


ARTICLE

# Variation in ground-dwelling arthropod communities across differently managed infiltration basins

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## Abstract

As urbanisation continues to increase, terrestrial arthropod diversity declines. Urban green spaces represent unique opportunities to preserve arthropod diversity in urban environments. We quantify how ground-dwelling arthropod communities vary both seasonally and across three differently maintained stormwater infiltration basins in Ellisville, Missouri, United States of America. One basin was routinely mowed, a second was mowed only seasonally, and a third was intentionally planted to attract pollinators and was not mowed during this study. We expected higher plant diversity to correlate with higher arthropod diversity. Therefore, we expected the unmowed basin to have the highest levels of arthropod diversity and abundance and the mowed basin to have the lowest. Four collection periods spanned spring, early summer, late summer, and fall. During each collection period, five pitfall traps were placed throughout each basin for 48 hours. In total, 5686 specimens were collected and identified, representing 59 families. Arthropod communities did not vary across basins or collection dates, largely contrasting with existing literature. The results of this study do not indicate that different mowing regimens in infiltration basins will affect the ground-dwelling arthropod communities at a family level, although the effect on species-level diversity remains to be investigated.

## Introduction

Terrestrial arthropod populations and biodiversity are declining worldwide (van der Sluijs 2020). The extent of this decline is uncertain, largely due to a lack of global arthropod abundance monitoring, especially when compared with other, better-studied taxa (Wagner *et al.* 2021). Arthropod decline reporting is also geographically restricted, with most studies coming from western and northern Europe (Wagner 2020) and North America (Wagner *et al.* 2021). The scarcity of arthropod population data in other areas, especially tropical regions, is largely due to limited funding, hyperdiversity, and incipient taxonomy (Wagner *et al.* 2021). As such, estimates of the proportion of arthropods at risk for extinction vary widely. The Intergovernmental Science–Policy Platform on Biodiversity and Ecosystem Services (Bonn, Germany) cautiously estimates that 10% of insect species are at risk for extinction (Purvis *et al.* 2019), but other estimates place this number over 40% (Sánchez-Bayo and Wyckhuys 2019). Whatever the exact number is, terrestrial arthropod population collapse could spell disaster for the global ecosystem functions they make possible (van der Sluijs 2020).

Terrestrial arthropods serve many vital roles within their environments. They act as pollinators and decomposers (Weisser and Siemann 2008) and are important prey for many predatory

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animals (Denno *et al.* 2002). Insect herbivores break down organic matter within soil, aiding in decomposition, plant productivity, and nutrient cycling (Hartley and Jones 2008). Food web interactions through arthropod competition and predation are essential for ecosystem processes and have a range of positive, neutral, and negative outcomes on the plants and other components of the ecosystem (Sabelis *et al.* 1999; Janssen and Sabelis 2008). Predatory spiders provide biological control, suppressing pest species within an ecosystem (Michalko and Pekár 2017). Certain soil arthropods, such as ants, termites, isopods, and millipedes, are considered ecosystem engineers: they alter their environment through bioturbation, reworking the soil by incorporating plant litter and other organic matter (Gonçalves *et al.* 2021). Loss of arthropod populations could have far-reaching cascading effects, leading to biodiversity loss at higher trophic levels while also impairing ecosystem resilience and stability (van der Sluijs 2020).

Many arthropods also are indicator species due to their niche partitioning and ease of monitoring (Hunter and Jaros-Su 1997). For example, some aquatic ecosystems rely on aquatic insect species as indicators for ecological integrity (Rosenberg and Resh 1993). In other cases, various arthropods have been used to monitor environmental pollution due to their sensitivity to environmental change (Parikh *et al.* 2020). Soil arthropods can act as bioindicators for soil quality and can be used for more sustainable management of agroecosystems (Gonçalves *et al.* 2021). Losing these indicators would mean loss of important research tools for ecosystem health monitoring.

