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Abstract

Green spaces can represent important habitats for animals in fragmented urban environments. Despite the high conservation status and the ecological importance of bats, little is known about their activity patterns and habitat use in urban green spaces. The purpose of this study was to determine activity patterns of bats among urban green spaces and to assess the potential habitat connectivity of green spaces across the city of Vienna, Austria. Bat activity was monitored using batcorders at 106 survey points in parks and residential areas. To assess threshold distances for the connectivity analysis I determined the proportion of land cover classes within five different buffers surrounding the survey points to evaluate which scale may predict total bat activity. Furthermore, I used literature research on foraging distances of common bats in the study area to determine species specific threshold distances. I identified calls from 12 bat species and 3 species groups. The most common species were *Hypsugo savii*, *Pipistrellus pygmaeus*, *Nyctalus noctula*, *P. pipistrellus* and the *P. kuhlii*/*nathusii* group. There was no significant difference in total bat activity between small parks, large parks and residential areas, but the activity of *P. kuhlii*/*nathusii* group was significantly higher in small parks than in residential areas. The overall connectivity of urban green spaces was quite low in Vienna, but increased with higher threshold distances and when adding forest patches to the analysis. With smaller threshold distance (400 m) small green patches acted as important stepping stones between larger ones. For models using 1500 m as threshold distance, large green space patches were the main contributors to overall connectivity. In terms of bat conservation in fragmented environments, these results are able to guide urban green space management and detailed species specific connectivity studies, to protect a highly mobile animal group in urban areas.

Zusammenfassung

In fragmentierten städtischen Gebieten können Grünflächen wichtige Lebensräume für Tiere darstellen. Obwohl Fledermäuse strengen Artenschutzbedingungen unterliegen und wichtige ökologische Funktionen erfüllen, ist sehr wenig über ihre Aktivitätsmuster und die Rolle urbaner Grünflächen als Fledermaushabitate bekannt. Ziel der Arbeit war es, die Aktivität von Fledermäusen in urbanen Grünflächen zu untersuchen und die potentielle Konnektivität von Grünflächen als Fledermaushabitate im Bereich der Stadt Wien zu bestimmen. Hierzu wurde mittels Batcordern die Fledermausaktivität an 106 Untersuchungspunkten in Grünanlagen und Wohngebieten ermittelt. Zur Festlegung der Grenzwertdistanzen für die Konnektivitätsanalyse habe ich für fünf Pufferzonen rund um die Untersuchungspunkte den jeweiligen Anteil von Landbedeckungsklassen bestimmt, um anschließend zu evaluieren, welcher Maßstab die allgemeine Fledermausaktivität prognostizieren kann. Zur Bestimmung artspezifischer Grenzwertdistanzen recherchierte ich in der Literatur nach Nahrungssuchdistanzen häufiger Fledermausarten des Untersuchungsgebiets. Ich konnte für Wien insgesamt 12 Fledermausarten und 3 Artengruppen nachweisen. Die häufigsten Arten waren *Hypsugo savii*, *Pipistrellus pygmaeus*, *Nyctalus noctula*, *P. pipistrellus* und die *P. kuhlii*/*nathusii* Gruppe. Es wurden keine signifikanten Unterschiede der allgemeinen Fledermausaktivität in kleinen bzw. großen Grünanlagen und Wohngebieten festgestellt, jedoch war die Aktivität der *P. kuhlii*/*nathusii* Gruppe in kleinen Grünanlagen signifikant höher als in Wohngebieten. Insgesamt war die Konnektivität der Wiener Grünflächen recht gering, stieg jedoch bei einer höheren Grenzwertdistanz und der Einbeziehung von Waldflächen in die Konnektivitätsanalyse an. Für die geringere Grenzwertdistanz (400 m) agierten kleine Grünflächen als wichtige Trittsteine zwischen größeren. Für die Modelle mit einer Grenzwertdistanz von 1500 m leisteten große Grünflächen den wichtigsten Beitrag zur Gesamtkonnektivität. Die Ergebnisse meiner Arbeiten dienen als wichtige Grundlage für urbanes Grünflächenmanagement, ausführliche artspezifische Konnektivitätsstudien sowie dem Schutz einer hochmobilen Tiergruppe in stark fragmentierten Lebensräumen.

