Building and Environment 228 (2023) 109865

Contents lists available at ScienceDirect

Building and Environment

journal homepage: www.elsevier.com/locate/buildenv

Contribution of green roofs to urban arthropod biodiversity in a Mediterranean climate: A case study in València, Spain

Vicent Benedito Durà [a,*], Enric Meseguer [b], Carmen Hernández Crespo [c], Miguel Martín Monerris [c], Ignacio Andrés Doménech [c], M. Eugenia Rodrigo Santamalia [d]

[a] *Departamento de Ingeniería Hidráulica y Medio Ambiente, Universitat Politècnica de València, Camino. de Vera s/n, 46022, València, Spain*
[b] *Departamento de Ecosistemas Agroforestales, Universitat Politècnica de València, Camino de Vera s/n, 46022, València, Spain*
[c] *Instituto de Ingeniería del Agua y Medio Ambiente. Universitat Politècnica de València, Camino. de Vera s/n, 46022, Valéncia, Spain*
[d] *Instituto Agroforestal Mediterraneo, Universitat Politècnica de València, Camino de Vera s/n, 46022, Valencia, Spain*

ARTICLE INFO

Keywords:
Arthropods
Green roofs
Biodiversity
Mediterranean
SUDs

ABSTRACT

Studies on biodiversity on green roofs have been carried out in parallel to the degree to which these infrastructures have been implemented in different countries. There are no studies about biodiversity of fauna in these habitats in Spain and other countries of Mediterranean Europe, except France. This study compares the arthropod community of a green roof to that of a conventional flat roof with gravel covering and two ground-level gardens, in a geographical area where the territory is a matrix of urban and agricultural lands. The interest of the results is twofold: no such work is available in this Mediterranean region, and the stressful environmental conditions. No significant differences appeared for relative abundance, taxa richness, Shannon Index, and effective number of taxa between the studied habitats. However in taxa composition and abundance terms, the arthropod community on green roofs differed significantly from that of ground-level gardens, and exclusive species were captured in both habitats. The relative abundance of the different functional groups captured showed different seasonal patterns in ground level gardens and green roof. Our results also confirm that green roofs significantly increase biodiversity compared to conventional roofs. Thus, in the climate conditions of Mediterranean region, our study suggests that green roofs may also be suitable solutions to increase and improve the arthropods biodiversity in urban areas.

1. Introduction

The main problem that the human population faces, and which acts in combination with climate change, is biodiversity loss on our planet. Before the onset of global warming, human populations were already causing biodiversity loss in the natural environment [1,2]. In simple and practical terms, biodiversity can be defined as "the number of species", although more detailed definitions have been proposed by Ref. [3] or by the Convention on Biological Diversity [4]. The main causes of biodiversity loss are pollution, hunting, invasive species, overexploitation of some species, climate change, natural disasters and habitat loss, the last of which is the most important [5–9]. The fact that wildlands, that is those lands with no human occupation or land use, occupy 22% of the Earth's ice-free surface [10] is sufficiently illustrative. Biodiversity loss consequences directly affect human populations because our lives depend on good ecosystem status [11,12]. The value of biodiversity and the need for its conservation are beyond doubt [13].

Although habitat loss is a global phenomenon, it has been especially important in certain areas of the planet that have undergone intenser anthropic occupation. In geographical areas like the Spanish coast in the western Mediterranean Basin, intensive agriculture together with the urbanisation of residential and industrial zones, have meant that the natural environment in very large areas has been totally or partially eliminated, or has been degraded or fragmented. In Spain, as in many other countries, legislation defines protected areas and land stewardship initiatives ([14]). Yet despite all the existing protection initiatives, the planet is far from achieving the conservation of an area (estimated to be half of the planet) that could save a very large part of existing life forms [15].

In these circumstances, it is necessary to resort to all resources to mitigate loss of natural habitats, and this involves the proper management of all available urban spaces, such as landscaped areas, green walls

---

* Corresponding author.
E-mail address: vibedu@hma.upv.es (V. Benedito Durà).

https://doi.org/10.1016/j.buildenv.2022.109865
Received 30 June 2022; Received in revised form 22 November 2022; Accepted 23 November 2022

and green roofs, which can be valuable for the conservation of all wildlife types [16,17]. Of these urban habitats, green roofs are especially innovative structures, and are one specific solution in sustainable urban drainage systems (SUDS), utilizing a range of technologies and techniques more sustainable than conventional solutions. This is based on the philosophy of replicating as closely as possible the natural predevelopment drainage of an urban environment. Although green roofs were initially based on energy saving, aesthetic aspects, runoff regulation, water quality management and control of heat island effects [18–23], it has subsequently been recognised as a suitable habitat for many fauna types, especially arthropods [24–29]. However, few studies have investigated the contribution of green roofs to arthropod biodiversity compared to that of unvegetated roofs [30,31].

Several studies have characterised the fauna found in these infrastructures and its abundance, the factors that condition it, the relations and dependence that it establishes with conventional landscaped areas, and how they complement them, as well as relations with surrounding areas [30,32–46]. Thus green roofs may be of far greater ecological importance than initially imagined, because they could act as complementary habitats to support biodiversity in urban areas, and in such a way that they could constitute reserves of diverse fauna types in intensely anthropic landscapes.

Studying these functionalities would be particularly interesting in some countries of the western Mediterranea Basin, and specifically on the Spanish coast. There are two circumstances of special interest in this region: on the one hand, climate conditions are quite stressful during part of the year, which will be exacerbated by climate change; on the other hand, the natural habitat in this territory has undergone intense loss and fragmentation.

There are no studies on the biodiversity of the fauna associated with green roofs in Spain, nor in other Mediterranean countries, except France, but this country is further north and its climate conditions are not as demanding. Perhaps this is because the development of SUDS in Mediterranean countries, specifically in Spain, is recent. Nonetheless, SUDS have found their way in urban drainage because many barriers have been overcome, especially in technical terms [47].

The aim of this work is to obtain a first characterisation of arthropods biodiversity (specific richness, abundance, functional groups) on a green roof and to compare it to an adjacent conventional roof and two surrounding ground-level gardens in a semirural village in the Valencian Community (Spain). The objective of the research is to check under these climatic conditions if green roof could complement the arthropod fauna of ground-level gardens and to what extent they could constitute a habitat that improves the conditions of conventional roof to host biodiversity.

2. Materials and methods

2.1. Study area

Fieldwork was carried out in the town of Benaguasil (Valencia, Spain) (Fig. 1) in spring and summer 2018 in two urban ground-level gardens (GLG1 and GLG2), and also on the roof of a Town Hall building employed for social services. The building is 10 m high with a flat roof covering 950 m^2, of which one part consists of an extensive green roof (GR) covering 241 m^2, and the other part is a conventional roof (CR) of 632 m^2, with a layer of non-structural concrete and a waterproofing membrane covered with a 10 cm gravel layer. From north-east to south-east the roof is oriented facing agricultural fields from which it is separated by 20 m. In other orientations it is surrounded by buildings. The GR has a substratum depth of 15 cm and was planted with four species of *Sedum* L., 1753 *non.* Adans., 1763, (*S. sediforme* (Jacq) Pau, *S. acre* L., *S. album* L., and *S. spurium* M. Bieb.) in the same proportions (Fig. 2). A detailed description of the GR's hydrological performance can be found in Ref. [48].

GLG1 consists in a sunny south-oriented garden with vegetation consisting mainly of *Spartium junceum* L., *and Lavandula* sp.*pl.*, and with trees of the species *Salix babylonica* L.. GLG2 is a shadowed north-oriented garden with *Salix babylonica and Ceratonia siliqua* L., whose soil has a dense cover of *Hedera helix* L. (Fig. 2). From the hydraulic perspective, GLG1 is an infiltration basin whose main function is to reduce runoff in an industrial area with public space availability. GLG2 is a detention-infiltration set of interconnected basins used to detain sediment and reduce runoff peaks. Both sites perform well from the hydrological perspective and help to cushion negative effects downstream [49].

