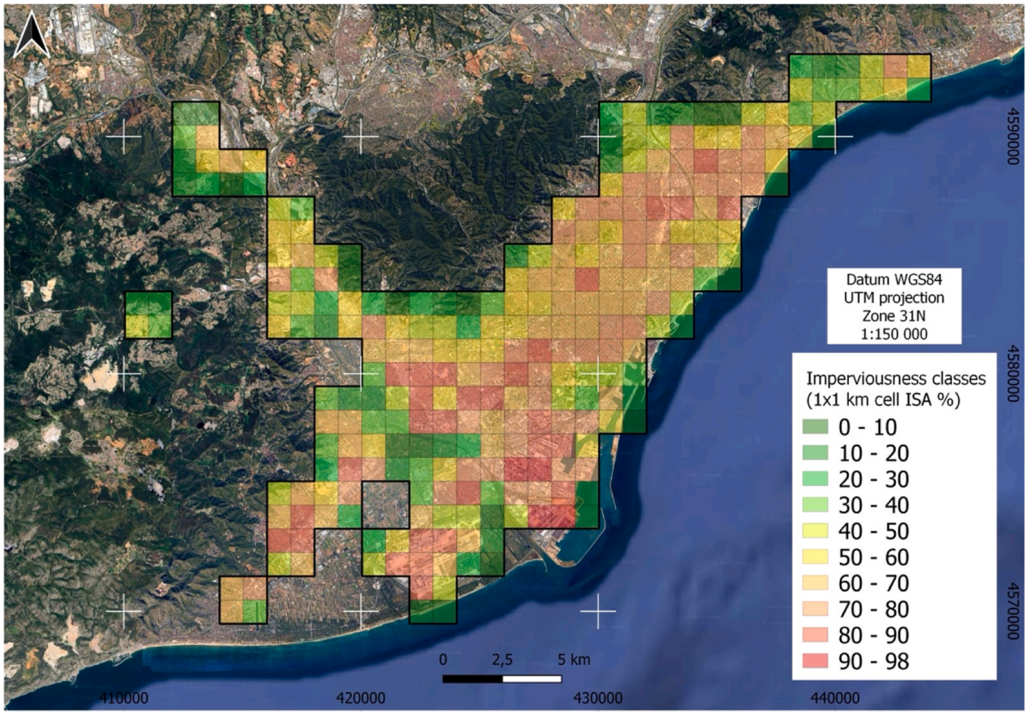

Fig.3. Deciles of imperviousness: 0–100% low-high impervious surface area (urbanisation), Barcelona, Spain.

2.3. Data analysis

To calculate influences of NBS presence on invertebrates in each randomly selected cell, biodiversity datasets for the controls and NBS sites were compared, addressing the null hypothesis that no difference would be evident. T-tests (unrelated, i.e. independent means) were performed to analyse differences using one-tailed hypotheses at significance level 0.01, for each indicator of invertebrate biodiversity (abundance, richness and biomass). Testing for normality of each dataset was performed using Kolmogorov-Smirnov tests, also for transformed datasets (square root; logarithms). Where data or transformed data were not normally distributed, non-parametric Mann-Whitney tests were performed to explore differences.

Correlations between imperviousness and differences in invertebrate biodiversity indicators were analysed using the Pearson coefficient, or Spearman's rank correlation coefficient where residuals were not normally distributed. To calculate differences, indicator values for each control site were subtracted from the paired values for each NBS site, Thus, total abundance for each control site was subtracted from the corresponding total abundance score for the paired NBS site, in each randomly selected cell. The same process was followed for invertebrate richness and biomass. Data were plotted showing each biodiversity indicator dataset (for each city) against the associated ISA value i.e. imperviousness levels within each 1 km x 1 km cell.

Next, treating datasets for controls and NBS sites separately, correlations between imperviousness and invertebrate biodiversity indicators were also tested for NBS and control sites, again using the Pearson coefficient (for transformed datasets) or Spearman's *rho* where data or transformed data were not normally distributed. All statistical testing was undertaken in Microsoft Excel with results being verified using the open-source statistical software Jamovi (Jamovi project, 2025).

3. Results

Analysis of sampling results indicated that invertebrate biodiversity indicators were affected by both imperviousness and presence or absence of NBS, for all measures including invertebrate abundance, richness and biomass. Table 1 shows the mean richness, given as totals for sweep net plus beat-sampling results, for each decile of imperviousness (research questions *RQ1&2*). This table presents: (a) mean richness for all control sites within each decile; and (b) mean richness for all NBS sites within the decile. Mean invertebrate richness tended to decrease with urbanisation intensity (ISA). Summary statistics of the available data shown in Annex 1, providing an overview of data quantity and quality.

3.1. Invertebrate biodiversity values for NBS and control sites (RQ1)

To examine influences of NBS presence on invertebrate biodiversity (*RQ1*), differences between abundance, richness and biodiversity for NBS versus control sites were tested as follows.

Difference in abundance: NBS sites exhibited higher invertebrate abundance ($M = 2.29$; $SD = 0.41$) compared to equivalents for the control sites following square root transformation ($M = 1.83$; $SD = 0.66$), demonstrating significantly higher biodiversity using this indicator (Table 2).

Difference in richness: A Mann-Whitney U test was performed to compare invertebrate richness for NBS sites versus control sites (one-tailed hypothesis). The results indicate that NBS sites exhibited significantly higher richness than random (control) sites (Table 2).

Difference in biomass: NBS sites also demonstrated higher invertebrate biomass (M =-0.87; $SD = 0.37$) compared to equivalents for the control sites following Log_{10} transformation ($M = -1.28$; $SD = 0.63$), with significantly higher biomass levels in NBS sites (Table 2).