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Introduction

Today more than half of the world population lives in urban areas and a progressive trend for the next decades is projected (UN 2014). The urban population of the world currently comprises 3.9 billion people (54% of the world population) and population growth in urban areas is predicted to increase 2.5 billion people by 2050 which corresponds to more than 66% of the world's urban population (UN 2014).

This demographic development results in rapidly expanding urban areas worldwide (Pauchard et al. 2006; Grimm et al. 2008) and urbanization is highlighted as driver of biodiversity change (Grimm et al. 2008; Faeth et al. 2011), biotic homogenization (McKinney 2006) and species endangerment (Wilcove et al. 1998; Czech et al. 2000).The effects of urbanization differ between taxa, but generally have a declining effect on species richness and diversity of birds and arthropods, whereas plant species richness often increases (McKinney 2008, Faeth et al. 2011). Furthermore urban biodiversity is often characterized by nonnative and invasive species (McKinney 2008, Faeth et al. 2011).

Urbanization is characterized by drastic land-use and land-cover transformations, causing several pressures on biodiversity (Grimm et al. 2008), such as habitat loss and fragmentation (Scolozzi and Geneletti 2012). These landscape alterations lead to spatial heterogeneous land use mosaics (Cadenasso et al. 2007), where remaining vegetated habitats such as urban green spaces are distinct patches of various sizes distributed within the urban matrix and profoundly disturbed by human activities (Avila-Flores and Fenton 2005). Therefore, it is important to understand the effects of urban landscape structure (e.g. patch area, landscape connectivity) on different taxa, for an effective management of urban biodiversity.

Landscape ecology principles provide sufficient tools to assess at different spatial scales how urban green spaces affect and promote urban biodiversity (Clergeau et al. 2006; Breuste et al. 2008; Goddard et. 2010). Previous studies on birds revealed that increasing area of urban green space is positively correlated with species richness and diversity (Clergeau et al. 1998; Chace and Walsh 2006,

Aronson et al. 2014). Comparable patch area effects were also demonstrated for other animal taxa like amphibians, arthropods and mammals in urban environments (Parris 2006; Sadler et al. 2006; Magle et al. 2009).

When it comes to the spatial arrangement of habitat patches, animal movement between patches becomes an important factor to access resources and maintain biodiversity in disturbed environments (Jeltsch et al. 2013). In landscape ecology several metrics exist to assess landscape connectivity (Calabrese and Fagan 2004), defined as the degree to which the landscape facilitates or impedes movement among resource patches (Taylor et al. 1993). Basically there are three different ways to measure landscape connectivity: (1) structural connectivity metrics measure the simple physical structure of habitat patches (e.g. distance to the nearest neighbor), (2) potential connectivity metrics include basic or indirect information about animal movement (e.g. graph theory metrics), and (3) actual connectivity metrics are based on observed movement of organisms among patches (e.g. observed emigration, immigration, or dispersal rates) (Calabrese and Fagan 2004). Whereas potential and actual connectivity are rooted in the functional connectivity concept (Taylor et al. 1993), structural connectivity does not include any information about animal movement and just refers to structural arrangement of habitat patches (Tischendorf and Fahrig 2000).

In urban environments potential habitat patches like green spaces are fragmented (Breuste et al. 2008). Therefore, landscape connectivity of remaining habitat patches becomes crucial to understand the habitat use of different taxa in cities. An increasing number of studies have assessed the functional connectivity of urban green spaces for different taxa such as ground dwelling mammals (FitzGibbon et al. 2007; Magle et al. 2009; Braaker et al. 2014) and birds (Shanahan et al. 2011), highlighting the importance of connectivity analyses of green spaces within urban environments and its potential to effectively manage urban animal biodiversity.

