



Can botanic gardens serve as refuges for taxonomic and functional diversity of Odonata? The case of the botanic garden of Castilla–La Mancha (Spain)

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Received: 26 February 2022 / Accepted: 17 September 2022 / Published online: 13 October 2022
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Abstract

In a scenario with declining biodiversity and habitat loss, botanic gardens could serve as refuges for invertebrates, but the opportunities they offer for animal conservation are still poorly understood. Odonata is a good model group for conservation studies, because it includes threatened species and responses to habitat disturbance are well documented. In this study, we assessed the role of the botanic garden of Castilla-La Mancha in Spain as a refuge for members of Odonata by analysing their taxonomic and functional diversity. We explored if the small size of the botanic garden might constrain the taxonomic diversity of Odonata and if low habitat diversity might limit their functional diversity. We sampled adult Odonata from five water bodies along a gradient of human impact and characterized the Odonata communities based on 12 functional traits in Odonata. We used a species–area relationship to control for differences in the size of water bodies. Compared with natural lakes, the Odonata communities contained less species and their functional diversity was lower in the botanic garden ponds, where generalist species were basically hosted. Despite these limitations, the botanic garden ponds hosted the number of species expected for natural water bodies with the moderate surface area and functional diversity, thereby demonstrating that they are a valuable habitat for Odonata in an urban environment. Appropriate management involving the removal of exotic fish and habitat diversification, including creating lotic environments, would increase the taxonomic and functional diversity of Odonata in this urban system.

Keywords Damselfly · Dragonfly · Functional trait · Insect conservation · Urban ecology

Introduction

Invertebrate biodiversity is declining rapidly worldwide (Eisenhauer et al. 2019) due to ongoing global change caused by large-scale, worldwide environmental impacts induced by human activities (Johnson et al. 2017). The insect biomass is decreasing (Hallmann et al. 2017; Lister and Garcia 2018; Wagner 2020) and species are increasingly

under threat (Sánchez-Bayo and Wyckhuys 2019) or becoming extinct (Cardoso et al. 2020) mainly because of habitat alterations driven by agricultural expansion, deforestation, and urbanization (Sánchez-Bayo and Wyckhuys 2019; Cardoso et al. 2020). Declines are occurring in the insect biomass and diversity, compromising functional and phylogenetic diversity, as well as ecological networks and ecosystem services (Cardoso et al. 2020; Wagner 2020), and thus urgent conservation actions are needed (Montgomery et al. 2020).

Urbanization is a frequent cause of severe declines in insect abundances (van Klink et al. 2020) and diversity (Fenoglio et al. 2020; Samways 2020) due to the presence of impervious surfaces, exotic species, habitat fragmentation, pollution, electromagnetic radiation, the urban heat island effect, and road impacts (Samways 2020; Fenoglio et al. 2021). However, cities can sometimes provide refuges for a substantial number of species (Theodorou et al. 2020; Gardiner et al. 2021). In particular, green spaces in urban areas can play key roles in insect conservation provided

Handling Editor: Roger P. Mormul.

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that they satisfy some minimum standards in terms of size, habitat heterogeneity, proportion of native plants, and plant diversity (Samways 2020; Samways et al. 2020; Fenoglio et al. 2021). Botanic gardens preserve 30% of global plant biodiversity and 41% of all threatened flora (Mounce et al. 2017). These gardens have heterogeneous designs and features around the world, and many of them should meet the required standards. Despite the conservation potential of botanic gardens, their true capacity for harbouring and preserving animal diversity is still poorly understood (van der Hoek 2015). Previous studies suggest that botanic gardens may host a high number of arthropods (Osborn and Samways 1996; Clark and Samways 1997; Ito et al. 2001; Suh and Samways 2001; Pryke and Samways 2009; Bora et al. 2014; Babošová et al. 2019; Arteaga et al. 2020), and even more than natural areas (Pryke and Samways 2009). Therefore, further exploration is needed to understand the possible roles of botanic gardens as faunal refuges, especially at mid-latitudes where 93% of these institutions are located (Mounce et al. 2017).

Within Odonata, 10% of the world's species (Sánchez-Bayo and Wyckhuys 2019) and 15% of European species (Kalkman et al. 2010) are facing the risk of extinction. Moreover, they are highly sensitive to habitat modification and degradation (Bulánková 1997; Chovanec and Raab 1997; Kutcher and Bried 2014; Martín and Maynou 2016; Vilenica et al. 2020, 2021). However, artificial or urban ponds can host Odonata communities, which usually comprise generalist species (Willigalla and Fartmann 2012; Goertzen and Suhling 2015; Villalobos-Jiménez et al. 2016; Seidu et al. 2018; Vilenica et al. 2020), although they can even offer suitable habitats for some endangered species if the macrophyte vegetation is rich and well structured (Chovanec 1994; Goertzen and Suhling 2015; Vilenica et al. 2016, 2020). The richness and abundance of these communities will depend on the habitat heterogeneity, presence and management of aquatic vegetation, fish, pollutants, surrounding pristine habitats, fluctuations in water levels, and degree of urbanization around ponds (Chovanec 1994; Willigalla and Fartmann 2012; Goertzen and Suhling 2013; Jeanmougin et al. 2014; Simaika et al. 2016; Villalobos-Jiménez et al. 2016; Seidu et al. 2018; Vilenica et al. 2020). Thus, urban ponds can act either as refuges or ecological traps for Odonata species depending on the occurrence of stressors and their tolerance by species (Harabiš and Dolný 2012; Villalobos-Jiménez et al. 2016).

To accurately assess the potential roles of botanic gardens as refuges for Odonata, the usual taxonomic approach should be complemented with an assessment of functional diversity. The taxonomic diversity can be characterised by high functional redundancy or high functional uniqueness (Ricotta et al. 2016, 2020; Rego et al. 2019). Functional diversity provides information about the redundancy in terms of niche

occupancy and the ecosystem functions performed by different species (McGill et al. 2006). Therefore, conservation priorities should consider not only species richness but also the functional distinctness of the species preserved (Vandewalle et al. 2010). The functional diversity of Odonata is sensitive to habitat disturbance in space (Dolný et al. 2012; Modiva et al. 2017; Dalzochio et al. 2018) and time (Assandri 2020; Dalzochio et al. 2020). At the community level, human impacts increase the functional traits associated with generalist Odonata species, whereas those of specialists tend to decrease or even disappear (Dolný et al. 2012; Dalzochio et al. 2018, 2020), especially lotic ones (Assandri 2020).