Along with pollution, diseases and parasites, and climate change, the main driver of these global declines is habitat change caused by human activities, including agriculture, industrialisation, and urbanisation (Sánchez-Bayo and Wyckhuys 2019). Urbanisation has directly impacted terrestrial arthropod abundance and diversity by way of pollution, habitat loss and fragmentation, and land conversion (Shuisong *et al.* 2013; Piano *et al.* 2020). Urban land cover since 2000 is projected to triple by 2030 (Seto *et al.* 2012). As urbanisation increases, these arthropod communities are at risk of further declines, extirpation, and possible extinction (Shuisong *et al.* 2013).

Although urbanisation has caused overall declines in biodiversity, urban green spaces have proven to be essential tools for maintaining high levels of biodiversity within urban areas (Mata *et al.* 2017). Also referred to as “green infrastructure,” urban green spaces constitute any area of vegetation within an urban landscape. They include parks, community gardens, lawns, golf courses, cemeteries, green roofs, and sporting fields (Gairola and Noresah 2010; Wolch *et al.* 2014).

Green spaces can maintain diverse communities of insects and other arthropods, and these communities can differ significantly from those found in the surrounding nonurban areas (Burkman and Gardiner 2014). In one study in the Swiss Plateau (Switzerland), urban areas were shown to support species that otherwise either did not appear or were scarce within the study region, with 13.3% of species being recorded only in urban areas (Sattler *et al.* 2011). Unique abiotic factors in urban areas contribute to these differences (Dale and Frank 2018). For example, cities tend to be hotter than surrounding rural areas, due to the urban heat island effect (Yang *et al.* 2016). Arthropods respond to this differently, depending on their latitude, with arthropods in higher-latitude areas generally increasing in abundance (Youngsteadt *et al.* 2016). As climate change continues to alter global temperatures, some ant species have been reported to migrate to urban areas from lower altitudes because of their higher temperatures and drier climates (Menke *et al.* 2011). Generalist arthropods like some ants and spiders are often more resistant to the effects of urbanisation, further changing arthropod community composition in these areas (Thompson and McLachlan 2007; Lowe *et al.* 2017). As such, urban green spaces not only represent opportunities to sustain arthropod communities comparable to those in surrounding areas but are often distinct ecosystems with their own unique arthropod communities and arthropod-driven ecosystem services.

Ground-dwelling arthropod communities have been shown to vary across differently maintained urban green spaces (Mata *et al.* 2017). Increased mowing frequency generally corresponds with lower plant and arthropod diversity (Watson *et al.* 2019; Proske *et al.* 2022). Conversely, reduced mowing frequency in urban green spaces has been shown to increase plant diversity (Chollet *et al.* 2018; Sehrt *et al.* 2020). Green spaces with higher plant diversity and richness generally have more diverse arthropod communities (Haddad *et al.* 2001).

Ground-dwelling arthropod community composition within a site also changes throughout the year. Arthropod seasonality differs between species even within the same site, as different species respond variably to changes in temperature, day length, and humidity (Wolda 1988). Many arthropod life cycles are largely temperature-driven, with the timing of different milestones such as oviposition, egg hatching, and emergence from pupation depending on outside temperature (Powell and Logan 2005). This changing thermoperiod, combined with changing photoperiods as day length changes during the year, is a primary driver of terrestrial arthropod presence and activity (Beck 1983). In one study concerning temporary pools within a temperate urban green space in Buenos Aires City, Argentina, seasonality was shown to be a primary driver of changing arthropod community structure (Fontanarrosa *et al.* 2009).

In this study, we examined how higher plant diversity in three differently managed stormwater infiltration basins corresponds with ground-dwelling arthropod diversity. Infiltration basins are tools used to manage storm runoff by infiltration of water through the soil into the groundwater aquifer. They are distinct from retention basins, which permanently hold water, and detention basins, which manage runoff through discharge to a surface water body. The basins can be planted with various kinds of foliage to aid the infiltration of water through the soil, allowing for more effective removal of pollutants from the runoff (Pekarek *et al.* 2011).