Across all sampling sites, a total of 15 orders and 161 morphospecies were detected, with the majority of those orders being found in all study cities (Supplement B & Annex 2). The exceptions were: Diplopoda – not detected in Buenos Aires and São Paulo; Odonata - not detected in São Paulo and Barcelona; Homoptera – not detected in Barcelona; Neuroptera – not detected in Barcelona; Trichoptera – not detected in São Paulo. Coleoptera, Diptera, Hemiptera and Hymenoptera dominated our samples in each of the cities in terms of the number of morphotypes and individuals detected. Only one species was found in common between the three study cities, the western honey bee *Apis mellifera*. At the city

Table 1
Mean invertebrate richness for control and NBS sites - variation with urbanisation intensity in terms of imperviousness ISA deciles for all cities (standard deviations shown in brackets).

Imperviousness class – deciles (ISA)	0–10%	10–20%	20–30%	30–40%	40–50%	50–60%	60–70%	70–80%	80–90%	90–100%
All controls, all cities, mean richness (SD)	5.13 (1.45)	4.82 (1.03)	5.73 (1.60)	4.13 (1.31)	3.86 (1.41)	2.27 (0.68)	1.82 (0.92)	1.71 (0.75)	0.86 (0.64)	1.33 (0.47)
All NBS, all cities, mean richness (SD)	4.63 (0.48)	4.27 (0.75)	5.55 (0.89)	4.13 (1.02)	4.29 (1.98)	4.07 (1.00)	4.41 (1.29)	4.12 (1.45)	3.14 (0.64)	4.67 (1.89)

Table 2
Differences in invertebrate abundance (√), richness and biomass (Log_{10}) for NBS versus control sites.

Indicator of invertebrate biodiversity	Statistic	Significance	Effect size (Cohen's d)
Difference in abundance (√; NBS vs control)	T-test, unrelated (t), one-tailed	$t(234) = 6.47$, $p < .001$	$d = 0.85$
Difference in richness (NBS vs control)	Mann-Whitney (z), one-tailed	$Z = 4.88$, $p < .001$	n/a
Difference in biomass (Log10; NBS vs control)	T-test, unrelated (t), one-tailed	$t(228) = 6.06$, $p < .001$	$d = 0.80$

level, more orders were found for NBS than random sites for all cities studied (Annex 3). In only 8 of 42 cases were more orders found in random sites than in NBS sampled (Annex 3).

3.2. *Effects of NBS on invertebrate biodiversity across urban-rural gradients (RQ2)*

Differences between invertebrate abundance, richness and biomass scores – for each pair of sampling sites – are shown plotted against imperviousness in Figs. 4–6 respectively (all cities, all sites; sweep plus beat sampling totals). Differences for all three indicators between NBS site versus control site values increased with urbanisation intensity (ISA; *RQ2*). Imperviousness was most strongly related to differences in richness (Fig.5; Table 3), then abundance (Fig.4; Table 3) with biomass showing the weakest relationship (Fig.6; Table 3). Residuals were normally distributed for richness and abundance, but not biomass.

Comparable patterns were evident across the imperviousness gradient for each place, for the indicators of invertebrate abundance, richness and biomass (note coloured markers and distinct symbols for each city in Figs. 4–6; see also Annex 1 for summary statistics).

Taken together, the results indicate that invertebrate biodiversity values tended to decrease with increasing urbanisation intensity, yet the relative influence of NBS on invertebrate biodiversity tended to increase in proportion with imperviousness. Differences between NBS and control site values of invertebrate richness, abundance and biomass indicators tended to be larger and more discernible in more heavily urbanised areas, and less so in rural locations.

3.3. *Quantification and prediction of biodiversity effects of NBS (RQ3)*

To further investigate the impacts of NBS on biodiversity across imperviousness gradients, control site results were plotted separately from NBS site biodiversity levels (Figs. 7–9). Imperviousness and abundance (√) were found to be negatively correlated for the control sites, whereas for NBS sites, imperviousness and abundance (√) were not correlated (Fig.7a&7b; Table 4). This relationship between imperviousness and invertebrate abundance was effectively 'switched off' by the presence of NBS (Fig.7b; Table 4). Comparable patterns were observed for richness and biomass. Imperviousness and richness were found to be negatively correlated for control sites, but these variables were only weakly negatively correlated for NBS sites (Fig.8a and 8b; Table 4). Likewise, imperviousness and biomass (Log_{10}) were strongly negatively correlated for control sites, but only weakly negatively correlated for NBS sites (Fig. 9a and 9b; Table 4).

These results and significance of correlations are summarised in Table 4.

4. Discussion

4.1. *Invertebrate biodiversity values were higher for NBS than for control sites (RQ1)*

The results presented here indicate that relationships between the presence of NBS and biodiversity levels can be measured and quantified (RQ1) by assessing invertebrate richness, abundance and biomass. Invertebrate abundance, biomass and morphotype richness were significantly higher for NBS compared to random (control) sites (Table 2). Effects tended to be stronger for abundance and richness (Fig. 4 and 5) than for biomass (Fig.6; Fig.9); Table 4), which has been questioned as a proxy for biodiversity (Vereecken et al., 2021; see also Seress et al., 2020). The results indicate that invertebrate biodiversity was significantly higher for NBS locations (SUDS) compared with control sites (Table 2; Figs.7–9). The biodiversity effect of NBS that we demonstrate concurs with similar findings for interventions to restore