In addition, size limitations must be considered when assessing the roles of botanic gardens as refuges for Odonata diversity. Compared with natural water bodies, four scenarios are possible for Odonata diversity in botanic gardens (Fig. 1). First, the Odonata diversity in water bodies in botanic gardens could match that in natural water bodies if both have similar ecological and size characteristics (Pryke and Samways 2009). Second, the availability of a limited area can impose an upper limit on Odonata diversity (Oertli et al. 2002; Kadoya et al. 2004; Ruggiero et al. 2008; Oertli and Parris 2019). Third, habitat heterogeneity and vegetation structure may be more important than the available area for Odonata diversity (Osborn and Samways 1996; Kadoya et al. 2004; Honkanen et al. 2011;

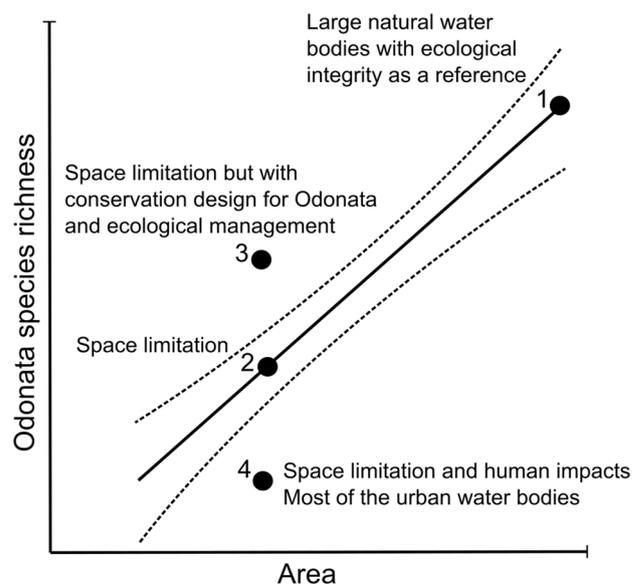


Fig. 1 Conceptual model of the potential of botanic garden ponds as refuges for Odonata considering the naturality degree and water body size. The figure shows theoretical representations of the four possible scenarios (indicated by the numbers) predicted for Odonata taxonomic diversity in the botanic gardens (black circles) using well-preserved natural lakes with no space limitation as the reference. The black and dashed lines represent the species–area relationship and its confidence interval, respectively

Goertzen and Suhling 2013), and thus the appropriate design of water bodies in botanic gardens could partially overcome their size limitation. Fourth, low biotope quality and current human impacts will decrease the value of botanic gardens as refuges for Odonata diversity, which is also the case for most urban water bodies (Villalobos-Jiménez et al. 2016; Oertli and Parris 2019).

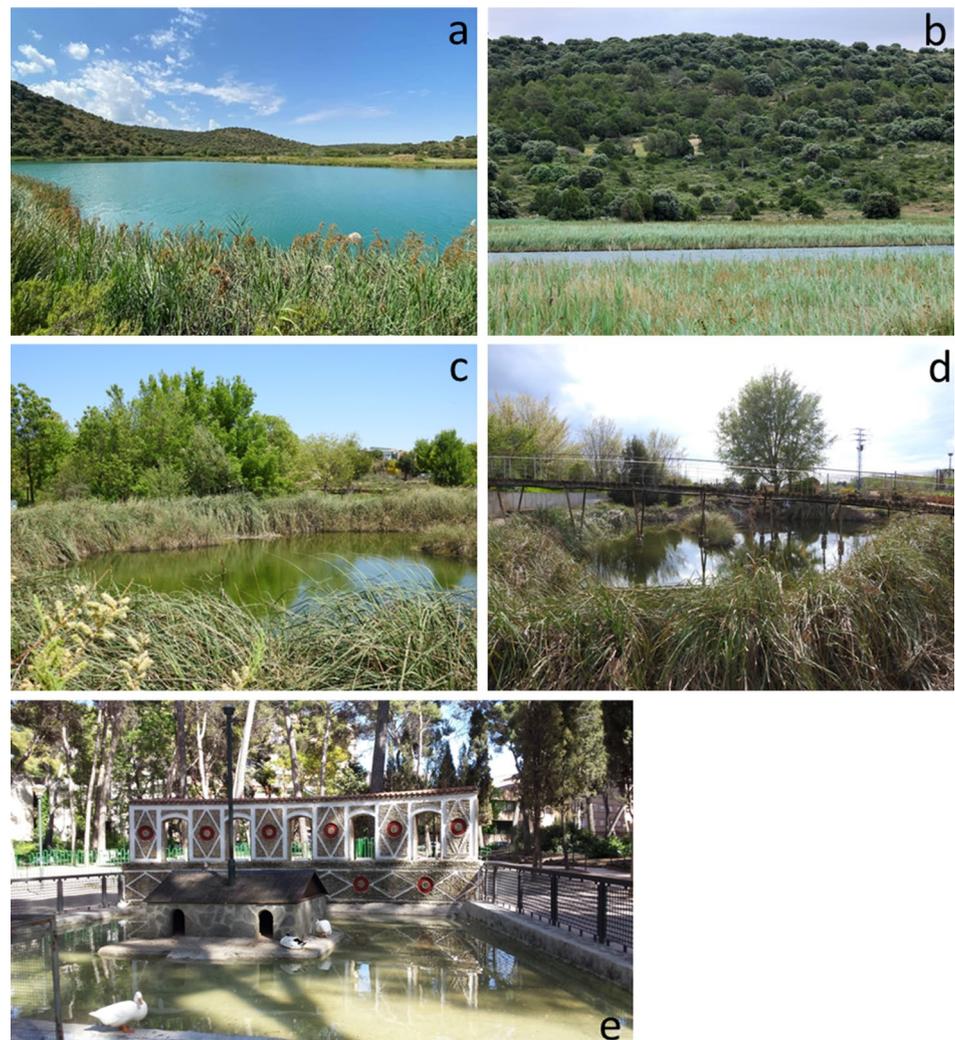
In this study, we assessed the capacity of the botanic garden of Castilla-La Mancha (Albacete, Spain) to act as an urban refuge for Odonata taxonomic and functional diversity. In particular, we compared the species richness, abundance, composition, and functional traits of adults in the Odonata communities between ponds in the natural systems and the botanic garden, while also controlling for differences in the size of water bodies.