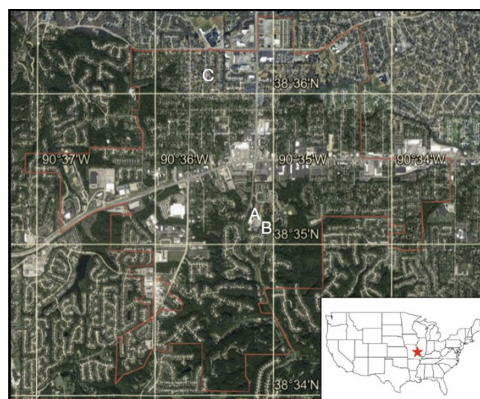
The three study sites were chosen based on their different maintenance regimens. We expected the basins with higher plant diversity and richness to contain higher levels of arthropod diversity. One basin, the “mowed basin,” had been planted with a small variety of short grasses and was routinely mowed: we expected it to have the lowest plant and ground-dwelling arthropod diversity. Another basin, the “reduced mowing basin,” was maintained seasonally, being mowed only once or twice per season. We expected its plant and ground-dwelling arthropod diversity to be higher than the mowed basin but lower than the third basin, the “unmowed basin,” which had been planted intentionally to attract pollinators and was not mowed during the study period.

Arthropods were collected from the three basins across four different collection periods: spring, early summer, late summer, and fall. We identified the collected arthropods to family level, as family-level diversity can be used as a proxy for species-level diversity, albeit with trade-offs between identification costs and more comprehensive data (Roy *et al.* 1996; Zou *et al.* 2020). We expected arthropod community composition across all basins to vary seasonally due to climatic variation, as outlined by previous studies (Wolda 1988; Thomsen *et al.* 2015)

## Methods

### Study sites

Three infiltration basins in Ellisville, Missouri, United States of America were selected because of their differing maintenance regimens (Fig. 1). The mowed basin (38.585331, -90.585043) is approximately 250 m<sup>2</sup> and had been planted with short grasses, mainly *Festuca* spp., that were routinely mowed. It was initially constructed as an inlet to keep water off of a nearby trail but was rebuilt as an infiltration basin in 2008. The reduced mowing basin (38.586388, -90.585844) was built in 2008 and is approximately 593 m<sup>2</sup>. It usually was overgrown with tall grasses and weeds and underwent maintenance only seasonally. Plants present at this basin included *Lonicera maackii* (Ruprecht) (Caprifoliaceae) and *Eupatorium* Linnaeus (Asteraceae). The unmowed basin (38.601222, -90.592673) is approximately 75 m<sup>2</sup> and had been intentionally planted with plants



**Figure 1.** Map showing location of basins (A, mowed basin; B, reduced mowing basin; and C, unmowed basin) in Ellisville, Missouri, with bottom-right inset showing location of Ellisville (red star) within the United States of America.

that attract pollinators, including *Sonchus asper* (Linnaeus) (Asteraceae) and *Asclepias incarnata* Linnaeus (Apocynaceae). It was constructed in 2020 and was not mowed during this study. The mowed and reduced mowing basins are approximately 170 m apart, and the unmowed basin is located approximately 2.4 km from them.