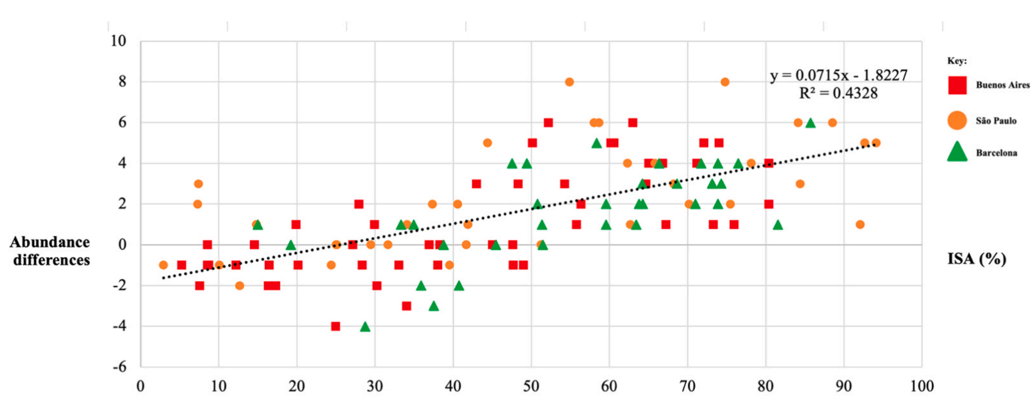

Fig.4. Relationship between Impervious Surface Area (ISA) and differences in invertebrate abundance (sweep net & beat sampling), all sites (3 cities): NBS minus control.

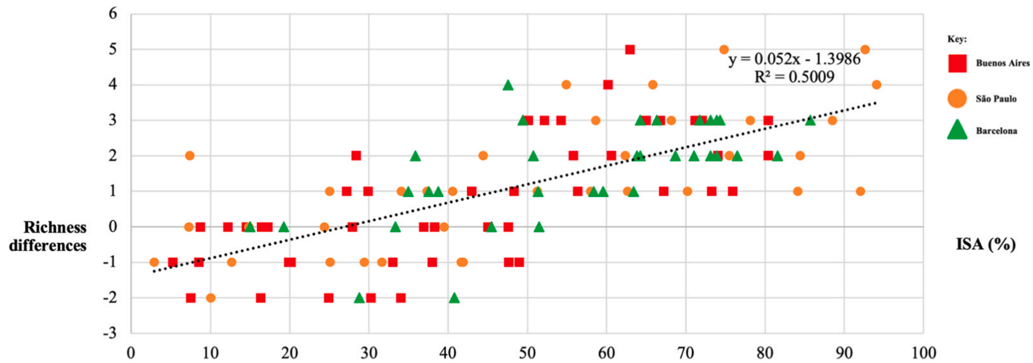

Fig.5. Relationship between ISA and differences in invertebrate richness (sweep net & beat sampling), all sites (3 cities): NBS minus control.

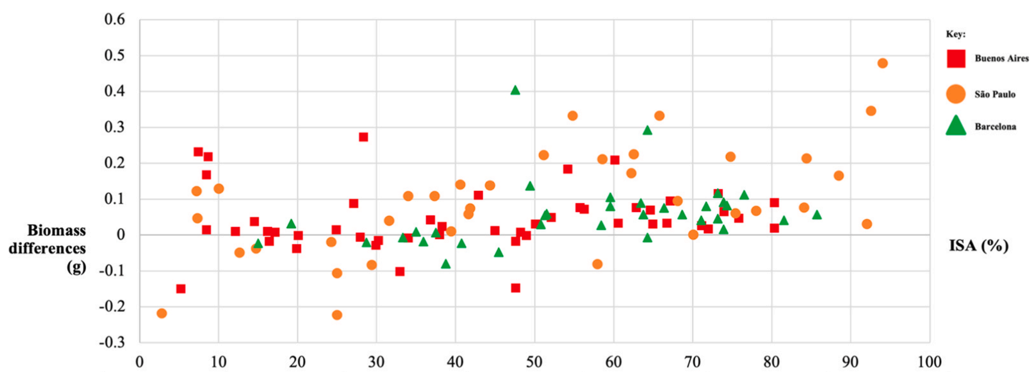

Fig.6. Relationship between ISA and differences in invertebrate biomass (sweep net & beat sampling), all sites (3 cities): NBS minus control.

Table 3
Correlations between imperviousness and differences in invertebrate abundance, biomass and richness for NBS versus control sites (summary).

Indicator of invertebrate biodiversity	Statistic	Significance
Difference in abundance (NBS vs control)	r (Pearson's correlation coefficient)	r(116) = .66, p < .001
Difference in richness (NBS vs control)	r (Pearson's correlation coefficient)	r(116) = .70, p < .001
Difference in biomass (NBS vs control)	rs (Spearman's rank correlation coefficient)	rs = .41, p (2-tailed) = 0

urban terrestrial ecosystems (Süle et al., 2025) and aquatic ecosystems, with invertebrate biodiversity being affected by the presence and makeup of vegetation (Baattrup-Pedersen et al., 2024).

Biodiversity benefits as measured using invertebrate communities have also been established for urban green roofs (Wooster et al., 2022), meadows as compared with grass lawns (Norton et al., 2019; Marshall et al., 2023), restored riverbank vegetation (Janssen et al., 2021) and urban rewilding sites (Taoran & Bingqin, 2021). Longitudinal studies have also exhibited improved invertebrate communities for urban stream restoration (Nguyen et al., 2024), with land-use and evaporation being important predictors. Few other studies have used insect biodiversity to assess the impacts of NBS such as urban blue-green interventions (Bowler et al., 2025).