Despite their ecological importance (Kunz et al. 2011; Kasso and Balakrishnan 2013), high diversity (Hutson et al. 2001; Dietz et al. 2007), worldwide conservation status (Hutson et al. 2001) and their potential as bioindicators (Fenton 2003; Jones et al. 2009), little is known about bats' activity patterns and habitat use in urban green spaces. Yet a wide variety of studies, mainly conducted in North America, Europe and Australia, implicate that urbanization is detrimental to urban bat assemblages (Kurta and Teramino 1992; Gaisler et al. 1998; Guest et al. 2002; Avila-Flores and Fenton 2005; Johnson et al. 2008; Fabianek et al. 2011; Jung and Kalko 2011; Coleman and Barclay 2012; Hale et al. 2012; Threlfall et al. 2012; Luck et al. 2013). In general species richness of bats declines with increasing levels of urban development (e.g. high density housing) and the activity of urban bat assemblages is dominated by a subset of bat species less affected by urbanization (Avila-Flores and Fenton 2005; Jung and Kalko 2011; Coleman and Barclay 2012; Threlfall et al. 2012). However, several studies have shown that species richness and/or species-specific activity increased in the suburban landscapes, characterized by low density housing, green spaces such as private gardens and/or fragments of semi natural areas (Gaisler et al. 1998; Avila-Flores and Fenton 2005; Jung and Kalko 2011; Threlfall et al. 2012; Luck et al. 2013). Furthermore, studies have shown that large urban green spaces have the potential to increase bat diversity and activity within cities (Avila-Flores and Fenton 2005; Scanlon and Petit 2008; Fabianek et al. 2011). Although these papers illustrate that urban bat assemblages may benefit from the extent of urban green spaces, there is no study directly investigating the habitat use of bats in urban green spaces and assessing if these patches represent a connected habitat network on a subregional level, the whole city. This would be a further step to identify the conservation requirements of bats in an urban environment by investigating overall habitat connectivity and the contribution of certain patches to habitat reachability/availability.

The objective of this study was to determine bat activity and patterns of habitat use in urban green spaces and to assess if green spaces represent a network that enables bats to move between habitat patches and to access resources across the city of Vienna, Austria. First, I determined bat activity

and species richness within small parks, large parks and residential areas. I expected a higher total bat activity in large parks than in small parks and residential areas (Avila-Flores and Fenton 2005; Fabianek et al. 2011) and that species-specific activity patterns would differ between habitats types (Gaisler et al. 1998). Second, I assessed the threshold distance for the connectivity analysis using two different approaches: (1) linear regression analysis using proportions of land cover for different buffers around the survey points to evaluate at which scale urban green space is a predictor for bat activity (Lookingbill et al. 2010); (2) using foraging distances of common bats in the study area based on literature. In a last step, I determined the overall connectivity, the relative importance of distinct green space patches, their contribution to overall connectivity and the effect of patch size on overall connectivity. Due to the high portion of urban green spaces in Vienna (ca. 50%, Municipal Department 23 2013) I expected high overall connectivity of urban green spaces and that with smaller threshold distance small patches become important connectors for the habitat network because they act as stepping stones between larger habitat patches. These findings can guide the management of bats in the city by identifying important habitat patches to successfully protect a highly mobile group of organisms of worldwide conservation concern.

Materials and Methods

Study area

The study was conducted in the Central European city of Vienna (48° 07' 06'' to 48° 19' 23'' North; 16° 10' 58'' to 16° 34' 43'' East) located in the Vienna Basin next to the easternmost foothills of the Alps in northeastern Austria. The urban area comprises 414.87 km² (Municipal Department 23 2013) with a population density of more than 4,200 people per km² (Statistik Austria 2013). Vienna occupies a special position within the European metropolises because of its high green space fraction which is almost 50 % (189.12 km²) of the total area (Municipal Department 23 2013). The elevation ranges from 151 m above sea level in the floodplain Lobau in the south eastern to 543 m above sea level at the Hermannskogel, a foothill of the eastern Alps situated in the north of the Vienna Woods (Municipal Department 23 2013). The predominant climate is a transition between the oceanic and humid continental. The average yearly temperature is approximately 10 °C and the annual precipitation is around 607 mm (ZAMG 2002). The Köppen−Geiger climate classification for this part of Austria is Cfb (C: warm temperate, f: fully humid, b: warm summer; Kottek et al. 2006).