The territory in this region consists of a semirural intensive agricultural area, with orange tree orchards and vegetable crops surrounding the town. We define it as semirural because of its high-density urbanisation. The studied habitats (GR, CR, GLG1 and GLG2) had similar connectivity with neighbouring vegetated habitats in terms of the possible transfer of organisms from these environments, because are on

Fig. 1. Images of the geographical location of the city of Benaguasil (Valencia, Spain) and the study sites in 2018. GR + CR (green roof and conventional flat roof), GLG1 and GLG2 (ground level-gardens 1 and 2 respectively). (GoogleEarth, Image Landsat/Copernicus, 2022). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

Fig. 2. Images of the studied sites in Benaguasil (Valencia, Spain) in 2018. (A) Building with the green roof and conventional roof (red dots represents trap location). (B) *Sedum* sp. on the green roof. (C, D) Ground-level gardens GLG1 and GLG2 respectively (red circle marks the sticky trap). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

the edge of the city, a few metres from the crops and with no obstacles or barriers separating them. The distances between the three studied habitats (GLG1, GLG2 and the studied roof) were 1.2–1.3 km (Fig. 1).

Benaguasil has a typical Mediterranean climate, with minimum temperatures in winter that drop slightly below 0 °C and maximum temperatures in august of up to 35 °C. The average annual rainfall is quite low, with 13 mm during the rainy season and 6 mm during the driest season (June–August). A good climatic characterisation of Benaguasil can be found at Weather Spark [50] .

2.2. *Insect sampling and processing*

Five surveys were carried out on 10 May, 6 June, 26 June, 18 July and 26 August at all the sampling sites (GLG1, GLG2, GR, CR). They consisted in locating two yellow sticky traps at each site and date, which were replaced on the indicated dates. The traps were always placed in the same point at each sampling site. On the roofs and ground level garden 2 these were located at 30 cm above the ground (Fig. 2 D), and in the ground level garden 1, hanging on the shrub vegetation at approximately 50 cm in height (Fig. 2 B). The location of the traps on the green and conventional roof is showed in Fig. 2 A. At the laboratory, the specimens in traps were identified with the help of a stereomicroscope (Leica MZ16) at the lowest possible taxonomic level. The specific taxonomic keys were consulted. The taxonomic adscription of fauna was done according to Refs. [51,52]. The obtained taxa were assigned to the functional groups, phytophagous, parasitoids, predators and pollinators. A group called "Other" was created to include Psocoptera, Diptera and Hymenoptera (Formicidae).

2.3. *Data processing and diversity metrics*

As taxonomic resolution was not the same for all the specimens, to assess biodiversity we worked with family level and some superfamilies. Number of families, the Shannon Entropy (H′) [53,54], Effective Number of families (from Exp H'), Gini-Simpson Index and Effective Number of Families (from 1/1-Gini-Simpson index) [55,56] were calculated. In this way, we contemplate the circumstances of considering more scarce and less frequent families (zero-order diversity), all families as equally likely (order 1 diversity) and considering more abundant and frequent families (order 2 diversity), according with [55].

The representativeness of the sampling procedure [57] was tested with the species richness estimators, Chao 1 [58,59], Jackknife 1 [60–63], ACE [64,65] and Bootstrap [63], calculated by means of the open-access EstimateS 9 software application [66]. Species dominance at each sampling site was calculated by the Biologic Value Index (BVI) [67] according to the methodology proposed by Ref. [68].

To statistically process data, ANOVA, MANOVA and canonical discriminant analyses were carried out using the SPSS statistical package [69]. The similarities and clustering of samples were calculated by the Bray-Curtis test [70] with the PRIMER-E v.6 software [71]. The data were transformed to fit the normality prior to analysis.

3. Results

3.1. *General sampling results*

In tis study 99 taxa were identified corresponding to 5 orders, 1 suborder, 8 superfamilies, 24 families, 1 subfamily, 35 genera and 24 species. The largest total number of captured taxa was obtained in GLG1, and the values for GLG2 and GR were similar (Table 1). This number was considerably lower on CR.

The mean abundance per trap of the captured taxa is shown in

Table 1
Number of taxa captured throughout the study of the different functional groups, and ratios of beneficial taxa to total and phytophagous taxa.

	GLG1	GLG2	CR	GR
Phytophagous taxa	13,00	10,00	9,00	8,00
Predator taxa	12,00	12,00	8,00	12,00
Parasitoid taxa	50,00	47,00	36,00	45,00
Pollinator taxa	5,00	5,00	5,00	5,00
Other	3,00	3,00	3,00	3,00
Beneficial taxa/total taxa	0,81	0,83	0,81	0,85
Beneficial taxa/Phytofagous taxa	5,15	6,40	5,44	7,75

Supplementary Table 1. Significant differences (F = 3.24, $p = 0.0344$) were found among the mean number of taxa per trap captured at GLG1 (34.6 ± 2.6), GLG2 (33.4 ± 2.6), GR (32.9 ± 2.5) and CR (24.4 ± 2.6). On CR, values were significantly lower than at the other sites ($p < 0.05$). In spring, the traps on GR captured more taxa than in the GLGs, with the opposite pattern shown in summer (Fig. 3).

The total taxa richness corresponding to the different functional groups was 56 for parasitoids, 16 for predators, 16 for phytophagous, 8 for pollinators and 3 taxa included in other group. The two ratios, beneficial taxa/total taxa, and beneficial taxa/phytophagous taxa, were higher on GR. For beneficial arthropods, GLGs and GR registered the same number of predator and pollinator taxa, and in the first habitat, the number of parasitoid taxa was slightly larger (Table 1).

Exclusive taxa were captured in all sampled habitats (Table 2), being more numerous in GLGs than in GR. All the exclusive taxa on GR were beneficial insects. It should be emphasised that some taxa were caught only in GLG1 and on GR, but none was common only to GLG2 and GR. Sixteen taxa of those captured on GR, were not found on CR. On CR, only one taxon was exclusive, not captured in the other habitats (Supplementary Table 1).

3.2. *Abundance and occurrence of functional groups*

According to the mean relative abundance of arthropods per trap (Fig. 4), a different pattern was observed depending on season and sampling sites. Phytophagous showed the highest relative abundance in all the habitats in spring, and also on GR and CR in summer. Diptera (other group) were the most abundant in the two GLGs also in summer. Parasitoids were always the second most abundant group. Pollinators were always the least abundant, and were slightly more abundant during both seasons in GLGs than on GR. The relative abundance of predators on GR was always higher than in GLGs. On CR, the pattern observed for all the functional groups followed that of GR.

3.3. *Biologic value index (BVI)*

This index combines the temporal constancy and abundance of taxa to describe community structure. Table 3 shows the values obtained for this index at each sampling site for the 25 more dominant taxa, which represented between 95 and 100% of the captured arthropods and constituted 25% of all the taxa obtained in this study. Assuming that the differences in taxonomic categories that we have been able to achieve could change the order of importance of the taxa at each site, the basic objective of this section is to compare the results between the different sites studied. The criteria for classifying specimens have been the same for all habitats and each specimen has only been recorded in one taxonomic category.

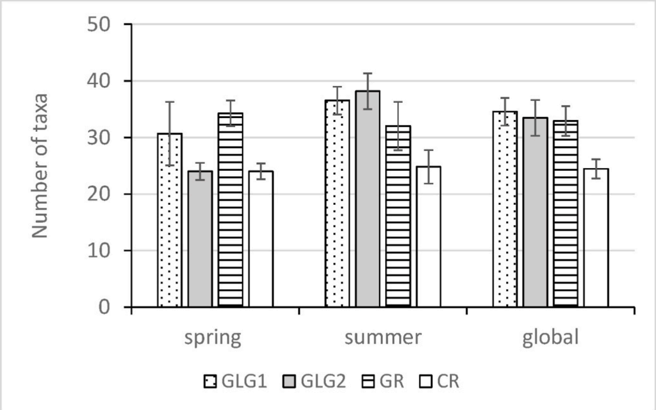

Fig. 3. Mean number of taxa per trap at sampling sites and season in 2018 in Benaguasil (Valencia, Spain). GLG1 and GLG2 (ground-level gardens), green roof (GR) and conventional roof (CR).