Previous research has suggested that the benefits of NBS, including biodiversity impacts, have not been rigorously assessed (Seddon et al., 2020; Ryfisch et al., 2023; Nguyen et al., 2024). NBS impact assessment guidance has been criticised (Alva, 2022) and additional impact evaluation protocols are being developed to complement existing NBS

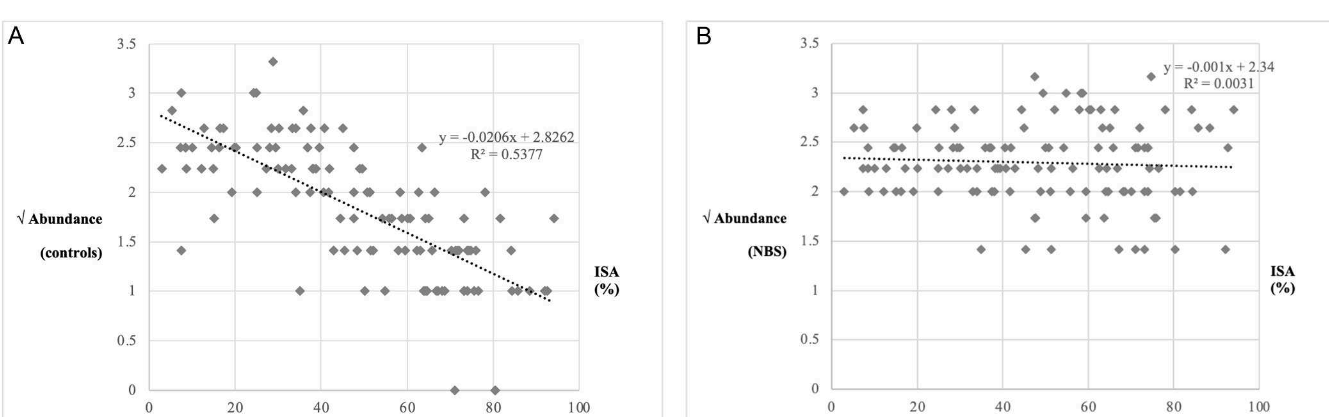

Fig.7. a. ISA vs. invertebrate abundance √ (controls). Fig.7b. ISA vs. invertebrate abundance √ (NBS).

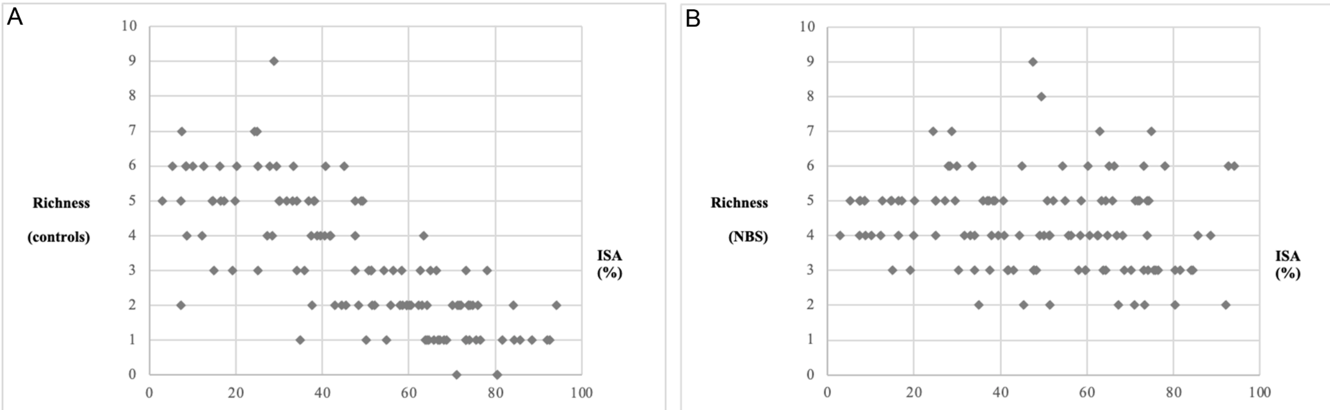

Fig.8. a. ISA vs. invertebrate richness (controls). Fig.8b. ISA vs. invertebrate richness (NBS).

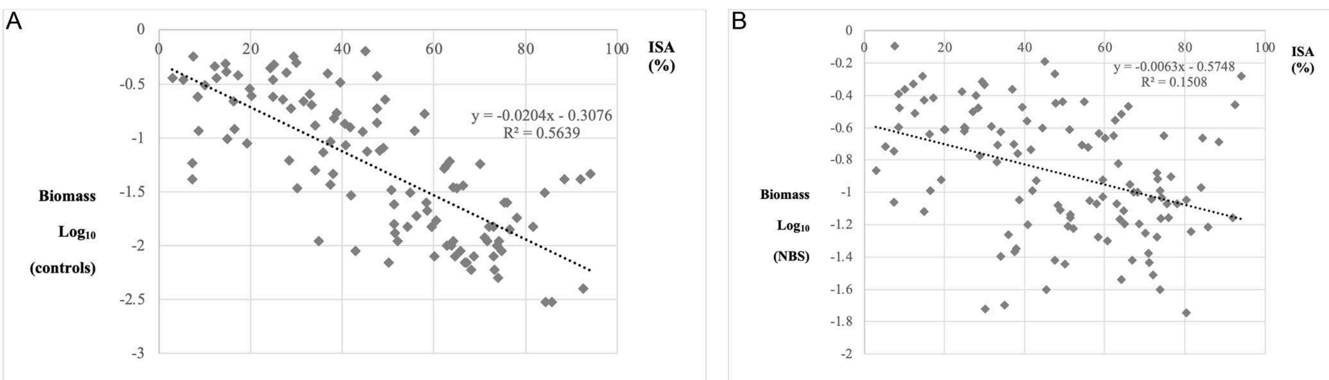

Fig.9. a. ISA vs. invertebrate biomass Log_{10} (controls). Fig.9b. ISA vs. invertebrate biomass Log_{10} (NBS).

Table 4

Correlations between imperviousness (ISA) vs. abundance (√), richness and biomass (Log_{10}) (for NBS and control sites).