Survey points

I conducted fieldwork at 106 survey points across the city of Vienna. All sampling points (**Fig. 1**) have been part of former bird census conducted by BirdLife in 2001 and 2002 (Wichmann et al. 2009). These sampling points were located in three different categories (i.e., habitat types): large parks (GP; >1.5 ha; n=40), small parks (KP; <=1.5 ha; n=27) and residential areas (EB; n=39). The large and small parks represented all kinds of urban green spaces (e.g. meadows, parks, school yards, cemeteries.). Residential areas corresponded to areas with low density housing and private gardens. When there was no access to private properties or it was not possible to install the batcorders in a safe location, the original survey point was moved as close as possible to the original location and in the same habitat

type. Few survey points (n=8) were more than 30 m away of the original habitat type, thus they were newly categorized based on their new location.

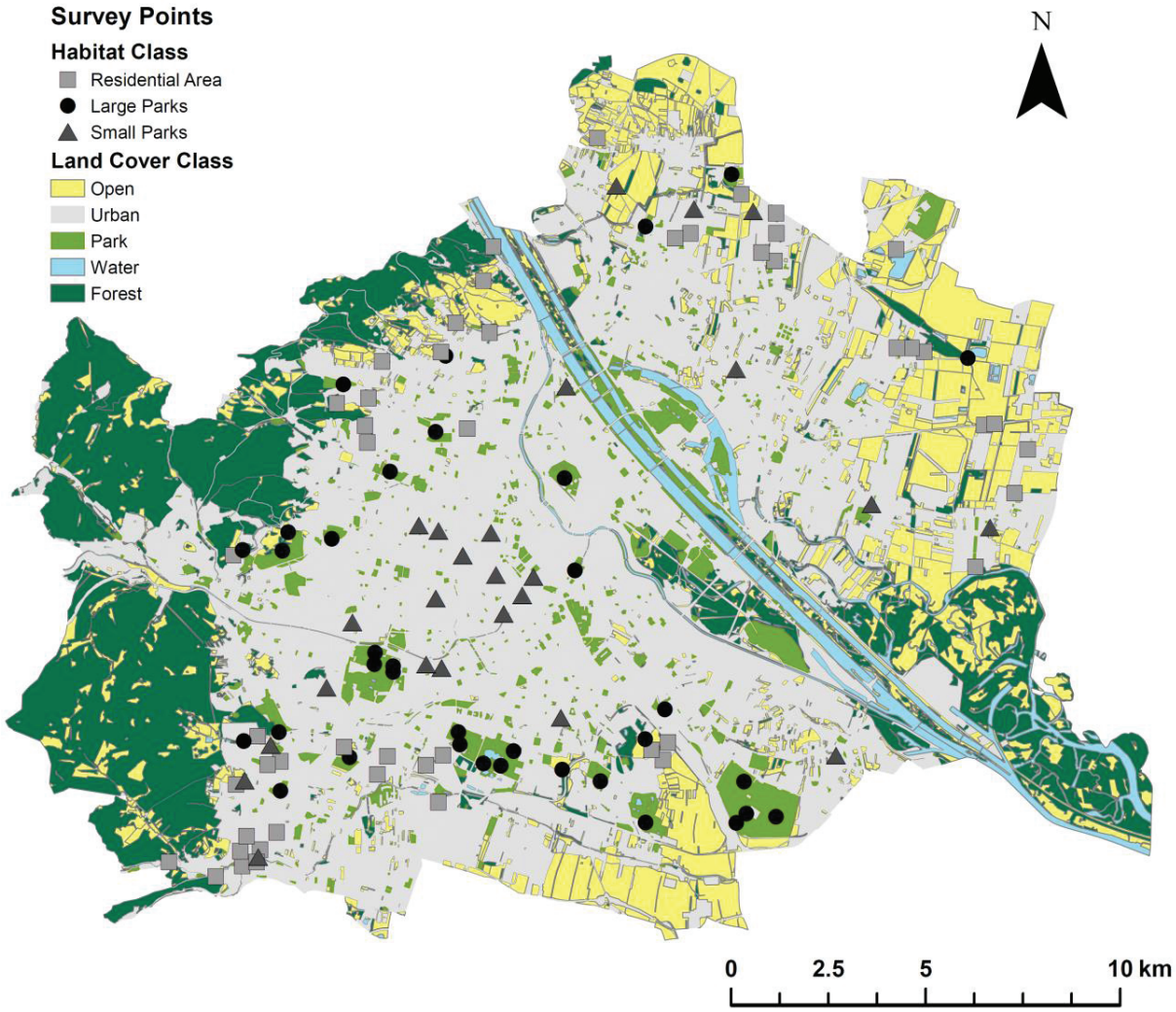

Fig. 1. Location of the 106 survey points across Vienna, Austria. The survey points were: residential areas (gray squares), large parks (black circles), and small parks (dark-gray triangles). Displayed land cover classes were derived from the combination of distinct land use classes based on the land use map of Vienna (Municipal Department 18 and 41 2009).

Bat surveys

Automated acoustic bat surveys were conducted to determine species richness, overall and species-specific bat activity per site. Study period was from June to September 2013. During this time each site was surveyed two times in auto timer mode and default settings using batcorders (ecoObs GmbH, Nürnberg, Germany). Automated bat call recordings were performed stationary 1h hour before sunset to 1h after sunrise. The advantage of such a passive acoustic monitoring over actively triggered data recordings performed by a subjective fieldworker is its higher potential as standardized method for acoustic bat surveys on a landscape scale (Stahlschmidt and Brühl 2012). All batcorders were equipped with omnidirectional electret microphones and were installed at least 2 m away from adjacent structures at a height of ca. 2.5 m to minimize echoes and overlapping bat call recordings.