Materials and methods

Study sites

We studied five water bodies in the province of Albacete (Spain, southeast of the Iberian Peninsula) along a gradient of human impact: two natural lakes, two ponds in a botanic garden, and one urban pond (Fig. 2). The two natural freshwater lakes in the Sierra de Alcaraz, Arquillo (38°45′09″N; 2°21′40″W; 990 m a.s.l.) and Ojos de Villaverde (38°48′29″N; 2°22′16″W; 921 m a.s.l.) are located 5.32 km and 47 km apart from Albacete City, respectively. These lakes with clear water and abundant riparian and submerged vegetation (Table S1) are fed by rivers or streams (Cirujano Bracamonte 1990), and they host populations of the invasive fish *Micropterus salmoides*. Arquillo is a karstic lake with an area of 6 ha and surrounded by *Quercus ilex* subsp. *ballota* (Desf.) Samp. woodland. Ojos de Villaverde

Fig. 2 Photographs of the study sites. Natural lakes: **a** Arquillo and **b** Ojos de Villaverde. Botanic garden wetland representations: **c** endorheic pond and **d** karstic pond. Abelardo Sánchez urban park: **e** artificial pond (free image uploaded to Wikimedia Commons by JPOK with the title “Estanque Parque Abelardo Sánchez de Albacete (patos)” under a Creative Commons license 3.0: www.creativecommons.org)



is an endorheic lake with an area of 7 ha and surrounded by *Juniperus thurifera* L. woodland.

The two artificial permanent ponds in the botanic garden of Castilla-La Mancha (38°58'27.6"N; 1°51'38.0"W; 700 m a.s.l.; located in the outskirts of the city of Albacete) are representations of endorheic wetland with an area of 0.065 ha and karstic wetland with an area of 0.050 ha. Both ponds resemble the natural lakes included in the study. The ponds are located 92 m apart and they were created in 2005. They have slightly eutrophic water and abundant riparian vegetation planted in 2015, with quite similar species compositions and structures to those of natural lakes (Table S1). The ponds host dense populations of the invasive fish *Cyprinus carpio* (common carp), which were introduced in 2010 and they prevent the growth of submerged vegetation. The botanic garden of Castilla-La Mancha occupies 7 ha and it contains around 29,000 plants belonging to about 2100 taxa, which include 25% of the protected flora in the region. For conservation and educational purposes, a substantial proportion of this flora is arranged in representations of over 40 natural habitats in Castilla-La Mancha region (central-south-eastern Spain). Furthermore, this botanic garden is certified as being “of ecological excellence” (i.e., managed by strictly ecological methods according to EU legal regulations; Santiago et al. 2021). The natural lakes selected for the study were the only ones available with sufficient ecological integrity and similarity to serve as references for the botanic garden ponds.

In addition, we selected an artificial pond with an area of 0.04 ha in Abelardo Sánchez urban park, which is a green space with an area of 7 ha in the centre of Albacete City (48°59'10"N; 1°51'25"W; 684 m a.s.l.; 1.5 km from the botanic garden), to serve as a reference for a typical urban water body managed only for ornamental purposes. This pond with clear water and no vegetation hosts a dense population of domestic ducks. The pond is surrounded by a *Pinus halepensis* plantation and it is frequently drained for cleaning operations. This was the only urban pond available for the study because the remaining city ponds were drained as a preventive measure during the COVID-19 epidemic (from 13 March 2020 onwards), which coincided with the sampling period.

Odonata sampling

At each water body, adult Odonata were sampled each week from 27 May to 24 July, 2020, and on three additional occasions from mid-September to mid-October 2020 until no further species were found. Odonata surveys were conducted between 10:30 and 15:00 on sunny and windless days by two observers (two from Guillermo García-Saúco, Manel Pinilla-Rosa, and Pablo Ferrandis). During each visit, we identified and counted all Odonata observed along permanent transects

on the shore of each water body, where they were assigned to species level according to Dijkstra and Lewington (2006). Individuals that were difficult to recognise, such as some females or teneral, were captured with an entomological net and released after identification (permission was granted by the Wildlife Service of Albacete province). Sampling was adaptive (Sato and Riddiford 2007), where a minimum of 30 min was spent per water body at each visit and 10 min extra were added if new species appeared. In total, 11 synchronised sampling surveys were performed at each of the natural water bodies and botanic garden water bodies. However, sampling was discontinued at the urban park pond after the three first visits due to the lack of Odonata.

Functional traits

Twelve Odonata functional traits (Table S2) were scored for each species based on previously published data. Body size, flight period, and detectability were as described by Dijkstra and Lewington (2006), distribution area by Boudot et al. (2009), and emergence period by Cham (2011) and Maynou and Martín (2019). Data for water type preference, voltinism, activity type, territoriality, oviposition type (endophytic or exophytic), oviposition behaviour (in tandem or solitary), and water temporality were as described by Harabiš and Hronková (2020). For the species not described by Harabiš and Hronková (2020), data were inferred from their sister species and the descriptions of Dijkstra and Lewington (2006). The descriptions given by Corbet et al. (2006) and Guebailia et al. (2016) were also used for *Trithemis* spp. When several values for voltinism were reported for the same species, we used the value with the highest number of generations due to the typically high spring and summer temperatures in the province of Albacete, which are expected to accelerate larval development (Suhling et al. 2015). Lotic species were scored as those that develop in low current waters to separate them from those that develop exclusively (lentic species) or facultatively (indifferent species) in standing water.