Invertebrate biodiversity indicator	Statistic	Significance
Invertebrate abundance: control (√)	r (Pearson's correlation coefficient)	r(116) = -0.733, p < .001
Invertebrate abundance: NBS (√)	r (Pearson's correlation coefficient)	r(116) = -0.056, p = 0.55 (not significant)
Invertebrate richness: control	rs (Spearman's rank correlation coefficient)	rs(116) = -0.788, p < .001
Invertebrate richness: NBS	rs (Spearman's rank correlation coefficient)	rs(116) = -0.210, p = 0.022
Invertebrate biomass: control (Log10)	r (Pearson's correlation coefficient)	r(113) = -0.751, p < .001
Invertebrate biomass: NBS (Log10)	r (Pearson's correlation coefficient)	r(116) = -0.388, p < .001

assessment frameworks (van der Jagt et al., 2023; van Lierop et al., 2025). Paucity of biodiversity impact assessment evidence has been reported for various NBS types and associated sectors e.g. nature-based flood management (Hoegh-Guldberg et al., 2024), urban woodlands (Qu et al., 2025) and agroecological practices (Hartmann and Six, 2023).

Insects are among the most commonly used indicators to assess biodiversity, along with trees and marine benthic animals (Akoumianaki and Pakeman, 2023). Calls have been made to implement monitoring using replicated before-after-control-intervention (BACI) experiments to gauge impacts on invertebrates such as pollinators (Dorian et al., 2025). However, such trials are expensive, require long periods of time to account for colonisation delays, and critically, may not account for differential impacts across the urban-rural gradient, discussed below. This research focussed on SUDS-type NBS, and whilst the results may not be representative for some other forms of NBS such as green roofs (e.g. drier habitats), methods used here may be adapted. The results indicate that observational studies of invertebrate biodiversity can help fill key gaps in NBS impact assessment protocols.

4.2. Varying effects of NBS on invertebrate biodiversity across urban-rural gradients (RQ2)

The results indicate stronger effects of NBS presence on invertebrate biodiversity observed in urban locations i.e. at higher levels of imperviousness (Table 1 and 4). Significant correlations between imperviousness and differences in observed abundance, richness and biomass (Table 3) also support this conclusion. Diamond et al. (2023) found diverse patterns of insect biodiversity variation across urban gradients (positive, negative, level), contending also that nature-based urban infrastructure can yield associated biodiversity benefits.

Although higher NBS biodiversity levels were evident across entire urban-rural gradients (Fig.4-9) and Tables 2–4), imperviousness was correlated with the majority of biodiversity indicators for both NBS and control sites. The exception was for abundance in NBS sites (Fig.7b), where the relationship between biodiversity and imperviousness was effectively switched off, in sharp contrast with abundance in control sites (Fig.7a; Table 4). These findings concur with biodiversity analyses addressing habitat provision across geographical gradients undertaken for other biota, including mammals, birds and macrophytes (Bonnington et al., 2014; Thaweepworadej & Evans, 2022; Szoszkiewicz et al., 2025). This pattern of stronger correlations between imperviousness and biodiversity for control sites was particularly evident for the indicators of richness and abundance (Figs. 4,5). Again, invertebrate biodiversity differences between pairs of NBS and control sites were less strongly correlated for biomass. Similar patterns have been observed for avian biodiversity, and have been discussed in relation to the potential higher density of species being able to exploit urban environments (e.g. Chamberlain et al., 2017).

Mechanisms through which urban intensity (and imperviousness) might affect invertebrate biodiversity cannot be inferred from the present study, but the relevant literature offers additional insights. Effects of varying levels of urbanisation intensity upon terrestrial biodiversity remain poorly understood, but invertebrate levels can vary greatly with imperviousness gradients (Campos et al., 2024). More broadly, the impacts of urban development on biodiversity are well documented (McDonald et al., 2020) although causal mechanisms driving invertebrate loss are less clear (Eisenhauer et al., 2019). Increasing densification within cities, including infilling of urban green spaces, can also negatively impact biodiversity (Egerer et al., 2024).

Invertebrate communities are impacted by a broad range of pressures in urban environments including pollution (Ryalls et al., 2024; Qu et al., 2025), water quality (Koziel et al., 2019) and habitat quality, connectivity and management (Janssen et al., 2021; Bowler et al., 2025). Invertebrate biodiversity at greening intervention sites can be comparable with adjacent sites (Anderson et al., 2014) and regional drivers can 'overrule' the benefits of habitat enhancement measures at the local level, with diversity decreasing at greater levels of urbanisation intensity (Janssen et al., 2021). However, urbanisation is not the only pressure: Tommasi et al. (2021) report decreasing pollinator richness associated not only with imperviousness, but also with agricultural land cover.

Together, these findings and those in the literature suggest that NBS impacts may indeed vary greatly with urbanisation intensity (RQ2) and that potential biodiversity gains may be lower in more rural areas. At the same time, NBS implementation costs may be higher and private sector subsidy less viable in heavily urbanised settings (Wild et al., 2017; 2019). However, if NBS do offer significantly greater biodiversity benefits in more urbanised areas, this has important implications in the allocation of resources to restore ecosystems and their services where they are most degraded and lacking (Kowarik et al., 2025).