### Collection methods

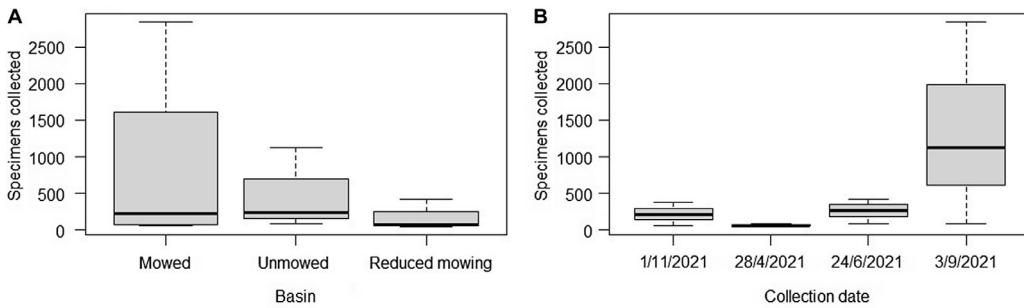
To compare ground-dwelling arthropod diversity across basins, pitfall traps were set at each basin for four 48-hour periods across 2021: 25–27 April (spring), 22–24 June (early summer), 1–3 September (late summer), and 30 October–1 November (fall). We restricted sampling to 48-hour periods that were forecasted to have no precipitation to ensure traps would not be flooded. Five pitfall traps were placed in an approximate “X” pattern at each basin, with one trap set at each point (the end of each arm) of the “X” and a fifth set at the centre. The traps were spread evenly across each basin, with each point of an “X” being approximately 1–2 m from that basin’s perimeter. Each trap consisted of one plastic cup (5.7 cm bottom diameter, 9.2 cm top diameter, 11.7 cm height) with a funnel secured in the top to allow arthropod entry but prevent escape. The bottom of each trap was filled with approximately 100 mL of 70% ethanol. The traps were installed flush to the ground and were intended to collect primarily ground-dwelling arthropods, such as beetles, spiders, ants, and crickets.

After 48 hours, the contents of each trap were emptied into Ziploc plastic bags and labelled accordingly. All specimens were then driven to the home lab setting, where they were identified to family level. Identification was completed using a Solomark Portable Stereo Microscope (<https://www.solo-mark.com>). Specimens were identified using the key in Marshall (2006) and supplementary identification materials, including Bradley (2012) and online resources such as iNaturalist (<https://www.inaturalist.org>) and BugGuide.net (<https://bugguide.net/node/view/3/bgpape>). Taxon concepts and taxonomic classification were derived from Marshall (2006) and Bradley (2012).

To determine differences between basins, plant diversity and richness for each basin were quantified. For each pitfall trap, all plants within a 1-m radius were identified. To properly quantify the plant diversity, both generic flora richness and abundance were assessed. Plants were identified to genus level by primarily using online resources such as iNaturalist.

### Statistical methods

Data analysis was completed using R, version 4.1.2 (R Core Team 2021). We calculated Shannon diversity and total individual counts of arthropods at each basin and collection date



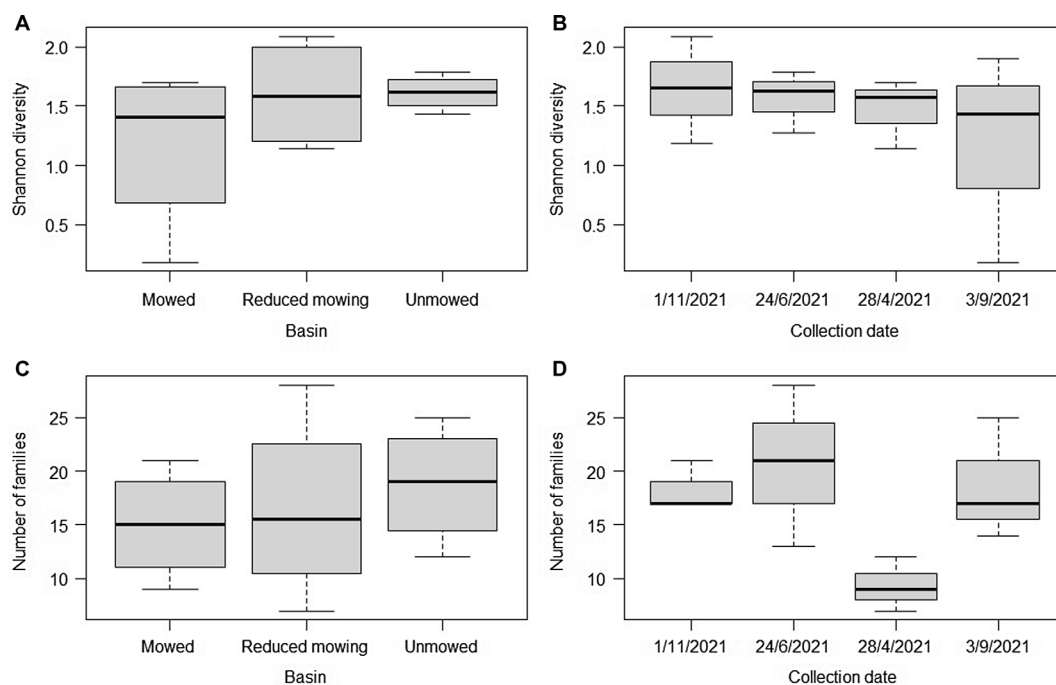
**Figure 2.** Arthropod abundance at each **A**, basin and **B**, collection date. There are no significant differences in abundance across basin types or collection dates.