Table 2
Number of taxa of functional groups caught exclusively at a sampling site, and taxa in common only present in the indicated relationships.

	Phytophagous	Predators	Parasitoids	Pollinators
GLG1	3	1	3	0
GLG2	2	3	2	1
GR	0	1	3	1
CR	0	0	0	1
GLG1- GLG2	1	0	5	1
GLG1-GR	1	1	2	0
GLG1-CR	1	0	0	0
GLG2-GR	0	0	0	0
GLG2-CR	1	0	0	0
GR-CR	0	0	0	0

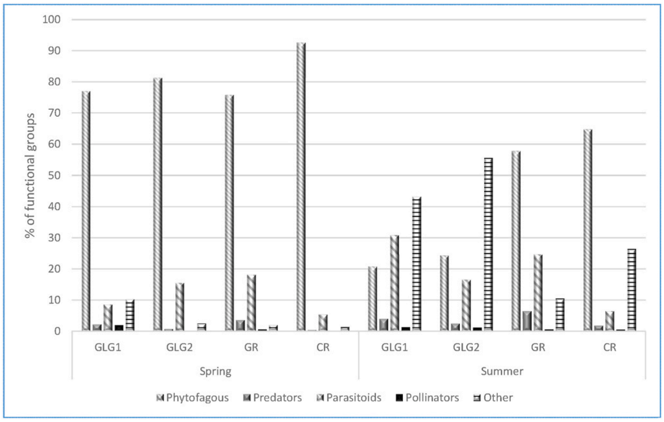

Fig. 4. Relative abundance of funcional groups depending on season and sampling sites in 2018 in Benaguasil (Valencia, Spain).

The values obtained in GLG2 were higher, with a value above 100 for practically all 25 taxa. This means that taxa abundance remained over time. The values for GLG1 and GR were lower than in the previous habitat, and 12 and 10 taxa were, respectively, above the value 100. At both the GLGs and GR sites, the dominance of Diptera, Thysanoptera and Aphididae coincided, and Ceraphronoidea was also dominant. At GLG2, four more taxa were fairly dominant: Cicadellidae, Ichneumonoidae, Cynipoidea and Metaphycus. Of the 25 most frequent and abundant taxa on GR, four (Anagyrus sp, Mymaridae, Polynema and Encarsia) were not found in both GLGs, *Encarsia* and Coccidae were not found at GLG1 and *Encarsia*, *Trichogramma*, *Polinema*, *Pollistes gallicus* and *Syrphophagus,* did not appear at GLG2. However, only Formicoidea and Aphytis were observed in GLGs, but not on GR.

3.4. *Composition of the arthropod community on roofs and ground-level gardens*

To establish the existence of differences between the habitats studied, a MANOVA test was carried out. The data used were the abundances of all the taxa in the traps at each site. The taxonomic resolution is as shown in Supplementary Table 1. The MANOVA analysis showed significant differences in the faunal composition of the sampling sites at level F Pilai (9, 99) = 1.98, $p < 0.1$.

When all the captured taxa were ordered according to relative abundance and total abundance, the first 36 taxa coincided with both criteria with few variations in the ordering (Supplementary Table 2). In all the studied habitats, these 36 taxa accounted for almost 100% of the captured arthropods. By performing the MANOVA analysis with these 36 taxa, significance increased (F Pilai 15, 93 = 2.807, $p < 0.05$). A discriminant analysis confirmed that the communities at the GLGs, GR and CR sites were significantly different ($p < 0.005$) (Fig. 5), and

Table 3
Values obtained for the Biological value index (BVI) at each sampling site, GLG1, GLG2, GR and CR.(Ph: phytofagous; Pa: parasitoids; Pre: predators; Po: pollinators; O: other).

Functional group	Taxa	IVB GLG1	Functional group	Taxa	IVB GLG2
O	Diptera	242	Ph	*Frankinella occidentalis*	304
Ph	*Frankinella occidentalis*	225	Ph	*Aphis* sp.	298
Ph	*Aphis* sp.	225	Ph	Cicadellidae	295
Pa	Ceraphronoidea	186	O	Diptera	276
Pa	Platygastroidea	146	Pa	Ichneumonidae	251
Pa	*Aphelinus*	130	Pa	Ceraphronoidea	203
Pa	*Metaphycus*	127	Pa	Cynipoidea	203
Po	*Polistes gallicus*	124	Pa	*Metaphycus*	204
Ph	Cicadellidae	120	Pa	Platygastroidea	181
Pa	*Syrphophagus*	111	Pr	*Scymnus* sp.	178
Pa	Braconidae	100	Pa	Braconidae	166
Pre	Araneae	110	O	Psocoptera	163
Pre	*Scymnus* sp.	97	Pa	*Gonatocerus*	152
Pa	*Idris*	96	Pa	*Aphytis*	151
O	Formicidae	86	Pa	Eulophidae	134
Pa	Cynipoidea	85	Ph	Coccidae	131
Pa	*Ceranisus menes*	84	Pa	*Stethynium triclavatum*	129
Pa	*Trissolcus*	84	Ph	Aleyrodidae	129
Pa	*Trichogramma*	81	Pr	Araneae	128
Pa	*Gonatocerus*	81	Pa	*Ceranisus menes*	126
Pa	*Alaptus*	75	Pr	Staphyllindae	126
Pa	Encyrtidae	75	Pa	*Vespa germanica*	99
Pa	*Aphytis*	72	O	Formicidae	115
Pa	Eulophidae	72	Ph	Miridae	101
Pa	Aphiinidae	68	Po	Aphiinidae	86

Functional group	Taxa	IVB GR	Functional group	Taxa	IVB CR
Ph	*Frankinella occidentalis*	225	Ph	*Aphis* sp.	129
O	Diptera	201	Ph	*Frankinella occidentalis*	116
Ph	*Aphis* sp.	191	O	Diptera	107
Pa	Ceraphronoidea	127	Ph	Cicadellidae	*73*
Pr	*Scymnus* sp.	127	Pa	Ceraphronoidea	73
Pa	*Aphelinus*	125	Pa	*Ceranisus menes*	68
Pa	Encyrtidae	118	Pa	*Aphelinus*	64
Pa	Anagyrus sp.	117	Pa	Aphiinidae	59
Ph	Cicadellidae	106	Pa	Platygastroidea	56
Pa	Platygastroidea	100	Pa	Encarsia	46
Pa	Aphiinidae	99	Pr	*Scymnus* sp.	46
Pa	Cynipoidea	92	Pa	Trichogramma	41
Ph	Miridae	86	Pr	Staphyllindae	36
Pa	*Gonatocerus*	82	Pa	Encyrtidae	35
Pa	Mymaridae	79	Ph	Coccidae	34
Pa	Syrphophagus	77	Pa	Eulophidae	31
Pa	*Metaphycus*	74	Pa	*Anagrus atomus*	30
Po	*Polistes gallicus*	62	Pa	*Polinema*	27
Pa	Braconidae	56	Pa	*Megastigmus*	27
Pa	Polinema	55	Pa	*Syrphophagus*	27
Ph	Coccidae	52	Pa	Agaonidae	26
Pa	Trichogramma	51	Pa	Cynipoidea	26
Pr	Araneae	51	Pa	*Baryscapus*	24
Pa	Encarsia	39	Pr	Cecidomyiidae	22
Pa	Eulophidae	38	Po	*Polistes gallicus*	22

Supplementary Table 3 provides the most influential species for the differences between sampling sites. A Bray-Curtis similarity analysis (Supplementary Fig. 1) reported that the samples from roofs (GR and CR), and GLGs showed faunal composition, in which the effect of habitat and seasonality was noted. The samples belonging to the same habitat type had more similarities.

3.5. Taxa richness

In order to obtain meaningful results when comparing the biodiversity of the studied habitats, calculations were carried out using the family category. Zero order, order one, and order two diversity, were considered in order to cover the three possibilities: more scarce and less frequent species (number of families); all species as equally likely (Shannon entropy); and more abundant and frequent species (Gini-Simpson index), according with [55]. These metrics and the mean number of individuals per trap are shown in Fig. 6.