4.3. Quantification and prediction of NBS impacts on invertebrate biodiversity (RQ3)

Whilst the differences in NBS versus control site biodiversity were significant for invertebrate abundance, richness and biomass (Table 2 & 3), this did not translate into very strong correlation values when accounting for the effects of imperviousness (Table 4). Even though effect sizes were high for parametric tests of these differences (Table 2), relatively high values for standard deviations were also evident (Annex 1) and data were relatively scattered. Taken alone the datasets are not suitable for the development of more complex models integrating multiple predictor variables. It can be argued that doing so would require first modelling relationships between biodiversity metrics and direct drivers e.g. using threat-response models addressing pressure-impact relationships (Cooke et al., 2025). Mechanistic models for predicting insect responses show significant promise – these can go beyond the important contribution of correlative tools – but must address complex phenological, climatic and evolutionary factors (Maino et al. 2016). A further complication in the impact assessment of NBS on biodiversity relates to the challenges linked with time lags of establishment. The approach followed here focussed on testing well established features that were more likely to have been colonised by biota. This approach is promising, and future work can use these methods to assess if the results hold for established NBS in a wider range of settings.

Although our results confirm that imperviousness is strongly associated with invertebrate biodiversity the mechanisms through which these impacts operate in cities are not clear. Relevant mechanisms are manifold and complex, as is clear from the literature. Invertebrate biodiversity in study sites may have been affected by vegetation communities, cover and plant species, plus other factors at play in diverse urban biomes such as geographical isolation (Miyahara et al., 2022; Nania et al., 2024). This would be expected for NBS, which are vegetated systems. The biodiversity properties of SUDS as NBS for water management depends in part on vegetation makeup (Warner, 2022). NBS such as SUDS can also provide important stepping-stone habitat patches supporting functional connectivity between other green spaces (Bjørn and Howe, 2023). Other mechanisms through which urbanisation affects biophysical and biochemical processes are linked with diffuse pollution, land cover, material cycles, urban heat island and hydrological impacts (Novotny, 2002; Haase and Nuissl, 2007; Gaston et al., 2010; Jacobson and Ten Hoeve, 2012; Yan et al., 2015). Further work is required to assess which mechanisms might be involved in linking land use changes with invertebrate abundance, richness and biomass, and BACI experiments may be used to explore such cause-effect relationships (Dorian et al., 2025). The relatively high levels of scatter in Fig. 4-9 indicate that drivers other than NBS presence and imperviousness influence the relationships with invertebrate biodiversity. Due to such confounding factors in cities, the effects of NBS upon invertebrate biodiversity do not appear to be readily predictable (RQ3), even though they are measurable, using methods employed here.

4.4. Future research directions: imperviousness and vegetation as biodiversity predictor variables

A key limitation to our study was that we were unable to assess important aspects of invertebrate habitat relating to vegetation structure and diversity, soil depth and other environmental factors such as aspect, wind exposure and heat. Due to limited resources, the wide geographical scales involved, and data quantity needs, it was considered outside of the scope of this research to undertake detailed habitat surveys. The influence of vegetation makeup on the biodiversity qualities of NBS such as SUDS (Warner, 2022) warrants further attention. Taken together, imperviousness and vegetation characteristics may prove to be more powerful predictors of invertebrate biodiversity in NBS. Structural complexity in SUDS has been explored (Monberg et al., 2018) in terms of biodiversity benefits, but only using expert opinion; quantitative and qualitative experimental research is still required in this area. Future empirical research could build on the relevant literature as follows.

Urban aquatic NBS feature diverse macrophytes, providing invaluable habitats in urban settings (Szoszkiewicz et al., 2025). Studying these systems may also yield important information to help address known research gaps as regards SUDS design and management (Briers, 2014). Even basic interventions in terrestrial vegetation structure can increase biodiversity in urban green spaces, via increases in understorey vegetation and proportions of native plant species (Threlfall et al., 2017), and through linked environmental conditions such as shade, temperature and organic matter supply (Janssen et al., 2021). Similarly, in aquatic ecosystems, both vegetation surface area and macrophyte species diversity play key roles in determining macroinvertebrate abundance and richness (Baattrup-Pedersen et al., 2024). Keystone plant species, contributing much to biodiversity, stability and functions, may be prioritised for inclusion in NBS (Buckley et al., 2024) but this may also change with uncertain climate futures (Alizadeh & Hitchmough, 2020). As noted, a limitation of this study is that the results relate to SUDS-type NBS, and may not be applicable to NBS in general, although the methods can be applied more widely to other forms of NBS. Finally, the importance of impacts of vegetation management and NBS maintenance cannot be overstated, mediated through effects on flowering, biomass and productivity, and linked processes such as shading provision and evapotranspiration (Silva et al., 2021; Grilo et al., 2025). The importance of management has also been demonstrated for terrestrial urban habitats (Huchler et al., 2023).

Imperviousness and vegetation cover may affect invertebrate biodiversity – directly and indirectly – and singly or in combination. Teasing apart these two potential causal factors is not straightforward, reflects a limitation in our approach, and requires further research. Normalised Difference Vegetation Index (NDVI) is frequently employed in research to investigate habitat quality, biodiversity and land degradation at the city- scale (Pettorelli, 2013; Yengoh et al., 2015). The use of NDVI as a proxy for vegetated habitats in such research includes applications in the cities studied here (Gascon et al., 2016; Miyahara et al., 2022; Dobbs et al., 2023). This may point towards application of NDVI alongside imperviousness (ISA) to create more statistically powerful mixed models. However, a further complication relates to autocorrelation, which must be addressed in the development of predictive statistical models (Bahlai, 2023). NDVI and imperviousness tend to be correlated, and data closely related to NDVI are often used in the methods to calculate ISA (Guo et al., 2018; Yang et al., 2020; Xu and Fina, 2023). Our methodology for calculating imperviousness also involved the use of related datasets (Normalised Difference Water Index, NDWI; Xu, 2006). Thus the development of more complex, large spatial models integrating vegetation and imperviousness metrics shows promise, but this requires due caution.