Bat call identification

Automated identification software was used to identify the bat call recordings at the most specific taxonomic level possible to determine species richness and activity patterns of bats in the study area. Bat call identification was performed in three steps using separate software programs to achieve different goals. First step was the storing and measurement of the bat call characteristics (e.g. length of call, frequency range) using bcAdmin ver. 2.24 (ecoObs). Second step was the automatic classification of bat calls in which the unspecified bat call recordings were compared with reference calls (ca. 600 calls/species) using the software batIdent ver. 1.03 (ecoObs). This automatic classification is based on decision trees and computed with the statistical software R (R package 'randomForest', Liaw and Wiener 2002). Last step was a manual validation, to verify the automatic bat call classification using the sound analyzing program bcAnalyze ver. 2.0 (ecoObs) and the decision instructions of a batcorder specific identification manual (Zahn et al. 2009). This validation was an important step to detect potential misclassifications, like false positive identifications or noises misclassified as bat calls. Due to overlapping call characteristics, it was impossible to distinguish between some species. Thus, I combined

those species into groups: *Pipistrellus kuhlii* + *P. nathusii* = *P. kuhlii/nathusii* group; *Myotis mystacinus* + *M. brandtii* = *M*. bart group; and all species of the genus *Plecotus* = *Plecotus* spp. group. If present, each species group represented one count for the species richness assessment. All bat calls not classified to species or species group level were included in the total bat activity. Bat activity was defined as recording length of the verified call sequences in seconds.