No significant differences were found for the number of individuals captured per trap between sites nor between number of families ($F = 1,21$, $p = 0,3228$and $F = 2,45$, $p = 0,0813$ respectively). The true diversity values (effective number of families), according Gini-Simpson and Shannon indexes were significantly lower in CR than in GLG1, GLG2 and GR ($F = 5,72$, $p = 0,0029$ and $F = 7,04$, $p = 0,0009$ respectively), and for both metrics the values in GR were also significantly lower than in GLG2.

Sampling representativeness (probable number of existing families) was over 85% for all the estimators in all the habitats. Thus, the sampling effort provided representative results in this study at the level of families (Table 4).

4. Discussion

This study, as part of a larger research project, evaluated the diversity of flying arthropod fauna (using sticky traps) on a green roof, and

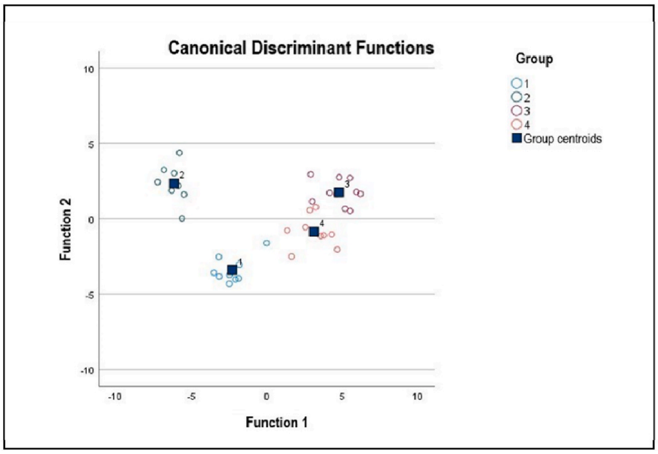

Fig. 5. Results of discriminant analyses according arthropod composition of the sampling sites in 2018 in Benaguasil (Valencia, Spain). (Samplig sites: 1:GLG1; 2: GLG2; 3: GR; 4: CR).

assessed the contribution of this type of habitat to urban biodiversity by comparing its population to those of a conventional flat roof and two ground-level gardens. We found that no study of this type has been carried out in countries of the Mediterranean Basin, except in France, where climate conditions differ from those in warmer southern countries like Spain. Our results are the first to be reported in our geographical area, which represents climate circumstances for green roofs that may condition the arthropod communities that they host and their dynamics. In the taxonomic characterisation, it was not always possible to reach the species category. For that reason biodiversity metrics were applied at the taxonomic family level. It has been suggested that higher taxonomic resolution levels can be representative of patterns at a lower level in terrestrial invertebrate communities [72].

In our study, fewer arthropod families per trap were captured on the conventional non-vegetated roof, and their effective number also was significantly lower than in GLGs and on GR. This is consistent with [30]; who reported significantly greater abundance and diversity of arthropods and other invertebrates on green roofs than on conventional roofs, and although no significant differences were found in community composition, the dominant taxa varied between roof types [73]. also reported more abundance of arthropods in green roof than in a bare roof. All the findings reflect conventional flat roof structural simplicity as well as lack of vegetation and foraging resources for arthropods.

The studied sites (GLG1, GLG2, GR and CR) had similar connectivity with the neighbouring vegetated areas because they were located in the

Table 4
Diversity methrics and sampling representativeness estimators calculated for each study site.

	Hábitat			
	GLG1	GLG2	GR	CR
Number of families	43	43	37	35
ACE	50,10	48,20	39,10	36,75
ACE SD	0	0	0	0
Chao 1	46,59	47,99	37,74	35,49
Chao 1 SD	3,85	5,54	1,42	1,02
Jackknife 1	50,11	50,11	41,5	39,44
Jackknife 1 SD	2,81	3,38	1,5	1,93
Bootstrap	46,54	46,32	39,12	37,23
Botstrap SD	0	0	0	0
Eficience ACE (%)	85,82	89,21	94,62	95,23
Eficience Chao 1 (%)	92,29	89,60	98,82	98,61
Eficience Jacknife 1 (%)	85,81	85,81	89,15	88,74
Eficience Bootstrap (%)	92,39	92,83	94,58	94,01

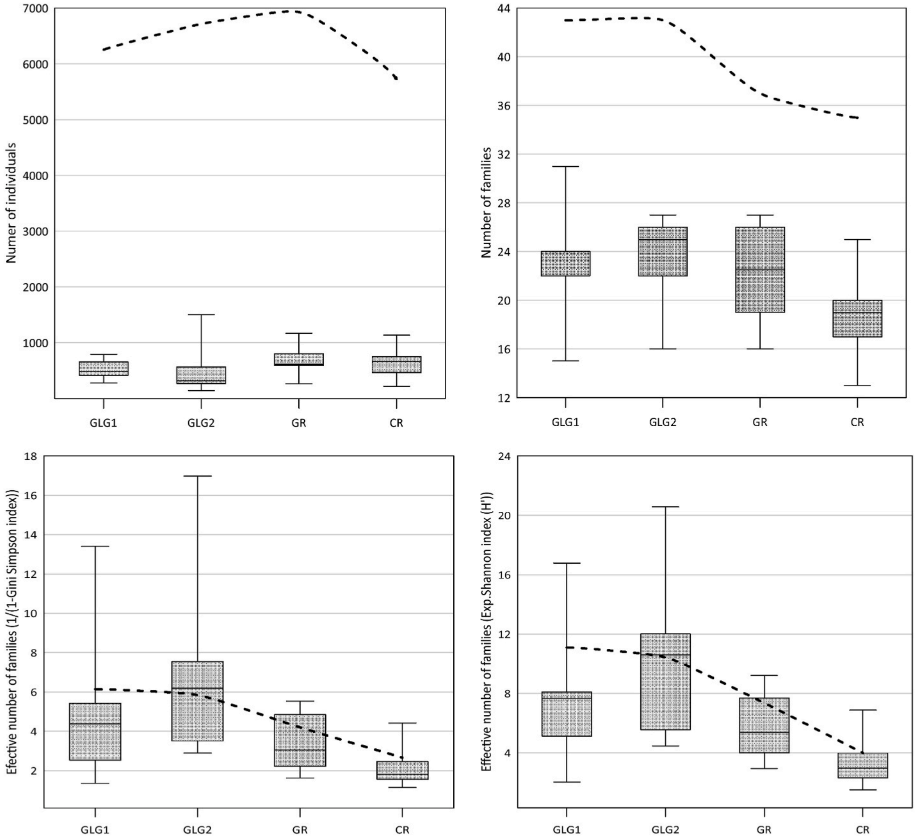

Fig. 6. Biodiversity metrics obtained at the sampling sites, GLG1 and GLG2 (urban ground-level gardens), GR (green roof) and CR (conventional roof). The dotted line represents the values calculated with all samples from each habitat. Box-and-whisker plots contain the values of all samples in each habitat.

outer part of the urban area. Consequently the surrounding vegetation equally influenced all the sites we studied in Benaguacil. It is generally agreed, that the existence of vegetated habitats in the vicinity of green roofs and their size and composition, can positively influence overall arthropod species richness or that of specific taxa [36,38,41–43,45, 74–81]. Thus, the taxa we captured in CR would correspond to the "background" arthropod fauna on the roofs, which come almost entirely from the surrounding areas, and in this case especially from the GR.

Regarding the comparison of arthropod communities on green roofs with those on ground level habitats, our results match the general patterns of higher abundance of individuals and taxa richness generally reported at the ground level [32,33,35,39,42,43,82]. Although these metrics did not showed significant differences between GR and GLGs, the effective number of families we reported in GR was significantly lower than in GLG2 and also lower than in GLG1, supporting the above model.