4.5. Reflections on the role of observational studies of biodiversity in urban NBS

The findings reported here confirm that observational studies of invertebrate biodiversity have a key role to play, especially in addressing potential confounding factors i.e. geographical gradients (Table 1) and delayed onset of colonisation by biota post-construction of NBS. Alternatives to using invertebrates to assess urban NBS biodiversity include studying avian biodiversity (Thaweepworadej and Evans, 2022), macrophytes (Szoszkiewicz et al., 2025) and terrestrial plants (Fenoglio et al., 2023). Other routes include examining ecological landscape quality at city-regional scales, addressing pre-existing elements and reducing potential biases (Wójcik-Madej & Sowińska-Świerkosz, 2022; Marselle et al., 2021). Benefits of NBS to invertebrate biodiversity may apply even if harmful urban environmental impacts such as pollution on invertebrate communities remain within NBS sites (Briers, 2014). Whilst NBS implementation cannot negate most of the natural degradation effects of urbanisation or resulting homogeneity of biodiversity in large city-regions such as São Paulo (de Camargo et al., 2022), our findings do indicate positive effects of NBS on invertebrate biodiversity in urban areas. Protocols for biodiversity monitoring of NBS still require improvement (Farrell et al., 2020; Giuliani et al., 2024). The need to consolidate evidence on invertebrate biodiversity of NBS interventions remains a priority (Bowler et al., 2025), and can be supported through data publishing guides such as those newly available via the Global Biodiversity Information Facility (Lento and Schmidt-Kloiber, 2025).

5. Conclusions

The results presented here indicate that biodiversity gains achievable through NBS may be greater in more urbanised locations as determined through levels of imperviousness. Results were consistent across all study cities. Similar patterns were observed for invertebrate richness, abundance and biomass. Of these, biomass showed the weakest relationship in each city. Results for abundance and richness show stronger relationships, across the urban-rural gradient, with effect sizes being significantly higher in more urbanised locations. Little evidence has been published on the effects of SUDS on biodiversity, and we present rare empirical data on this. Our findings confirm that NBS impart biodiversity benefits in cities, in the form of invertebrate abundance, biomass and richness, in agreement with biodiversity benefits reported in the literature for other biota. NBS benefits can be seen to differ across the urban-rural gradient, with the difference between biodiversity in NBS versus control sites being larger in more impervious areas, i.e. biodiversity benefit effects appear to be greatest in more urbanised locations. Using well-established biodiversity monitoring techniques, discernible differences can be detected, whereby NBS habitats can be successfully tested to quantify the biodiversity benefits realised.

CRediT authorship contribution statement

Thomas Charles Wild: Writing – review & editing, Writing – original draft, Methodology, Funding aquisition, Formal analysis, Data curation, Conceptualization, Supervision. **Andre Rochelle:** Writing – review & editing, Methodology, Investigation, Formal anlaysis, Data curation. **Karl Evans:** Writing – review & editing, Supervision, Methodology, Formal analysis, Conceptualization.

Declaration of Competing Interest

We assert that there is no conflict of interest regarding the content of this manuscript and that the work has not been published elsewhere and is not under consideration by another journal.

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Annex 1. . Summary statistics.Invertebrate abundance, richness and biomass - sampling sites In Buenos Aires (Argentina), São Paulo (Brazil) and Barcelona (Spain)

	No. of sampling sites (n)	Mean abundance (SD)	Mean richness (SD)	Mean biomass (SD)
Buenos Aires (NBS sites)	48	5.25 (1.63)	4.35 (1.19)	0.2 (0.18)
Buenos Aires (Control sites)	48	4.19 (2.44)	3.63 (1.96)	0.16 (0.18)
Buenos Aires (All sites)	96	4.72 (2.13)	3.99 (1.66)	0.18 (0.18)
São Paulo (NBS sites)	36	5.92 (1.87)	4.36 (1.15)	0.24 (0.12)
São Paulo (Control sites)	36	3.56 (2.03)	3.11 (1.8)	0.14 (0.16)
São Paulo (All sites)	72	4.74 (2.28)	3.74 (1.63)	0.19 (0.15)
Barcelona (NBS sites)	34	5.15 (2.09)	4.21 (1.7)	0.11 (0.11)
Barcelona (Control sites)	34	3.44 (2.5)	2.68 (1.84)	0.05 (0.06)
Barcelona (All sites)	68	4.29 (2.44)	3.44 (1.92)	0.08 (0.09)

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	No. of sampling sites (n)	Mean abundance (SD)	Mean richness (SD)	Mean biomass (SD)
Totals: NBS sites	118	5.42 (1.86)	4.31 (1.34)	0.19 (0.15)
Totals: Control sites	118	3.78 (2.35)	3.19 (1.9)	0.12 (0.15)
Totals: All sites	236	4.6 (2.27)	3.75 (1.74)	0.16 (0.16)

Annex 2. . Numbers of morphotypes per order detected in each city (individuals nos. in brackets)