using the *vegan* package, version 2.5.7 (Oksanen *et al.* 2020). Shannon diversity indices were also calculated for the plants at each basin. Number of families at each basin and collection date were calculated using the *rich R* package, version 1.0.1 (Rossi 2011). To test for differences in Shannon diversity and number of families, we ran two-way analyses of variance, with *basin* and *date* as independent variables. Shannon diversity and the number of families met the assumptions of parametric tests, whereas abundance had an extreme outlier, so we used Kruskal–Wallis tests for total counts for basin and collection date. To determine if community composition varied across basins and dates, permutational multivariate analyses of variance were completed using the *vegan* package, version 2.5.7 (Oksanen *et al.* 2020) and data were plotted with nonmetric multidimensional scaling plots using *ggplot2*, version 3.3.5 (Wickham 2016) and *ggrepel*, version 0.9.1 (Slowikowski 2021). Bray–Curtis dissimilarity distance was used for both the nonmetric multidimensional scaling plots and the permutational multivariate analyses of variance.

## Results

In total, 5686 arthropod specimens were collected and identified across the three basins, representing 59 families (Supplementary material, Table S1). Although abundance of specimens did not differ significantly between basins ( $H(2) = 1.4231$ ,  $P = 0.49$ ) or collection date ( $H(3) = 6.0769$ ,  $P = 0.11$ ), 3365 specimens were collected at the mowed basin, whereas 1700 specimens were collected at the unmowed basin, and 621 specimens were collected at the reduced mowing basin (Fig. 2A). The collection period yielding the highest arthropod abundance was the late summer period, whereas the spring period yielded the lowest arthropod abundance (Fig. 2B). Of the 3365 specimens collected from the mowed basin, it should be noted that 2837 were collected during one collection date, 97% of which were ants. Removing this sampling does not change the diversity relationship between the basins or collection dates, nor does it significantly impact the arthropod community composition across basins or dates. However, with this date removed, the unmowed basin would be the site with the most abundant collection.

The unmowed basin had the highest mean Shannon diversity index (1.612), whereas the mowed basin had the lowest mean Shannon diversity index (1; Fig. 3A), although no significant differences in Shannon diversity were observed between basins ( $F_{2,6} = 0.83$ ,  $P = 0.48$ ) or collection dates ( $F_{3,6} = 0.42$ ,  $P = 0.745$ ; Fig. 3B). In addition, no difference in the number of arthropod families was found between each basin ( $F_{2,6} = 0.50$ ,  $P = 0.632$ ; Fig. 3C) or collection date ( $F_{3,6} = 2.67$ ,  $P = 0.142$ ; Fig. 3D). Furthermore, the permutational multivariate analyses of variance revealed no significant differences in arthropod community assemblage for basin ( $F_{2,11} = 0.81$ ,  $P = 0.61$ ; Fig. 4A) or collection date ( $F_{2,11} = 1.35$ ,  $P = 0.23$ ; Fig. 4B).