Statistical analysis

All analyses were conducted in the R programming environment (R Core Team 2020). Coverage-based sampling completeness (Roswell et al. 2021) was calculated for each water body with the iNEXT package (Hsieh et al. 2016) using 100 permutations. We statistically tested the differences in species compositions among water bodies and sampling dates using two complementary methods with the vegan package (Oksanen 2011). First, non-metric multi-dimensional scaling (NMDS) was performed based on a matrix containing the abundance of each species per water body and date. Data were fourth-root transformed to down-weight the importance

of the most abundant species (Clarke and Gorley 2006). Second, two-way PERMANOVA (9999 permutations) was conducted to analyse the differences in composition between areas (natural vs. botanic garden) and between individual water bodies. In addition, pairwise taxonomic beta diversity values between water bodies and their two components, i.e., species replacement or species gain or loss (Cardoso et al. 2014), were obtained with the BAT package (Cardoso et al. 2015). Beta diversity represents the degree of change between communities in space (Baselga 2010) or time (Cardoso et al. 2014).

Functional diversity was characterized using four complementary methods to explore different functional properties of communities (Mammola et al. 2021): functional richness, functional dissimilarity, community weighted mean (CWM) of each functional trait, and the partition of functional beta diversity into trait replacement and trait gain or loss among communities. Functional richness represents the size of the functional space occupied by the community, which is measured as the sum of the branch lengths in the functional dendrogram for all species present in a community (Cardoso et al. 2015). Functional richness values were calculated for each water body with the BAT package and functional dendrograms were generated using the UPGMA method (unweighted pair group method with arithmetic mean; Cardoso et al. 2011) for total Odonata and separately for Zygoptera and Anisoptera. The Rao index represents the functional dissimilarity between conspecific and heterospecific pairs of individuals in a community, and it is bounded between zero (minimal dissimilarity) and one (maximal dissimilarity) (de Bello et al. 2016). Rao index values were calculated for each water body with the “melodic” function (de Bello et al. 2016) for the same three taxonomic groups considered for functional richness analysis. CWM represents the value of a trait weighted by species abundances in the community and it indicates the main functional features of the community (Lavelle et al. 2008). The CWM for each trait was calculated for each water body with the FD package (Laliberté et al. 2014) separately for Zygoptera and Anisoptera. Fourth, pairwise functional beta diversity values and their two components (trait replacement and trait gain or loss) between water bodies are the functional analogues of taxonomic beta diversity (Cardoso et al. 2015), and they were obtained in the same manner as for functional richness analysis. Except for CWMs, the Gower distance was used as a distance measure to calculate all of the functional metrics (Petchey and Gaston 2007; Laliberté and Legendre 2010; de Bello et al. 2013). We separated Zygoptera from Anisoptera in all functional analyses, because they may have clear functional differences (Sacchi and Hardersen 2013; Rocha-Ortega et al. 2020). We used the total data set for all analyses, but we included a second data set with only spring–summer data for Anisoptera CWMs.

The ponds in the botanic garden were considerably smaller than the reference natural lakes. To consider these differences in water body size in our study and to determine which of the four scenarios in our conceptual model (Fig. 1) best described the role of the botanic garden ponds as urban refuges, we regressed Odonata species richness on water body area using previously published data for natural or highly naturalised lakes with high ecological integrity in Iberian and Balearic Island systems (Ferrerías-Romero 1983; Agüero-Pelegrín and Ferrerías-Romero 1992; Lockwood 2006; Martínez and Ocharan 2006; Soler and Méndez 2009; Escolà et al. 2011; Maynou i Señé 2011; Gainzarain 2012; Campos et al. 2013; Torralba-Burrial et al. 2013; Evangelio Piñach and Díaz-Martínez 2017; Maynou et al. 2017; Moreno-Benítez 2018; Maynou and Martín 2019; Brotóns Padilla et al. 2020; Cuenca Espinosa 2020; Morales et al. 2022), including our two reference lakes. Overall, we included 53 water bodies with areas ranging from 0.000093 to 1400 ha. If the area of a water body was not available, we calculated it using Google Earth (www.google.com/intl/es/earth/). We checked the assumption of normality and absence of overdispersion using tests in the car package (Fox 2007), before fitting a linear regression model using the “lm” function in R. Finally, we overlaid the botanic garden ponds on the regression plot to determine the best match with the four scenarios described in Fig. 1.

Results

Species richness

Overall, we recorded 31 Odonata species, where 12 belonged to Zygoptera and 19 to Anisoptera (Table S3). The species richness was higher in natural lakes than botanic garden ponds (Table 1). The species richness from highest to lowest followed the order of: Arquillo > Ojos de Villaverde > both botanic garden ponds > the urban pond in Albacete City where no Odonata were observed (Table 1). Coverage-based sampling completeness was higher than 96% for all water bodies and the 95% confidence interval was always higher than 93% (Table 1), thereby demonstrating that our sampling surveys covered almost the whole Odonata community at each water body.

Species abundance

In total, we made 2180 Odonata observations: 676 Odonata adults in Arquillo, 1119 in Ojos de Villaverde, 130 in the endorheic pond, and 255 in the karstic pond at the botanic garden (Table S3). The most abundant members of Zygoptera were *Calopteryx haemorrhoidalis* in Arquillo, *Ceragrion tenellum* in Ojos de Villaverde, *Ischnura*

Table 1 Odonata species richness and percentage coverage with the 95% confidence interval (CI) for each water body

	Endorheic pond	Karstic pond	Arquillo	Ojos de Villaverde
Richness	12	12	24	19
Coverage (%)	96.2	98.0	99.0	99.6
95% CI	93.5–98.8	96.5–99.5	98.2–99.7	99.4–99.9

Endorheic and karstic ponds referred to in the column headings were the botanic garden ponds. Arquillo and Ojos de Villaverde correspond to the natural lakes

graellsii in the endorheic pond, and *Erythromma lindenii* in the karstic pond at the botanic garden (Table S3). The most abundant Anisoptera were *Sympetrum fonscolombii*, *Anax imperator*, and *Orthetrum coerulescens* in Arquillo, *Aeshna isocetes* in Ojos de Villaverde, *S. fonscolombii* and *A. imperator* in the endorheic pond, and *S. fonscolombii* in the karstic pond at the botanic garden (Table S3).