Study city	Buenos Aires			São Paulo			Barcelona		
Order	Random	NBS	All	Random	NBS	All	Random	NBS	All
Araneae	3 (6)	4 (9)	4 (15)	2 (4)	3 (4)	3 (8)	2 (3)	4 (6)	4 (9)
Blattodea	2 (9)	2 (8)	2 (17)	1 (1)	2 (4)	2 (5)	2 (3)	3 (6)	3 (9)
Coleoptera	8 (65)	7 (85)	10 (150)	5 (45)	5 (56)	5 (101)	6 (38)	11 (72)	11 (110)
Diplopoda	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	2 (2)	2 (2)
Diptera	3 (94)	4 (145)	4 (239)	2 (65)	5 (134)	5 (199)	7 (69)	11 (91)	13 (160)
Ephemeroptera	2 (3)	2 (14)	2 (17)	1 (7)	1 (5)	1 (12)	1 (1)	1 (2)	2 (3)
Hemiptera	7 (27)	7 (31)	8 (58)	4 (10)	6 (37)	6 (47)	4 (9)	4 (15)	6 (24)
Homoptera	1 (2)	2 (2)	2 (4)	2 (3)	2 (7)	2 (10)	0 (0)	0 (0)	0 (0)
Hymenoptera A (ants)	4 (56)	4 (69)	5 (125)	3 (36)	2 (48)	3 (84)	6 (38)	4 (38)	6 (76)
Hymenoptera (bees)	5 (28)	3 (32)	5 (60)	5 (22)	9 (43)	9 (65)	4 (7)	9 (46)	10 (53)
Lepidoptera	2 (4)	1 (1)	2 (5)	2 (4)	2 (5)	2 (9)	1 (1)	1 (1)	1 (2)
Mantodea	1 (1)	1 (3)	2 (4)	1 (2)	1 (4)	1 (6)	1 (1)	1 (3)	1 (4)
Neuroptera	1 (9)	1 (7)	1 (16)	1 (3)	1 (11)	1 (14)	0 (0)	0 (0)	0 (0)
Odonata	0 (0)	2 (3)	2 (3)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)
Orthoptera	3 (4)	3 (15)	3 (19)	1 (3)	2 (13)	2 (16)	1 (1)	3 (6)	3 (7)
Trichoptera	1 (3)	1 (2)	1 (5)	0 (0)	0 (0)	0 (0)	0 (0)	1 (2)	1 (2)

Annex 3. . Numbers of NBS and random sites at which invertebrate orders were found

Buenos Aires (Total no. of sites, n = 96)	**Random (No. of sites)**	**Random (% of all sites)**	**SUDS (No. of sites)**	**SUDS (% of all sites)**	**Both (No. of sites)**	**Both (% of sites)**
Araneae, Argentina	6	6%	8	8%	14	15%
Blattodea, Argentina	7	7%	3	3%	10	10%
Coleoptera, Argentina	33	34%	35	36%	68	71%
Diplopoda, Argentina	0	0%	0	0%	0	0%
Diptera, Argentina	29	30%	35	36%	64	67%
Ephemeroptera, Argentina	2	2%	9	9%	11	11%
Hemiptera, Argentina	20	21%	24	25%	44	46%
Homoptera, Argentina	1	1%	2	2%	3	3%
Hymenoptera A (ant), Argentina	27	28%	30	31%	57	59%
Hymenoptera B (bee), Argentina	17	18%	14	15%	31	32%
Lepidoptera, Argentina	4	4%	1	1%	5	5%
Mantodea, Argentina	1	1%	2	2%	3	3%
Neuroptera, Argentina	8	8%	6	6%	14	15%
Odonata, Argentina	0	0%	3	3%	3	3%
Orthoptera, Argentina	4	4%	15	16%	19	20%
Trichoptera, Argentina	3	3%	2	2%	5	5%
Sao Paulo (Total no. of sites, n = 72)	**Random (No. of sites)**	**Random (% of all sites)**	**SUDS (No. of sites)**	**SUDS (% of all sites)**	**Both (No. of sites)**	**Both (% of sites)**
Araneae, Brazil	2	3%	3	4%	5	7%
Blattodea, Brazil	1	1%	4	6%	5	7%
Coleoptera, Brazil	24	33%	18	25%	42	58%
Diplopoda, Brazil	0	0%	0	0%	0	0%
Diptera, Brazil	22	31%	26	36%	48	67%
Ephemeroptera, Brazil	6	8%	5	7%	11	15%
Hemiptera, Brazil	9	13%	21	29%	30	42%
Homoptera, Brazil	4	6%	7	10%	11	15%
Hymenoptera A (ant), Brazil	17	24%	16	22%	33	46%
Hymenoptera B (bee), Brazil	12	17%	24	33%	36	50%
Lepidoptera, Brazil	4	6%	5	7%	9	13%
Mantodea, Brazil	3	4%	4	6%	7	10%
Neuroptera, Brazil	3	4%	8	11%	11	15%
Odonata, Brazil	0	0%	0	0%	0	0%
Orthoptera, Brazil	3	4%	12	17%	15	21%
Trichoptera, Brazil	0	0%	0	0%	0	0%
Barcelona Total no. of sites, n = 68)	**Random (No. of sites)**	**Random (% of all sites)**	**SUDS (No. of sites)**	**SUDS (% of all sites)**	**Both (No. of sites)**	**Both (% of sites)**
Araneae, Spain	3	4%	6	9%	9	13%
Blattodea, Spain	3	4%	5	7%	8	12%
Coleoptera, Spain	20	29%	29	43%	49	72%

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Buenos Aires (Total no. of sites, n = 96)	Random (No. of sites)	Random (% of all sites)	SUDS (No. of sites)	SUDS (% of all sites)	Both (No. of sites)	Both (% of sites)
Diplopoda, Spain	0	0%	2	3%	2	3%
Diptera, Spain	23	34%	26	38%	49	72%
Ephemeroptera, Spain	1	1%	1	1%	2	3%
Hemiptera, Spain	7	10%	14	21%	21	31%
Homoptera, Spain	0	0%	0	0%	0	0%
Hymenoptera A (ant), Spain	17	25%	17	25%	34	50%
Hymenoptera B (bee), Spain	7	10%	18	26%	25	37%
Lepidoptera, Spain	1	1%	1	1%	2	3%
Mantodea, Spain	1	1%	2	3%	3	4%
Neroptera, Spain	0	0%	0	0%	0	0%
Ordonata, Spain	0	0%	0	0%	0	0%
Orthoptera, Spain	1	1%	5	7%	6	9%
Trichoptera, Spain	0	0%	2	3%	2	3%

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.ufug.2026.129368.

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