Land cover data

The land use map of Vienna (Municipal Department 18 and 41 2009) created using aerial photographic interpretation of (geometrically corrected) orthophotos was the basis for the land cover classification. In the highest level of detail, this data set consists of 32 land use classes, separated into three categories: building land (18 classes), traffic (5 classes) and green spaces (9 classes). To merge the land use classes for subsequent statistical analysis I followed the CORINE land cover nomenclature/ description for Austria (Environment Agency Austria 2014) to develop five land cover classes: urban, water, forest, open and parks. Urban represented artificial surfaces (e.g. urban fabric, industrial, commercial and transport units). Water combined inland water bodies and water courses into one class. Forest comprised all forest types within the urban area. Open represented agricultural areas like arable land, vine yards and meadows. Parks were artificial non-agricultural vegetated areas like parks/green areas, cemeteries or outdoor sport facilities and recreation areas. I used satellite images (world imagery, ESRI) and OpenStreetMap (OMSF 2014) to validate landscape characteristics around each survey point, to reduce the underestimation of urban green space in further analysis. Furthermore, I edited new polygons representing green spaces not present in the land use map of Vienna.

After this classification, I used ArcMap (ESRI, Redlands, California, USA, ver. 10.3) to determine the proportion of each land cover class within five buffers: 100, 200, 300, 400, 500 m around each of the 106 survey points. These buffers were used to assess at which spatial scale the land cover classes are the strongest predictor for bat activity and therefore identify a threshold distance for the connectivity

analysis. The maximum range of 500 m was chosen to reduce the overlap between survey points. Some buffers of survey points next to the city borders exceeded the defined area of the land use map of Vienna. Therefore, I took the coverage information of the city surroundings using the Urban Atlas (European Environment Agency 2010) and combined land cover proportions with the ones of the land use map of Vienna.

Statistical analysis

To assess at which spatial scale land cover surrounding the 106 survey points was a predictor for bat activity, the following variables were chosen. The continuous response variable for the analysis was total bat activity per survey point (arithmetic mean of call sequence length/night (s)). Explanatory variables were the land cover proportions within the five buffers around each survey point and the nominal habitat type variable. Due to the high variability of response variable, I performed log 10 transformation to normalize the data and to stabilize the variance. I checked collinearity between the explanatory variables using Pearson correlation coefficient and use > 0.7 as indicator of high correlation. All statistical analyses were computed in R (R Core Team 2014, ver. 3.1.2).

After preparation of the data, I performed linear regression and used the Akaike information criterion (AIC) for model selection. The significance level for the statistical analysis was 0.05. The full model contained the log 10 transformed total bat activity and all untransformed explanatory variables. Spatial autocorrelation patterns for this model were investigated by plotting the normalized residuals against the spatial coordinates of the survey points and validated by calculating Global Moran's I values for regression residuals. No significant patterns of spatial autocorrelation were detected ($I=-0.482$; $p=0.685$). Due to the nested design of the land cover proportions around survey points, I recalculated coverage data for the five buffers (Zuur et al. 2009). The proportion measured for the 500 m radius remained the same, while coverage measured at the 400, 300, 200 and 100 m was recalculated as the difference between the original 500 m value and the one that was nested in. When the variances were

heterogeneous, I applied generalized least squares (GLS) modelling (R package 'nlme', Pinheiro et al 2014) using the 'varIdent' variance structure for the nominal explanatory variable, to allow for different variances per stratum and for the proportional explanatory variables I selected the 'varExp' variance structure due to the great number of zeros in these variables. 'varExp' represents a variance structure defined by an exponential function of the covariates for the different land cover classes. The best fitting model contained the log 10 transformed total bat activity as response variable, the recalculated coverage explanatory variables to reduce the correlation within the different land cover classes, the 'varExp' variance structure for all buffer ranges of land cover class parks and the nominal independent variable habitat type with 'varIdent' variance structure. I performed backward selection to determine the significance of predictors using likelihood ratio tests. Significant predictors for total bat activity were taken as distance threshold for the network analysis of potential connectivity of urban green spaces.