Species composition

NMDS obtained a stress value close to 0.1 (Fig. 3a). The first ordination axis segregated records from both natural lakes, and partially those from natural and botanic garden water bodies (Fig. 3a). The second ordination axis mainly separated records from natural and botanic garden water bodies (PERMANOVA: $F = 14.966$; $df = 1$; $R^2 = 0.253$; $p < 0.001$; Fig. 3a), as well as individual water bodies within each group (PERMANOVA: $F = 3.117$; $df = 2$; $R^2 = 0.105$; $p < 0.001$), primarily for the spring–summer samples (Fig. 3a). Natural lakes had 16 unique species (Table S3) and higher presence or abundance values for 21 species (Fig. 3b). Botanic garden ponds had three unique species (Table S3) and higher presence or abundance values for 10 species (Fig. 3b). Lotic species were only observed in natural lakes (Table S3; Fig. 3b).

The taxonomic beta diversity was high in all pairwise comparisons and even between both botanic garden ponds despite their spatial proximity (Table 2). Species replacement was dominant in the comparison between natural lakes (Table 2). Species gain or loss was dominant in the three comparisons between natural and botanic garden water bodies (Table 2). Both components were of similar magnitude in the comparisons between Arquillo and the karstic pond at the botanic garden, as well as between botanic garden ponds (Table 2).

Functional diversity

The functional richness values were higher in natural lakes than botanic garden ponds for the three taxonomic groups studied (Table 3). The botanic garden ponds had the lowest Rao index values for Anisoptera but they were intermediate between those in natural lakes for the total Odonata and Zygoptera, although all were < 0.5 (Table 4). The CWMs for Zygoptera differed between the natural and botanic garden

water bodies for five traits (Fig. 4). Compared with the ponds in the botanic garden, the CWMs for Zygoptera in natural lakes were characterized by larger body size, a smaller distribution range and higher scores in lotic environments, semivoltinism, and indifferent period of emergence (Fig. 4). The CWMs for Anisoptera differed between the natural and botanic garden water bodies for three traits in the analysis of the total data set and for six traits in the spring–summer analysis (Fig. 5). Compared with the botanic garden ponds, the CWMs for Anisoptera in natural lakes were characterized by a smaller distribution range and higher scores in permanent waters, semivoltinism, spring emergence, endophytic oviposition, and flier activity (Fig. 5).

The functional beta diversity was high in all comparisons, especially between natural lakes and botanic garden ponds (Table 5). Trait gain or loss was dominant for all taxonomic groups in all comparisons, except for four cases (Table 5) where trait replacement was dominant in the comparisons between the natural lakes for Zygoptera and between the karstic pond in the botanic garden and Arquillo for Anisoptera, and both components were of similar magnitude in the comparisons between natural lakes for the total Odonata and between the botanic garden ponds for Zygoptera.

Species–area relationships

A significant positive relationship was found between Odonata species richness and water body surface area ($F_{1,51} = 24.32$; $R^2 = 0.323$; $p < 0.001$; Fig. 6). The species richness in both botanic garden ponds was within the confidence interval of the regression line (Fig. 6), thereby matching with the second scenario depicted in Fig. 1.

Discussion

In this study, we assessed the role of the botanic garden of Castilla-La Mancha as an urban refuge for Odonata taxonomic and functional diversity. Overall, both types of diversity were higher in the natural lakes than the botanic garden ponds, although the species richness was as expected in the botanic garden ponds given their size. In addition, the natural lakes hosted specialist species associated with lotic environments and habitat heterogeneity,

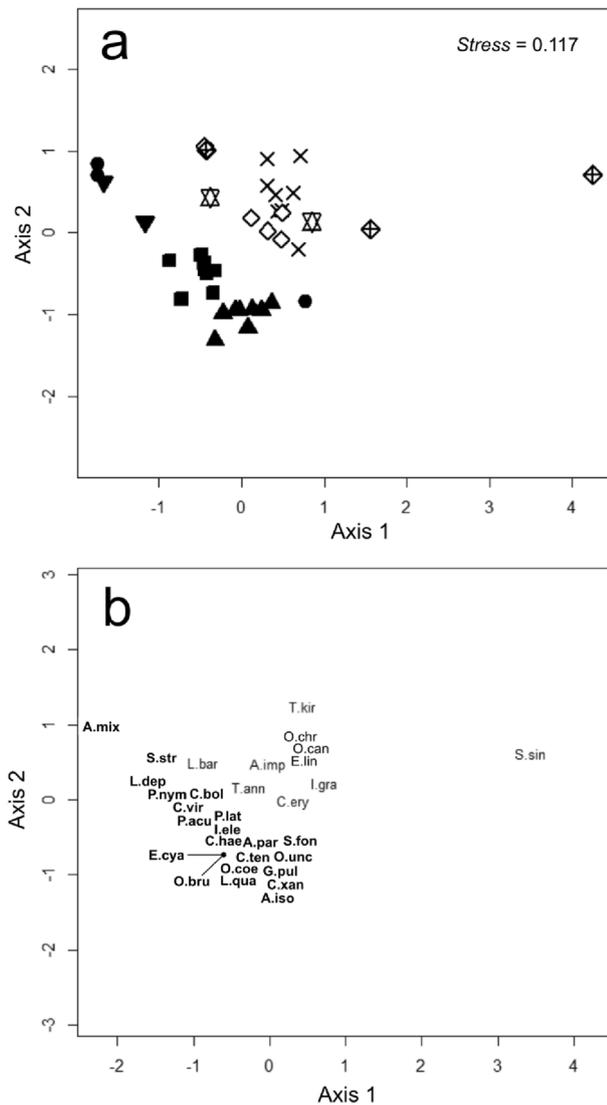


Fig. 3 **a** NMDS ordination of samples per site and date by species composition. Natural lakes: closed symbols; botanic garden ponds: open symbols. Arquillo: squares (spring–summer) and circles (fall); Ojos de Villaverde: triangles (spring–summer) and inverted triangles (fall); karstic pond in the botanic garden: diamonds (spring–summer) and crossed diamonds (fall); endorheic pond in the botanic garden: crosses (spring–summer) and stars (fall). No Odonata were detected on the last sampling dates at Ojos de Villaverde and the endorheic pond in the botanic garden. **b** Species scores obtained by NMDS ordination. Species codes comprise the first letter of the genus and three first letters of the species name presented in Table S3. Bold letters: species with high scores in the natural lakes; plain letters: species with high scores in the botanic garden ponds

where they probably came from rivers and streams connected to lakes. This functional group was absent in the urban ponds.