Patterns of composition, distribution and abundance of arthropod fauna on green roofs have been related to some ecological factors influencing terrestrial ecosystems in general. One of them is the habitat area, which in our study sites was smaller on GR than in GLGs. Species richness and abundance generally increased with habitat area [83]. This has also been demonstrated for urban green spaces and green roofs [43], where a larger surface area favours a greater diversity of the plant community [35,84–86]. The area has been also related to the structural complexity hypothesis [87], which has been used to explain how more structurally complex vegetation favours richness for some taxa. Another influencing factor could be the altitude. [36]; pointed out that altitude is considered a determinant of colonisation, and which taxa may exist, and [42] stated that higher green roofs have been associated with less total invertebrate abundance and dipteran family richness due to lower vertical connectivity. In spite of the altitude and smaller extension, the results we have obtained for arthropod abundance and diversity on GR are not so different from those recorded in GLGs, being very similar to those of GLG1. Therefore, we did not find the area nor the height influenced our results in the way that previous authors suggested. Specifically with respect to height [33], reported that a wide variety of insects (not necessarily the most mobile), can colonise green roofs. It could be that other factors were influencing GR to compensate for the effect of height or smaller area.

The stressful dryness conditions could have been another cause to consider influenced invertebrate richness and abundance. GR was no longer irrigated and managed, and dryness was important in summer. Dryness in GLG1 was also intense because this garden was not conveniently irrigated and received considerable insolation. GLG2 was the habitat that maintained the most constant and good humidity conditions because of its orientation and tree coverage. It has been suggested that green roofs are associated with stressful dryness conditions in relation to thin substrate, which conditions vegetation, and the environment becomes more inhospitable for fauna than in ground-level gardens [29,88, 89]. This would be consistent with the values we obtained for the BVI, that reflected GLG2 was a more equilibrated habitat in terms of a greater constancy for higher abundance taxa, than GLG1 and GR, that were more stressed by dryness.

Concerning functional groups, our results showed a general pattern that in both, GLGs and GR, parasitoids were the most diverse group, but phytophagous were the most abundant, followed by parasitoids. For the relative abundances, the general pattern showed phytofagous and predators dominated in GR and pollinators and parasitoids in GLGs. These results agree with those in the literature [43,45], in that phytophagous and parasitoids were the most diverse and abundant groups in both habitats (green roofs and ground level). While there is consensus that parasitoids dominate on green roofs, in some studies phytophagous, have been cited as dominating at ground level or in green roofs. For relative abundances there is agreement that predators are more abundant on green roofs. In some cases, no differences in functional diversity between ground level and greenroofs, have also been reported for some arthropod taxa [39]. Regarding specific groups [33], found a higher abundance for leafhoppers (Cicadellidae) on green roofs than at ground level, as did [27,44]; contrary to our results. This discrepancy could be due to the different vegetation types.

We observed that all functional groups presented variable relative abundances in GLGs and GR depending on the season (spring or summer). This suggests that the variations observed in different studies could be related to seasonality. Seasonality directly affects the vegetation as it conditions its growth cycle and the availability of plant or preys/hosts, and this could influence the presence of the great majority of functional groups but perhaps not all. The total and relative abundance of diptera was always higher in GLGs than in GR, which is consistent with [42]; who also reported significant differences in the composition of the diptera families between the two habitats. From their results, it appears that most dipterans find most favourable habitat conditions at ground level and that only some groups such as chironomids find suitable circumstances on green roofs. We have not been able to verify this.

For green roofs, it has been indicated that habitat area would increase the richness and abundance of phytophagous, parasitoids and predators, and reference has also been made to “natural enemies” hypothesis, according to which higher plant diversity (related with the area), favours predators and parasitoids because the offer of feeding resources and shelters is bigger [45,90,91]. . Other authors have reported higher bee diversity to be positively related to plant diversity on green roofs [35,84–86]. In our view, logic suggests that all of the above will apply equally to habitats at ground level. On this point, we obtained the same number of pollinator taxa in all three habitats despite the fact that in GLG2 flower availability was scarce, and the relative abundance of this functional group was clearly higher only in GLG1, while GlG2 and GR presented similar values. Concerning the other functional groups (phytofagous, predators, and parasitoids), GR only showed a lower taxa number for the phytofagous, while the other groups showed a similar richness to that found in GLGs. Thus, the interpretation of the effect of habitat size on our results, taking into account functional groups, does nothing to clarify the effect of area discussed above. There must be factors whose effect we have not assessed which compensated for the effect of the smaller GR area. This aspect needs to be tested in the future.

One of the most important results of this study is that significant differences existed between the arthropod communities of the GR and GLGs. These differences were due to two phytophagous taxa (Heteroptera, and Thysanoptera), and a larger group of beneficials (the parasitoids *Aphelinus*, Encyrtidae, *Gonatocerus,* the predators *Scymnus* and Araneae, and the pollinator *Pollistes gallicus*). The higher beneficial taxa/ phytophagous taxa ratio found in the GR was due to the lower number of phytophagous taxa in the GR, as the number of pollinators, predators and parasotoids taxa was similar in all three habitats. The analysis of differences in the faunal composition of the sampling sites were consistent with the results we obtained in the calculation of the Biological Value Index, and the taxa in common, which showed more similarities in community composition and abundance between GR with GLG1 than with GLG2. These similarities between GR and GLG1 could be due to similar environmental conditions and to more structurally similar vegetation type.

In addition to the differences between the arthropod communities of Gr and GLGs, another factor to be taken into account is the presence of exclusive taxa in these habitats. Five exclusive taxa recorded in the GR, all of them beneficials (one pollinator, three parasitoids and one predator) [33]. recorded 65 species not present at the ground level on green roofs that also included phytophagous. [30,31]; pointed out that green roofs extends the areas available for the many arthropod populations present in ground-level habitats, also allowing the presence of species not found in them. It can thus be concluded that the existence of taxa exclusive to green roofs is evidence that this habitat adds to the biodiversity found in urban habitats at ground level.

Although more confirmatory data are needed, green roofs may have a better number of beneficial species/number of phytophagous species ratios than ground-level urban habitats. The latter advantage might be important in two aspects: one is that it could provide support in pest control in urban vegetated areas, while in periurban agricultural environments where these stressful territories are affected by frequent pesticide and herbicide treatments, it could act as a refuge for fauna. The vegetated urban habitats in our study area might introduce more spatial heterogeneity than neighbouring agricultural areas, and also better environmental quality [92]. indicated that land-use heterogeneity favours diverse arthropod fauna in urban habitats. The opposite could actually be happening in our study area due to the intense anthropic pressure on the territory (occupation, monocultures and chemical treatments), so this is an aspect, that should be taken into account in future studies in our geographical.

It has been pointed out however that the creation of new urban habitats may also involve functional drawbacks as to the way it can affect population dynamics and community structure, and organisms, given the alteration to their behaviour, selection processes, and the population's genetic structure [93]. There are also records that urban habitats are global homogenisers of biodiversity and that actions to be performed must aim to restore the diversity of native species to not lose regional biotic singularity [94]. In our study area, apart from a harsh climate during part of the year, natural habitat loss is intense and widespread, and the use of biocides is substantial. So even with the considerations made by previous authors, and under the stressful conditions described for our region, green roofs and ground-level gardens could have a clearly positive effect. This subject needs to be more studied.

In this study only yellow sticky traps were used, however, a combination of several methods would have better determined how many species may actively use a habitat [43], and ensure a complete community characterisation (McIvor and Lumdholm, 2011). In future studies, we will use a combination of invertebrate trapping systems to obtain as most of the fauna present on green roofs as possible, in order to assess its true potential to host biodiversity.

Climate change projections indicate an increase in insect populations and the number of annual generations, their geographic expansion, overwinter survival, changes in their relation with host plants and natural enemies, increases in invasive species, and pest intensifications [95], with diverse effects on the natural enemies of pest species [96]. There is growing concern about a response involving the instense use of insecticides, which would lead to beneficial arthropod diversity loss [97]. In this context, the role that green roofs could play as arthropod refuges, especially for beneficial fauna species, could be very important. This may be more relevant in semirural areas, like those areas with substantial urban development in territories with intense agricultural activity, such as Mediterranean parts of Spain and especially the Valencian Community.