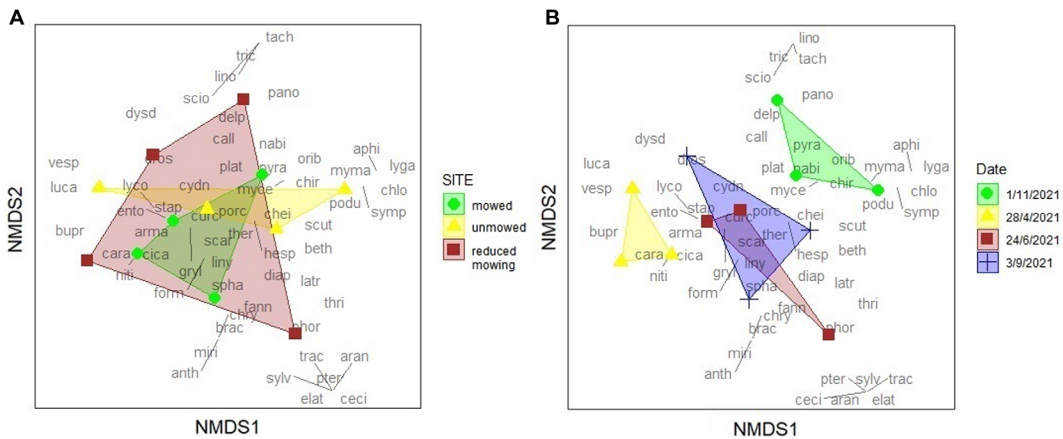
**Figure 3.** Shannon diversity index for each **A**, basin and **B**, collection date; number of families at each **C**, basin and **D**, collection date. There are no significant differences in diversity or number of families across basin types or collection dates.

Plant genera richness was highest at the reduced mowing basin and lowest at the mowed basin. The mowed basin had the lowest diversity index (approximately 0.78). Flora at this basin was primarily spread across three grass genera, *Festuca* Tournefort (Poaceae), *Digitaria* Haller (Poaceae), and *Sorghum* Moench (Poaceae). The reduced mowing and unmowed basins' Shannon diversity indices were more than double that of the mowed basin, with values of 1.91 and 1.94, respectively.

## Discussion

This study did not find variation in arthropod diversity at the family level across basin types. This observation contrasts with existing literature, as the unmowed and reduced mowing basins both had higher plant diversity and generic richness than the mowed basin did, and those characteristics were expected to lead to significantly higher arthropod diversity in these two basins (Haddad *et al.* 2009).

Certain families accounted for a large proportion of all arthropods collected and appeared in all three basins. These included Lycosidae, Formicidae, Carabidae, and Gryllidae. Some of these families, such as Lycosidae, have relatively high proportions of generalist species (Bedford and Usher 1994; Major *et al.* 2006). Generalist species are often more resistant to disturbance (Niemelä and Kotze 2009; Lowe *et al.* 2017), which may partially explain why some of these families appeared at all three sites, leading to a lack of variation. Families that appeared only in one basin, such as Curculionidae (mowed basin), Scarabaeidae and Trachelidae (reduced mowing basin), and Lucanidae (unmowed basin), were collected infrequently and subsequently accounted for a small proportion of all arthropods collected. Identification of arthropods to species level could reveal more significant differences between the communities, showing more clearly how adaptive



**Figure 4. A**, Nonmetric multidimensional scaling plot generated for arthropod communities across basins, showing significant overlap between communities; **B**, nonmetric multidimensional scaling plot generated for arthropod communities across collection dates, showing minimal overlap between fall and spring collection dates, with significant overlap between both summer collection dates. Abbreviations: anth, Anthocoridae; aph, Aphididae; aran, Araneidae; arma, Armadillidiidae; beth, Bethylinidae; brac, Braconidae; bupr, Buprestidae; call, Calliphoridae; cara, Carabidae; ceci, Cecidomyiidae; chei, Cheiracanthiidae; chir, Chironomidae; chlo, Chloropidae; chry, Chrysomelidae; cica, Cicadellidae; curc, Curculionidae; cydn, Cydnidae; delp, Delphacidae; dros, Drosophilidae; dysd, Dysderidae; elat, Elateridae; ento, Entomobryomorpha; fann, Fanniidae; form, Formicidae; gryl, Gryllidae; hesp, Hesperidae; latr, Latridiidae; lino, Linotaeniidae; liny, Linyphiidae; luca, Lucanidae; lyco, Lycosidae; lyga, Lygaeidae; miri, Miridae; myce, Mycetophilidae; myma, Mymaridae; nabi, Nabidae; niti, Nitidulidae; orib, Oribatida; pano, Panorpidae; phor, Phoridae; plat, Platygastridae; podu, Poduromorpha; porc, Porcellionidae; pter, Pteromalidae; pyra, Pyralidae; scar, Scarabaeidae; scio, Sciomyzidae; scut, Scutigeridae; spha, Sphaeroceridae; stap, Staphylinidae; sylv, Silvanidae; symp, Symphyleona; tach, Tachinidae; ther, Theridiidae; thri, Thripidae; trac, Trachelidae; tric, Trichoceridae; vesp, Vespidae.