The richness and abundance of Odonata species were higher in the natural lakes compared with the botanic garden water bodies, and no Odonata were present in the urban pond

in the city of Albacete (Scenarios 2 and 4, respectively, from our predictions in Fig. 1). Artificial naturalised ponds can host Odonata communities with similar species richness and abundance to those found in natural areas (Willigalla and Fartmann 2012; Simaika et al. 2016). However, the species richness and abundance in these communities depend on the habitat diversity, presence and management of aquatic vegetation, presence of fish, pollutants, surrounding pristine habitats, fluctuations in the water level, and degree of urbanization around ponds (Chovanec 1994; Willigalla and Fartmann 2012; Goertzen and Suhling 2013; Jeanmougin et al. 2014; Simaika et al. 2016; Villalobos-Jiménez et al. 2016; Seidu et al. 2018). Thus, the absence of Odonata in the urban pond in Albacete City was not unexpected, because it was a very poor environment for these animals due to the absence of vegetation, severe water level fluctuations, high degree of surrounding urbanization, and the presence of humans as well as ducks that prey on Odonata (Ruggiero et al. 2008).

The NMDS ordination, PERMANOVA, and high taxonomic beta diversity values indicated differences in the species composition among the water bodies investigated in the present study. The Odonata communities hosted by artificial ponds mainly comprised generalist species, as also found in previous studies (Willigalla and Fartmann 2012; Villalobos-Jiménez et al. 2016; Seidu et al. 2018; Vilenica et al. 2020). However, the differences in the species compositions were remarkable even between the two ponds in the botanic garden. Given the high coverage achieved in the sampling surveys, the differences among ponds were probably due to habitat differences, such as variations in their flora and/or surrounding plant collections, which are known to influence the composition of Odonata communities (Chovanec and Raab 1997; Sato and Riddiford 2007; Remsburg and Turner 2009).

The Odonata communities in natural lakes had higher functional spaces than those in the botanic garden, and the functional dissimilarity in Anisoptera was more pronounced. Furthermore, the functional beta diversity indicated a loss of traits from natural to botanic garden water bodies. The functional diversity of Odonata responds to habitat modification and degradation in space (Dolný et al. 2012; Modiva et al. 2017; Dalzochio et al. 2018) and time (Assandri 2020; Dalzochio et al. 2020). Functional traits associated with specialist species tend to be missing in anthropic environments, whereas traits associated with generalist species increase (Dolný et al. 2012; Dalzochio et al. 2018, 2020). Accordingly, the CWMs showed that the natural communities had high scores for specialist traits, such as preference for lotic environments, smaller distribution range, semivoltinism, and endophytic oviposition, which were missing in the botanic garden communities.

Table 2 Taxonomic beta diversity (β_{Total}) and its components, species replacement (β_{repl}) and species gain or loss (β_{rich}), between each pair of water bodies

	Endorheic pond–Karstic pond	Endorheic pond–Arquillo	Endorheic pond–Ojos de Villaverde	Karstic pond–Arquillo	Karstic pond–Ojos de Villaverde	Arquillo–Ojos de Villaverde
β_{Total}	0.744	0.943	0.977	0.955	0.974	0.901
β_{rich}	0.407	0.714	0.809	0.471	0.643	0.271
β_{repl}	0.337	0.229	0.168	0.484	0.331	0.630

Endorheic and karstic ponds referred to in the column headings were the botanic garden ponds. Arquillo and Ojos de Villaverde correspond to the natural lakes

Table 3 Functional richness values for total Odonata and separately for Zygoptera and Anisoptera for each water body

	Endorheic pond	Karstic pond	Arquillo	Ojos de Villaverde
Odonata	3.750	3.281	6.170	5.709
Zygoptera	1.708	1.004	2.337	2.327
Anisoptera	2.254	2.498	4.231	3.664

Endorheic and karstic ponds referred to in the column headings were the botanic garden ponds. Arquillo and Ojos de Villaverde correspond to the natural lakes

Table 4 Rao index values for total Odonata and separately for Zygoptera and Anisoptera for each water body

	Endorheic pond	Karstic pond	Arquillo	Ojos de Villaverde
Odonata	0.234	0.255	0.323	0.152
Zygoptera	0.152	0.096	0.319	0.027
Anisoptera	0.303	0.291	0.386	0.448

Endorheic and karstic ponds referred to in the column headings were the botanic garden ponds. Arquillo and Ojos de Villaverde correspond to the natural lakes