Our data constitute the first approach to study green roof arthropod populations in southern European countries, where climatic conditions are warmer and more stressful throughout the year. Many more studies will be necessary to acquire good knowledge of the communities that colonise these habitats and their ecological functioning under these environmental conditions. The herein presented results provide us with a message of hope about green roofs being useful in this Mediterranean region for enhancing urban biodiversity.

5. Conclusions

This is the first study in the Mediterranean basin focused on arthropod biodiversity on green roofs. Given the results, it has been demonstrated that green roofs are a suitable habitat for arthropod fauna in Mediterranean climate conditions. Under these circumstances, this type of habitat enhances the biodiversity of arthropods in urban areas compared to conventional roofs, and complements the biodiversity of ground-level gardens. Although more studies are needed, our results and those of other authors suggest that there is a basis to consider that green roofs favour beneficial arthropod populations. In this work we have realized that in order to adequately assess the contribution of green roofs to urban biodiversity a combination of sampling methods must be used. It is also important to dedicate efforts to study the importance of green roofs in territories with an intense loss of natural habitat and with stressful conditions like intensive agriculture, as seen in our case. In these circumstances, green roofs may act as refuges for wildlife.

CRediT authorship contribution statement

Vicent Benedito Durà: Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Software, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Enric Meseguer:** Investigation, Data curation. **Carmen Hernández Crespo:** Software. **Miguel Martín Monerris:** Validation, Software. **Ignacio Andrés Doménech:** Validation, Resources, Project administration, Funding acquisition. **M. Eugenia Rodrigo Santamalia:** Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Methodology, Investigation, Formal analysis, Data curation, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

Acknowledgments

This research has been developed within the ENGODRAIN project framework (Ref. RTI2018-094217-B-C31) funded by Ministerio de Ciencia e Innovación (MICIN)/Agencia Estatal de Investigación (AEI), Spain, through the General State Budgets and the European Regional Development Fund (ERDF).

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.buildenv.2022.109865.

References

[1] P.A. Delcourt, H.R. Delcourt, Prehistoric Native Americans and Ecological Change: Human Ecosystems in Eastern North America since the Pleistocene, Cambridge university Press, 2004, p. 204.

[2] S. Faurby, J.C. Svenning, Historic and prehistoric human-driven extinctions have reshaped global mammal diversity patterns, Divers. Distrib. 21 (2015) 1155–1166.

[3] E.O. Wilson, The Diversity of Life, Belknap press, Cambridge MA, 1992, p. 424.

[4] Convention on Biological Diversity, Global Biodiversity Outlook 5, 2020 (Montreal).

[5] A. Wood, P. Stedman-Edwards, J. Mang (Eds.), The Root Causes of Biodiversity Loss, Taylor & Francis, 2000, p. 416.

[6] T.M. Brooks, R.A. Mittermeier, C.G. Mittermeier, G.A.B. da Fonseca, A.B. Rylands, W.R. Konstant, P. Flick, J. Pilgrim, S. Oldfield, G. Magin, C. Hilton-Taylor, Habitat loss and extinction in the hotspots of biodiversity, Conserv. Biol. 16 (2002) 909–923.

[7] I. Hanski, The Shrinking World: Ecological Consequences of Habitat Loss, International Ecology Institute, Germany, 2005, p. 307. Oldendorf/Luhe.

[8] M.J. Groom, G.K. Meffe, C.R. Carroll, Principles of Conservation Biology, Sinauer, Sunderland, 2006.

[9] V. Singh, S. Shukla, A. Singh, The principal factors responsible for biodiversity loss, Open J. Polit. Sci. 6 (1) (2021), https://doi.org/10.17352/ojps.000026, 011-014.

[10] E.C. Ellis, N. Ramankutty, Putting people in the map: anthropogenic biomes of the world, Front. Ecol. Environ. 6 (8) (2008) 439–447, https://doi.org/10.1890/070062.

[11] Millennium Ecosystem Assessment, Ecosystems and Human Well-Being: Synthesis Repport, Island Press, Washington, 2005.