Network analysis

To assess potential connectivity of urban green spaces for bats, I used a graph theoretic approach for modelling spatial networks (Urban and Keitt 2001; Calabrese and Fagan 2004; Lookingbill et al. 2010). The strength of this framework is the relative high level of detail related to the data requirement for network analysis on a landscape scale (Calabrese and Fagan, 2004). A graph consists of two elements: the nodes and the links (Urban and Keitt 2001). Nodes represent habitat units (e.g. patches) and links correspond to the connections between nodes. A group of connected nodes represents a component, a subgraph within the network.

A critical step for assessing potential connectivity of landscapes is to determine habitat units and the threshold distance for the links. In this study, nodes were represented by urban green space and the threshold distance to decide if two green spaces were connected was the maximum range of the buffer radius at which land cover (i.e., forest, parks, open) was a significant predictor for total bat activity and was assessed in the statistical analysis section. Due to the limitation of the analysis window to a

maximum of 500 m, I additionally searched literature containing foraging distance data for common bats in the study area. Little is known about foraging distances of bats within cities, therefore I selected the average maximum foraging distance of *P. pipistrellus* from a rural study (1.5 km, Davidson-Watts and Jones 2006) as threshold distance. For the connectivity analysis I assumed undirected, binary (connected=1, unconnected=0) graphs where the possibility to move between connected patches was the same in both directions.

To measure connectivity as the amount of available and reachable habitat at the landscape scale, I calculated the Integral Index of Connectivity (IIC) (Pascual-Hortal and Saura 2006). IIC ranges from 0 to 1 and the value increases with improved connectivity. This metric is representing the interpatch (between-patch) and the intrapatch (within patch) connectivity (Pascual-Hortal and Saura 2006). Therefore the IIC also corresponds to patch attributes like patch area and was included for the network analyses of urban green spaces. Furthermore, I chose the Equivalent Connected Area for IIC (ECA(IIC)) (Saura et al. 2011). This metric is defined as the size of a single habitat patch that would provide the same IIC value than the actual habitat network in the landscape. It is expressed in the unit of the patch size (km^2). Last overall connectivity metric was the Number of Components (NC), which corresponds to the number of subgraphs within the landscape. To assess the value of distinct habitat patches on overall connectivity, I selected dIIC, defined as percentage of the variation in IIC caused by the removal of each individual element from the landscape (Saura and Rubio 2010). I used Conefor 2.6 (Saura and Torné 2009) to determine Euclidian edge to edge distances between habitat patches, to calculate the overall connectivity metrics (IIC, ECA(IIC)) on a landscape scale and to assess the relative importance (dIIC) of each patch in the network.

Results

Bat call identification

I recorded 20561 bat call sequences with a total length of 22350.66 s. This resulted in an average total bat activity of 11175.33 ± 2684.26 s (arithmetic mean ± 1 standard deviation) per survey. I detected a wide variation of average total bat activity per survey point (105.43 ± 184.91 s; n=106). The minimum of average total bat activity per survey point was 0.23 s and the maximum was 1121.35 s. More than 80% (9023.79 ± 2141.25 s; n=2) of the total average activity was classified to species or species group level. Species specific bat activity varied widely between species and species groups. Five species were dominated the total average activity: *Pipistrellus kuhlii/nathusii*, *Hypsugo savii*, *P. pygmaeus*, *Nyctalus noctula* and *P. pipistrellus*. These bats had a portion of more than 98% of classified bat activity (**Fig. 2**). More than 53% of the species specific activity was contributed by the *P. kuhlii/nathusii* group. Lowest activity of common bats was detected for *P. pipistrellus* (>6%). Remaining rare species had a fraction of 1.15% percent of species-specific activity.