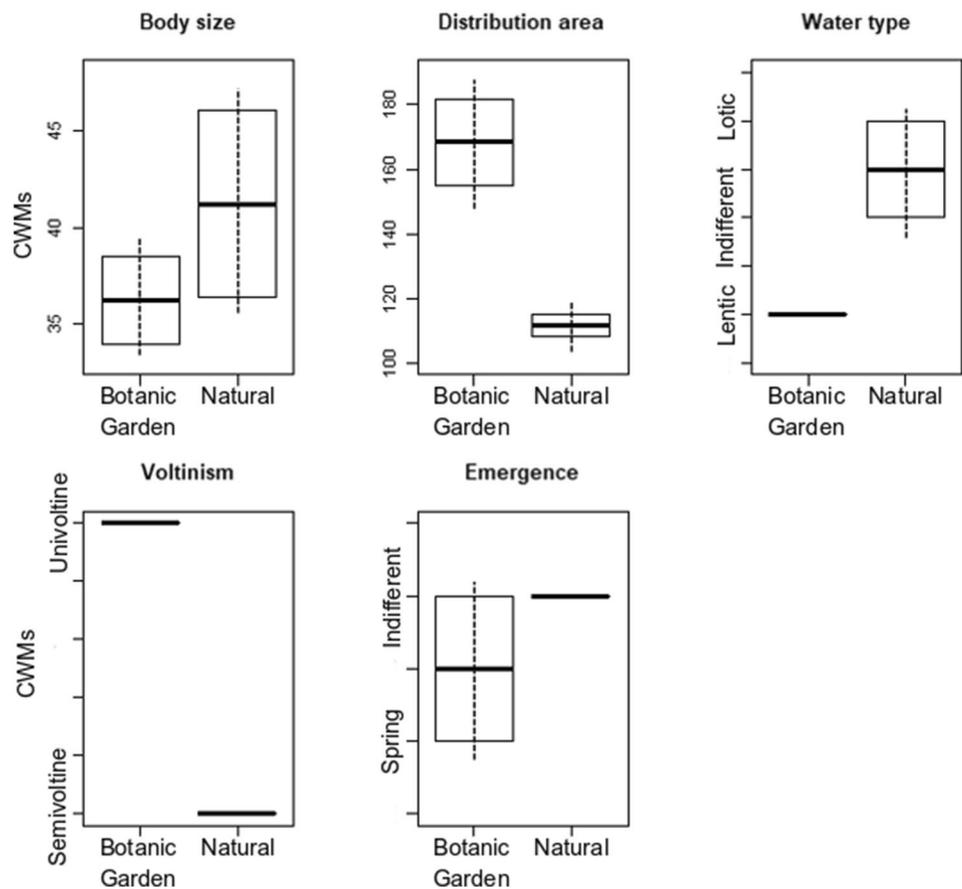
These findings highlight the challenges involved in recreating complex insect communities in botanic gardens, even those that host a highly diverse plant collection together with water bodies that recreate natural habitats. Nevertheless, the Odonata species richness in the ponds in the botanic garden of Castilla-La Mancha was that expected for their size, thereby indicating that this institution may serve as an urban refuge for Odonata, especially generalist species. The conservation value of this botanic garden should not be underestimated considering the total absence of Odonata in the pond monitored in the urban park, which is likely to be a generalised pattern in other ponds within the city because the management features of these ornamental ponds are similar (e.g., recurrent desiccation, lack of vegetation, and presence of domestic ducks). In addition, Odonata provide cultural services to people (Simaika and Samways 2008), which enhances the social importance of the botanic garden of Castilla-La Mancha. Overall, the high conservation value of botanic gardens should not be dismissed (Osborn and

Samways 1996; Clark and Samways 1997; Suh and Samways 2001; Pryke and Samways 2009; Bora et al. 2014; Arteaga et al. 2020).

Given the insurmountable limitation in terms of size, the appropriate management of ponds seems to be the most realistic option for increasing the richness of Odonata in botanic gardens (Scenario 3 in Fig. 1). In particular, we detected two main factors that might have limited the diversity of Odonata in the botanic garden ponds comprising the presence of predatory fish and the lack of lotic environments. The presence of large populations of common carp could explain the dominance of smaller species such as *E. lindemanni* and the lack of endophytic oviposition and flier activity according to the Anisoptera CWMs in the botanic garden ponds. Carps remove the submerged vegetation required for egg laying by Odonata species with endophytic oviposition (Guillermo-Ferreira and Del-Claro 2011; Villalobos-Jiménez et al. 2016). European Anisoptera with endophytic oviposition mainly belong to Aesnidae, which have flier activity (Harabiš and Hronková 2020), so the flier activity was also indirectly hindered by the presence of carp in the ponds. In addition, fish normally reduce the abundance of Odonata (Morin 1984; Johansson and Brodin 2003; Knight et al. 2005) and lead to the overrepresentation of small Odonata species by removing large competitors (Johansson and Brodin 2003). Nevertheless, the effects of fish on Odonata depend on the trophic level of the former. In natural lakes, top predators such as *Micropterus salmoides* can indirectly favour Odonata diversity by controlling herbivore populations, thereby reducing the pressure on aquatic vegetation and habitat heterogeneity (Maezono and Miyashita 2004). Thus, the fish present in ponds in botanic gardens should be carefully evaluated to determine the effects of the ecological characteristics of fish on other faunal groups, and they must even be removed if severe generalised negative effects are detected.

Lotic habitats were absent in the botanic garden of Castilla-La Mancha but present near the natural lakes, which may explain the lower functional diversity observed in the botanic garden ponds because lotic species have specialised traits, such as narrower distribution areas (Hof et al. 2006), lower voltinism (Corbet et al. 2006), a requirement for permanent water (Harabiš and Hronková 2020), and greater

Fig. 4 CWMs for traits in Zygoptera that differed between botanic garden ponds (Botanic Garden) and natural lakes (Natural). Horizontal black bars represent the mean value, boxes represent the variability range, and vertical dashed bars represent the standard deviation



body size in European Zygoptera (Dijkstra and Lewington 2006), as shown by the CWMs in the present study. In addition, the presence of lotic habitats near the natural lakes could have influenced the different compositions observed among water body types. Lotic species probably did not use the natural lakes for reproduction but they could have been utilized for other important activities, such as foraging, maturation, or as stepping stones for connecting populations (Harabiš and Dolný 2012; Villalobos-Jiménez et al. 2016; Maynou et al. 2017). Therefore, recreating lotic habitats in the botanic garden, if possible, may allow the establishment of specialist lotic species (Dijkstra and Lewington 2006; Hof et al. 2006; Kalkman et al. 2018), which seem to be particularly sensitive to human impacts (Assandri 2020) and they are usually absent in artificial environments (Willigalla and Fartmann 2012; Villalobos-Jiménez et al. 2016; Seidu et al. 2018).

In conclusion, the results obtained in this study greatly update our knowledge of the roles of botanic gardens for insect conservation. Urban biodiversity can increase public health (e.g., by controlling the spread of disease and enhancing mental health; Fuller et al. 2007;

Samways 2020), so we recommend improving the design of ponds in botanic gardens, and even in other green urban spaces, in order to increase the taxonomic and functional diversity of their Odonata communities. To achieve this aim, in addition to the presence of natural vegetation, urban ponds (1) must contain as many microhabitats as possible to compensate for their limited size, including lotic habitats where feasible to allow the establishment of specialist species; and (2) exotic fish should be removed, especially common carp. In addition, the different functional responses of Zygoptera and Anisoptera suggest that management should target their specific requirements. Ideally, improvements in management should aim to increase the presence of self-sustaining populations of Odonata. Thus, we advise sampling Odonata larvae as well as adults to assess the reproduction of species in green urban spaces. In this study, we also demonstrated the utility of functional diversity approaches for understanding how human impacts might influence insect communities and their conservation in urban environments.