[12] E. Chivian, A. Bernstein, How Our Health Depends on Biodiversity, 2010. https://www.bu.edu/sph/files/2012/12/Chivian_and_Bernstein_2010_How_our_Health_Depends_on_Biodiversity.pdf.
[13] R.E. Leakey, R. Lewin, The Sixth Extinction: Patterns of Life and the Future of Humankind, Anchor Books, 1996, p. 271.
[14] España LBPN, 42/2007., Ley de 13 de diciembre, de Patrimonio Natural y de la Biodiversidad. Boletín Oficial del Estado, 14 de diciembre de núm. (2007) 299.
[15] E.O. Wilson, Half-Earth: Our Planet's Fight for Life, Liveright Publ, 2016.
[16] N.S. Williams, J. Lundholm, J. MacIvor, Do green roofs help urban biodiversity conservation? J. Appl. Ecol. 51 (2014) 1643–1649, https://doi.org/10.1111/1365-2664.12333.
[17] C.D. Ives, P.E. Lentini, C.G. Threlfall, K. Ikin, D.F. Shanahan, G.E. Garrard, R. Rowe, Cities are hotspots for threatened species, Global Ecol. Biogeogr. 25 (2016) 117–126, https://doi.org/10.1111/geb.12404.
[18] H.F. Castleton, V. Stovin, S.B.M. Beck, J.B. Davison, Green roofs; building energy savings and the potential for retrofit, Energy Build. 42 (2010) 1582–1591, https://doi.org/10.1016/j.enbuild.2010.05.004.
[19] T. Susca, S.R. Gaffin, G.R. Dell'Osso, Positive effects of vegetation: urban heat island and green roofs, Environ. Pollut. 159 (2011) 2119–2126, https://doi.org/10.1016/j.envpol.2011.03.007.
[20] O. Saadatian, K. Sopian, E. Salleh, C.H. Lim, S. Riffat, E. Saadatian, A. Toudeshki, M.Y. Sulaiman, A review of energy aspects of greenroofs, Renew. Sustain. Energy Rev. 23 (2013) 155–168, https://doi.org/10.1016/j.rser.2013.02.022.
[21] UNEP, Green Infrastructure Guide for Water Management: Ecosystem-Based Management Approaches for Water-Related Infrastructure Projects, vol. 75, UNEP, 2014, ISBN 978-92-807-3404-1.
[22] B. Woods Ballard, S. Wilson, H. Udale-Clarke, S. Illman, T. Scott, R. Ashley, R. Kellagher, The SUDS Manual, CIRIA C753, London, UK, 2015. Available online: http://www.scotsnet.org.uk/documents/NRDG/CIRIA-report-C753-the-SuDS-manual-v6.pdf.
[23] K. Vijayaraghavan, Greenroofs: acritical review on the role of components, benefits, limitations and trends, Renew. Sustain. Energy Rev. 57 (2016) 740–752, https://doi.org/10.1016/j.rser.2015.12.119.
[24] W. Heinze, Results of an experiment on extensive growth of vegetation on roofs, Rasen Grünflachen Begrünungen 16 (3) (1985) 80–88.
[25] M. Köhler, Plant survival research and biodiversity: lessons from Europe, in: Paper Presented at the First Annual Greening Rooftops for Sustainable Communities Conference, 2003. Awards and Trade Show; 20–30 May 2003, Chicago.
[26] D. Gedge, G. Kadas, Bugs, bees, and spiders: green roof design for rare invertebrates, in: Paper Presented at the Second Annual Greening Rooftops for Sustainable Communities Conference, 2004. Awards and Trade Show; 2–4 June 2004, Portland, Oregon.
[27] R.R. Coffman, G. Davis, Insect and avian fauna presence on the Ford assembly plant ecoroof, in: Paper Presented at the Third Annual Greening Rooftops for Sustainable Communities Conference, 2005. Awards and Trade Show; 4–6 May 2005, Washington, DC.
[28] N. Baumann, Ground-nesting birds on green roofs in Switzerland: preliminary observations, Urban Habitats 4 (1) (2006) 37–50.
[29] E. Oberndorfer, J. Lundholm, B. Bass, R.R. Coffman, H. Doshi, N. Dunnett, S. Gaffin, M. Köhler, K.K.Y. Liu, B. Rowe, Green roofs as urban ecosystems: ecological structures, functions, and services, Bioscience 57 (2007) 823–833. Our world in data, https://ourworldindata.org/habitat-loss.
[30] K. Berthon, D. Nipperess, P. Davies, M. Bulbert, Confirmed at Last: Green Roofs Add Invertebrate Diversity. State of Australian Cities Conference, 2015. https://www.researchgate.net/publication/316282695_Confirmed_at_Last_Green_Roofs_add_Invertebrate_Diversity.
[31] E.I.F. Wooster, R. Fleck, F. Torpy, D. Ramp, P.J. Irga, Urban Green Roofs Promote Metropolitan Biodiversity: A Comparative Case Study, vol. 207, Building and environment, 2022, 108458, https://doi.org/10.1016/j.buildenv.2021.108458.
[32] S.R. Colla, E. Willis, L. Packer, Can green roofs provide habitat for urban bees (Hymenoptera: apidae)? Cities and the Environ. 2 (2009) 4.
[33] J.S. MacIvor, J. Lundholm, Insect species composition and diversity on intensive green roofs and adjacent level-ground habitats, Urban Ecosyst. 14 (2011) 225–241.
[34] B.Y. Schindler, A.B. Griffith, K.N. Jones, Factors influencing arthropod diversity on green roofs, Cities and the Environ. (CATE) 4 (1) (2011). Article 5, https://digitalcommons.lmu.edu/cate/vol4/iss1/5.
[35] R. Tonietto, J. Fant, J. Ascher, K. Ellis, D. Larkin, A comparison of bee communities of Chicago green roofs, parks and prairies, Landsc. Urban Plann. 103 (2011) 102–108, 2011.
[36] F. Madre, A. Vergnes, N. Machon, Ph Clergeau, A comparison of 3 types of green roofs as habitats for arthropods, Ecol. Eng. 57 (2013) 109–117, https://doi.org/10.1016/j.ecoleng.2013.04.029.
[37] A. Steck, S. Morgan, W. Retzlaff, J.B. Williams, Insect communities on green roofs that are close in proximity but vary in age and plant coverage, J. Living Architect. 2 (1) (2015) 1–11. Available at: http://greenroofs.org/resources/JOLA2015.2.1.Steck.pdf.
[38] L. Blank, A. Vasl, B.Y. Schindler, G.J. Kadas, L. Blaustein, Horizontal and vertical island biogeography of arthropods on green roofs: a review, Urban Ecosyst. 20 (2017) 1–7, https://doi.org/10.1007/s11252-016-0639-9.
[39] S. Braaker, M.K. Obrist, J. Ghazoul, M. Moretti, Habitat connectivity and local conditions shape taxonomic and functional diversity of arthropods on green roofs, J. Anim. Ecol. 86 (2017) 521–531, https://doi.org/10.1111/1365-2656.12648.
[40] M.M. Hofmann, S.S. Renner, Bee species recorded between 1992 and 2017 from green roofs in Asia, Europe, and North America, with key characteristics and open research questions, Apidologie 49 (2018) 307–313, https://doi.org/10.1007/s13592-017-0555-x, 2018.
[41] K. Ksiazek-Mikenas, J. Herrmann, S.B. Menke, M. Köhler, If you build it, will they come? Plant and Arthropod. Diver. Urban Green Roofs over Time', Urban Nat. Special Issue (2018) 52–72.
[42] J.R. Dromgold, C.G. Threlfall, B.A. Norton, N.S.G. Williams, Green roof and ground-level invertebrate communities are similar and are driven by building height and landscape context, J. Urban Econ. (2020) 1–9, 2020.
[43] M.V. Sánchez Domínguez, E. González, D. Fabián, A. Salvo, M.S. Fenoglio, Arthropod diversity and ecological processes on green roofs in a semi-rural area of Argentina: similarity to neighbor ground habitats and landscape effects, Lanscape and urban planning 199 (2020), 103816, https://doi.org/10.1016/j.landurbplan.2020.103816.
[44] C.F.M. den Biedman, M. Tanis, E.F. Drukker, S. de Waart, The leafhopper fauna of green roofs including the Mediterranean leafhopper Circulifer haematoceps new for The Netherlands (Auchenorrhyncha: Cicadellidae), Entomol. Ber. (Amst.) 81 (2) (2021) 46–51.
[45] D. Fabian, E. González, M.V. Sánchez, A. Salvo, M.S. Fenoglio, Towards the design of biodiverse green roofs in Argentina: assessing key elements for different functional groups of arthropods, Urban For. Urban Green. 61 (2021), https://doi.org/10.1016/j.ufug.2021.127107.
[46] S. Gonsalves, O. Starry, A. Szallies, S. Brenneisen, The effect of urban green roof design on beetle biodiversity, Urban Ecosyst. 25 (2022) 205–219, https://doi.org/10.1007/s11252-021-01145-z.
[47] I. Andrés-Doménech, J. Anta, S. Perales-Momparler, J. Rodriguez-Hernandez, Sustainable urban drainage systems in Spain: a diagnosis, Sustainability 13 (2021) 2791, https://doi.org/10.3390/su13052791.
[48] I. Andrés-Doménech, S. Perales-Momparler, A. Morales-Torres, I. Escuder-Bueno, Hydrological performance of green roofs at building and city scales under mediterranean conditions, Sustainability 10 (2018) 3105.
[49] S. Perales-Momparler, I. Andrés-Domenech, C. Hernández-Crespo, F. Vallés-Morán, M. Martín, I. Escuder-Bueno, J. Andreu, The role of monitoring sustainable drainage systems for promoting transition towards regenerative urban built environments: a case study in the Valencian region, Spain, J. Clean. Prod. 163 (2017) 113–124, https://doi.org/10.1016/j.jclepro.2016.05.153.
[50] Weather Spark. https://es.weatherspark.com.
[51] Y. Roskov, T. Kunze, T. Orrell, L. Abucay, L. Paglinawan, A. Culham, N. Bailly, P. Kirk, T. Bourgoin, G. Baillargeon, W. Decock, A. De Wever, V. Didžiulis (Eds.), Species 2000 & ITIS Catalogue of Life, 2014 Annual Checklist. Digital Resource at, Naturalis, Leiden, the Netherlands, 2014. Species 2000, www.catalogueoflife.org/annual-checklist/2014.
[52] Y. de Jong, M. Verbeek, V. Michelsen, P. Bjørn, W. Los, F. Steeman, N. Bailly, C. Basire, P. Chylarecki, E. Stloukal, G. Hagedorn, F.T. Wetzel, F. Glöckler, A. Kroupa, G. Korb, A. Hoffmann, C. Häuser, A. Kohlbecker, A. Müller, A. Güntsch, P. Stoev, L. Penev, Fauna Europaea – all European animal species on the web, Biodivers. Data J. 2 (2014) 4034. https://www.ncbi.nlm.nih.gov/pmc/articles/PMC4206781/pdf/biodiversity_data_journal-2-e4034.pdf.
[53] C.E. Shannon, A mathematical theory of communication, Bell System Technical J 27 (1948) 379–423.
[54] C.E. Shannon, W. Weaver, The Mathematical Theory of Communication, University of Illinois Press, Urbana, 1949.
[55] L. Jost, Entropy and diversity, Oikos 113 (2006) 363–375.
[56] C. Cultid-Medina, F. Escobar, Pautas para la estimación y comparación estadística de la diversidad biológica (qD), in: C.E. Moreno (Ed.), La biodiversidad en un mundo cambiante: Fundamentos teóricos y metodológicos para su estudio, Universidad Autónoma del Estado de Hidalgo/Libermex, Ciudad de México, 2019, pp. 175–202. En.
[57] R.K. Colwell, J.A. Coddington, Estimating terrestrial biodiversity through extrapolation, Philos. Trans. R. Soc. (Series B) 345 (1994) 101–118.
[58] A. Chao, Non-parametric estimation of the number of classes in a population, Scand. J. Stat. 11 (1984) 265–270.
[59] A. Chao, Estimating the population size for capture-recapture data with unequal catchability, Biometrics 43 (1987) 783–791.
[60] K.P. Burnham, W.S. Overton, Estimation of the size of a closed population when capture probabilities vary among animals, Biometrika 65 (1978) 623–633.
[61] K.P. Burnham, W.S. Overton, Robust estimation of population size when capture probabilities vary among animals, Ecology 60 (1979) 927–936.
[62] J. Heltshe, N.E. Forrester, Estimating species richness using the jackknife procedure, Biometrics 39 (1983) 1–11.
[63] E.P. Smith, G. van Belle, Nonparametric estimation of species richness, Biometrics 40 (1984) 119–129.
[64] R.L. Chazdon, R.K. Colwell, J.S. Denslow, M.R. Guariguata, Statistical methods for estimating species richness of woody regeneration in primary and secondary rain forests of NE Costa Rica, in: F. Dallmeier, J.A. Comiskey (Eds.), Forest Biodiversity Research, Monitoring and Modelling: Conceptual Background and Old World Case Studies, Parthenon Publishing, Paris, 1998, pp. 285–309.
[65] A. Chao, W.H. Hwang, Y.C. Chen, C.Y. Kuo, Estimating the number of shared species in two communities, Stat. Sin. 10 (2000) 227–246.
[66] R.K. Colwell, EstimateS: statistical estimation of species richness and shared species from samples. http://viceroy.eeb.uconn.edu/estimates/, 2019.
[67] H.L. Sanders, Benthic studies in Buzzard Bay. III. The structure of the softbottom community, Lymnol. Oceanogr. 5 (1960), 1338-153.
[68] D.H. Loya-Salinas, A. Escofet, Aportaciones al c5lculo del Indice de Valor Biol6gico (Sanders, 1960). Contriiution to the calculation of the Biological Value Index (Sanders, 1960), Cienc. Mar. X (2) (1990) 97–115, y.
[69] IBM Corp. Released, IBM SPSS Statistics for Windows, Version 27.0, IBM Corp, Armonk, NY, 2020.