radiation within families could lead to specialisation. Higher-taxon surrogacy was used due to time and resource constraints, but correlations with species composition at a local scale have been shown to decrease at the family level as compared to the genus level (Mandelik *et al.* 2007).

Although no significant differences were observed in the arthropod communities across collection dates, possibly due to lack of replication and small sample size, there are indications that these data follow similar phenological patterns to those shown in other similar studies. The early and late summer dates contained the highest overlap, whereas the arthropod communities collected in the spring and fall overlapped less with each other and with either of the summer dates. Arthropod communities have been found to experience significant turnover throughout the year (Thomsen *et al.* 2015; Seifert *et al.* 2021): it therefore makes sense that the more temporally distant collection dates yielded less overlap than the temporally closer ones did. This annual turnover can partially be attributed to bottom-up processes driven by temporal (Ekholm *et al.* 2019; Shinohara and Yoshida 2021) and spatial (Sobek *et al.* 2009) turnover of plant species. In one study, Auchenorrhyncha species abundance changed significantly across the year and related directly to the number of grass structures present at the site, which peaked in late summer (Stinson and Brown 1983). Those results align with the current study's most abundant collection period, the late summer period, although it was not more significantly abundant than the other collection dates. Analysis of seasonal variation in plant diversity, richness, spatiality, and architecture would help to determine the factors underlying these seasonal changes in arthropod community composition.

Due to time, resource, and spatial constraints, the replication level and sample size of this study were limited and may have contributed to the lack of variation. A larger sample size could be accomplished by placing more traps and by incorporating multiple types of traps. Increasing trap

abundance and diversity would also allow for a greater variety of arthropods to be sampled. The highly species-specific nature of arthropod response to changing vegetation could mean that the inclusion of more flying arthropods, in addition to ground-dwelling arthropods, in studies may reveal more noticeable differences between the communities. The addition of traps specialising in capturing flying insects, such as malaise traps, yellow pan traps, and light traps, could accomplish this (Devigne and de Biseau 2014).

The location and size of the basins also may have contributed to the homogeneity of the collected arthropod community because the infiltration basins selected were all located relatively close to each other, especially the mowed and reduced mowing basins. The unmowed basin was significantly smaller and newer than the other basins, which may have contributed to its lower plant richness and abundance. Due to the even placement of traps across each study site, the traps in the unmowed basin were placed more closely together than those in the mowed or reduced mowing basins were, and this may have contributed to the variation in the collected arthropod abundance between sites. Because many traps were placed close to the basins' edges, edge effects may have also skewed the results. Arthropod community composition can change significantly at habitat edges through species-specific responses that depend on resource requirements (Wimp and Murphy 2021). Because all of the basins were relatively small, their edge densities were relatively large, with a significant portion of each basin's total area being taken up by edge habitat. Edge density can negatively affect overall arthropod diversity while favouring generalist species (Peng *et al.* 2020). This relationship may have contributed to the overall homogeneity between the basins. A future study producing results with higher statistical power could aid city councils and urban planners in building infiltration basins, as well as urban green spaces in general, that are more conducive to ground-dwelling arthropod diversity. It is essential that urban planning considers arthropod diversity to mitigate the detrimental effects of urbanisation on overall biodiversity.

**Supplementary material.** To view supplementary material for this article, please visit <https://doi.org/10.4039/tce.2024.8>.

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**Competing interests.** The authors declare that they have no competing interests.

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