Fig. 5 CWMs for the six traits in Anisoptera that differed between botanic garden ponds (botanic garden) and natural lakes (Natural) in the spring–summer data analysis. Asterisks indicate the three traits that also differed between both types of water bodies in the total data set analysis, where all of the patterns were the same as those in the spring–summer analysis. Horizontal black bars represent the mean value, boxes represent the variability range, and vertical dashed bars represent the standard deviation

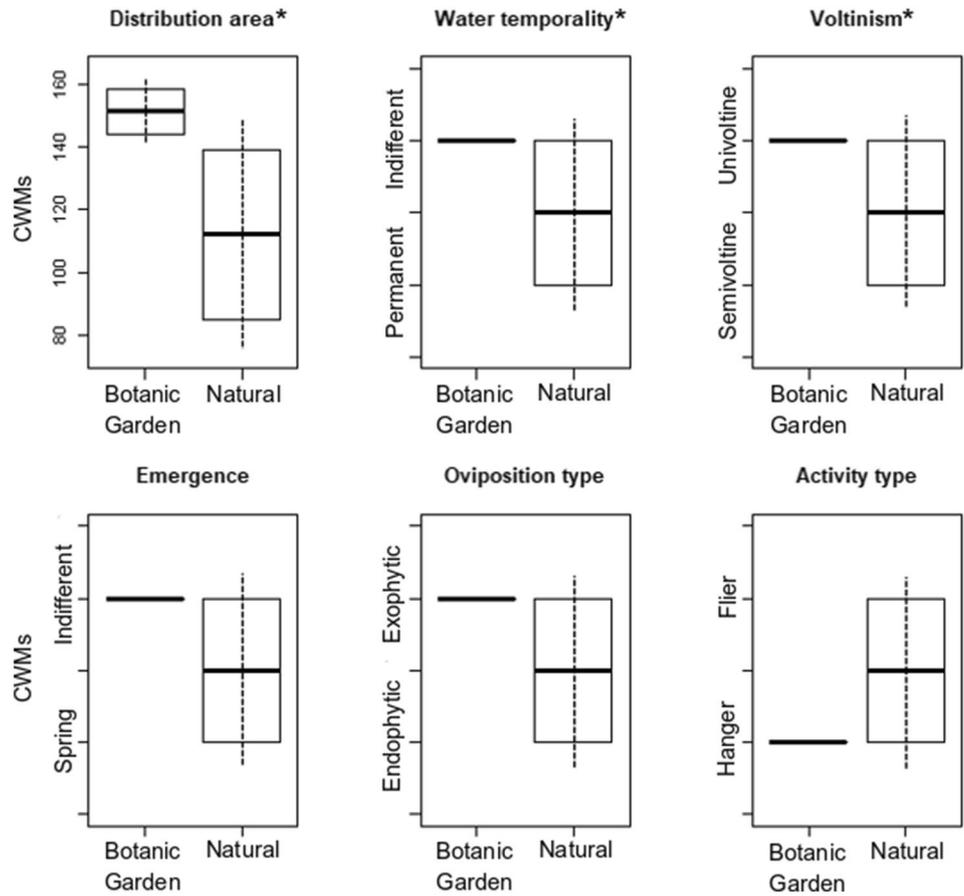


Table 5 Functional beta diversity (β_{Total}) and its components, functional replacement (β_{repl}) and functional gain or loss (β_{rich}), for total Odonata and separately for Zygoptera and Anisoptera between each pair of water bodies

	Endorheic pond–Karstic pond	Endorheic pond–Arquillo	Endorheic pond–Ojos de Villaverde	Karstic pond–Arquillo	Karstic pond–Ojos de Villaverde	Arquillo–Ojos de Villaverde
Odonata						
β_{Total}	0.680	0.827	0.926	0.811	0.905	0.768
β_{rich}	0.429	0.795	0.851	0.538	0.689	0.304
β_{repl}	0.251	0.032	0.075	0.273	0.216	0.464
Zygoptera						
β_{Total}	0.713	0.861	0.940	0.870	0.943	0.828
β_{rich}	0.379	0.808	0.855	0.593	0.710	0.266
β_{repl}	0.334	0.053	0.085	0.277	0.233	0.562
Anisoptera						
β_{Total}	0.591	0.620	0.886	0.433	0.708	0.689
β_{rich}	0.591	0.612	0.787	0.046	0.527	0.503
β_{repl}	0.000	0.008	0.099	0.387	0.181	0.186

Endorheic and karstic ponds referred to in the column headings were the botanic garden ponds. Arquillo and Ojos de Villaverde correspond to the natural lakes

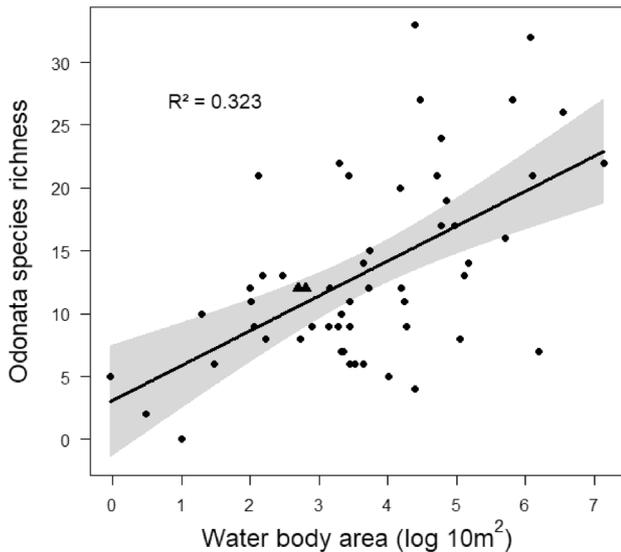


Fig. 6 Correlation between Odonata species richness and water body area ($\log_{10} \text{m}^2$). The black line and grey area represent the regression line and its confidence interval, respectively. Black circles: water bodies used for the analysis; triangles: botanic garden ponds

Supplementary Information The online version contains supplementary material available at <https://doi.org/10.1007/s10201-022-00704-3>.

Acknowledgements The authors thank the Wildlife Service of Albacete province, especially Antonio Catalán and José Antonio López Donate, for assistance with selecting natural systems for the study and permission for field sampling. We also thank two anonymous reviewers for their valuable comments on an early version of the manuscript.

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