[70] J.R. Bray, J.T. Curtis, An ordination of the upland forest communities of southern Wisconsin, Ecol. Monogr. 27 (1957) 325–349, https://doi.org/10.2307/1942268.

[71] K.R. Clarke, R.N. Gorley, PRIMER V7: User Manual/Tutorial, PRIMER-E, 2015.

[72] L.L. Timms, J.J. Bowden, K.S. Summerville, ChM. Buddle, Does species-level resolution matter? Taxonomic sufficiency in terrestrial arthropod biodiversity studies, Insect Conser. Divers. 6 (2013) 453–462.

[73] R. Davies, R. Simcock, R. Toft, Islands in the sky urban biodiversity enhance-ment in NZ on indigenous living roof landscapes, in: 44th Annual Conference of the Architectural Science Association, 2010 (ANZAScA).

[74] S. Braaker, J. Ghazoul, M.K. Obrist, M. Moretti, Habitat connectivity shapes urban arthropod communities: the key role of green roofs, Ecology 95 (2014) 1010–1021, https://doi.org/10.1890/13-0705.1.

[75] G.L. Pardee, S.M. Philpott, Native plants are the bee's knees: local and landscape predictors of bee richness and abundance in backyard gardens, Urban Ecosyst. 17 (2014) 641–659, https://doi.org/10.1007/s11252-014-0349-0.

[76] J.S. Macivor, Building height matters: nesting activity of bees and wasps on vegetated roofs, Israel J. Ecol. Evol. (2015) 1–9.

[77] J.S. MacIvor, K. Ksiazek, Invertebrates on green roofs in "green roof ecosystems", in: R.K. Sutton (Ed.), Serie. Ecological Studies, vol. 223, Springer, 2016, p. 447.

[78] S.M. Philpott, P. Bichier, Local and landscape drivers of predation services in urban gardens, Ecol. Appl. 27 (2017) 966–976, https://doi.org/10.1002/eap.1500.

[79] A.K. Brunbjerg, J.D. Hale, A.J. Bates, R.E. Fowler, E.J. Rosenfeld, J.P. Sadler, Can patterns of urban biodiversity be predicted using simple measures of green infrastructure? Urban For. Urban Green. 32 (2018) 143–153, https://doi.org/10.1016/j.ufug.2018.03.015.

[80] K. Kyrö, S. Brenneisen, J. Kotze, A. Szallies, M. Gerner, S. Lehvävirta, Local habitat characteristics have a stronger effect than the surrounding urban landscape on beetle communities on green roofs, Urban For. Urban Green. 29 (2018) 122–130.

[81] S.M. Philpott, S. Albuquerque, P. Bichier, H. Cohen, M.H. Egerer, C. Kirk, K. Will, Local and landscape drivers of carabid activity, species richness, and traits in urban gardens in coastal California, Insects 10 (2019) 112, https://doi.org/10.3390/10040112.

[82] G. Kadas, Rare invertebrates colonizing green roofs in london, Urban Habitats 4 (2006) 66–86.

[83] R.H. MacArthur, E.O. Wilson, The Theory of Island Biogeography. Series: Princeton Landmarks in Biology, Princeton University Press, 1967, p. 224.

[84] S.G. Potts, B. Vulliamy, A. Dafni, G. Ne`eman, P. Willmer, Linking bees and flowers: how do floral communities structure pollinator communities, Ecology 84 (10) (2003) 2628–2642.

[85] S.D. Hendrix, K.S. Kwaiser, S.B. Heard, Bee communities (Hymenoptera: apoidea) of small Iowa hill prairies are as diverse and rich as those of large prairie preserves, Biodivers. Conserv. 19 (2010) 1699–1709.

[86] V. Wojcik, Resource abundance and distribution drive bee visitation within developing tropical urban landscapes, J. Pollinat. Ecol. 4 (7) (2011) 48–56.

[87] T.R.E. Southwood, V.K. Brown, P.M. Reader, The relationships of plant and insect diversities in succession, Biol. J. Linn. Soc. 12 (1979) 327–348.

[88] S. Brenneisen, Space for urban wildlife: designing green roofs as habitats in Switzerland, Urban Habitats 4 (1) (2006) 27–36.

[89] G. Grant, Exten. Green Roofs London Urban Habit. 4 (1) (2006) 51–65.

[90] R.B. Root, Organization of a plant-arthropod association in simple and diverse habitats: the fauna of collards (Brassica oleracea), Ecol. Monogr. 43 (1973) 95–124, https://doi.org/10.2307/1942161.

[91] M. Björkman, P.A. Hambäck, R.J. Hopkins, B. Rämert, Evaluating the enemies hypothesis in a clover-cabbage intercrop: effects of generalist and specialist natural enemies on the turnip root fly (Delia floralis), Agric. For. Entomol. 12 (2010) 123–132, https://doi.org/10.1111/j.1461-9563.2009.00452.x.

[92] N.E. McIntyre, J. Rango, W.F. Fagan, S.H. Faeth, Ground arthropod community structure in a heterogeneous urban environment, Landsc. Urban Plann. 52 (2001) 257–274.

[93] E. Shochat, P.S. Warren, S.H. Faeth, N.E. McIntyre, D. Hope, From patterns to emerging processes in mechanistic urban ecology, Trends Ecol. Evol. 21 (4) (2006).

[94] M.L. McKinney, Urbanization as a major cause of biotic homogenization, Biol. Conserv. 127 (2006) 247–260, https://doi.org/10.1016/j.biocon.2005.09.005.

[95] S. Skendžic, M. Zovko, I.P. Živkovic, V. Lešic, D. Lemic, The impact of climate change on agricultural insect pests, Insects 12 (2021) 440, https://doi.org/10.3390/insects12050440, 2021.

[96] L.J. Thomson, S. Macfadyen, A.A. Hoffmann, Predicting the effects of climate change on natural enemies of agricultural pests, Biol. Control 52 (2010) 296–306, https://doi.org/10.1016/j.biocontrol.2009.01.022.

[97] K.W. McCravy, A review of sampling and monitoring methods for beneficial arthropods in agroecosystems, Insects 9 (2018) 170, https://doi.org/10.3390/insects9040170.