In total, I identified bat calls from 12 different species and 3 species groups. Overall species richness per survey point ranged from 0 (n=3) to 8 (n=1) species (**Fig. 3**). Most survey points had a species richness of 4 (n=29). The *P. kuhlii/nathusii* group was verified for more than 90% (n=97) of the survey points, followed by *N. noctula* (n=94), *H. savii* (n=79), *P. pygmaeus* (n=69) and *P. pipistrellus* (n=59). Rare species were verified for less than 7% of the survey points: *M.* bart group (n=7), *Barbastella barbastellus* (n=6), *Eptesicus nilssonii* (n=4), *Plecotus* spp. (n=3), *E. serotinus* (n=2), *Myotis emarginatus* (n=2), *M. nattereri* (n=2), *M. daubentonii* (n=2), *N. leisleri* (n=1) and *Vespertilio murinus* (n=1).

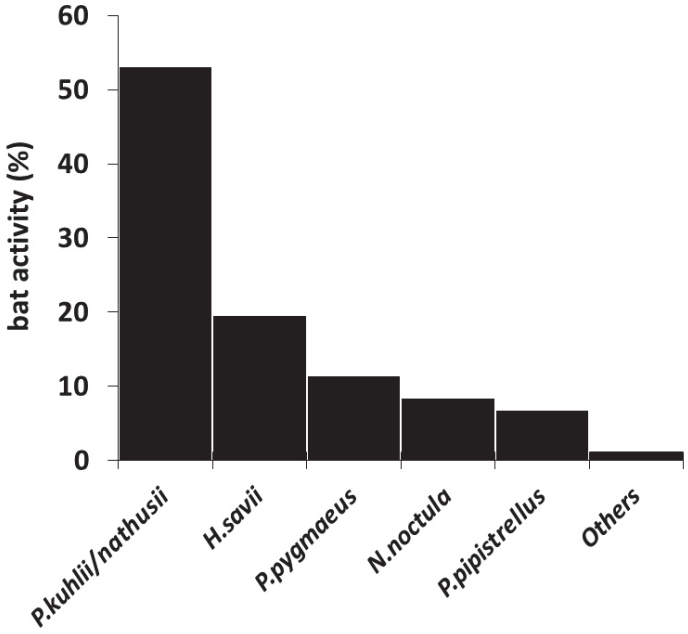

Fig. 2. Relative portion (%) of bat activity of the classified bat call sequences (9027.99 s), divided for the five common species/species group: *Pipistrellus kuhlii*/*nathusii* (4786.86 s), *Hypsugo savii* (1758.65 s), *P. pygmaeus* (1021.11 s), *Nyctalus noctula* (747.83 s), *P. pipistrellus* (609.97 s) and others i.e., total bat activity of rare species (103.56 s).

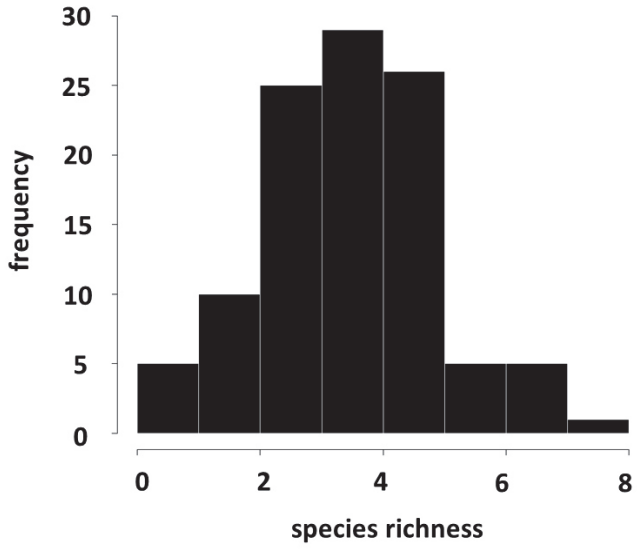

Fig. 3. Histogram of overall species richness (total counts of bat species) and the corresponding frequency of the survey points (n=106) where the number of bat species was detected.

Habitat Classes

All common bat species were present in each habitat class. The only rare species also verified for all habitat classes was *B. barbastellus*. Due to the low activity of rare species I was not able to assess habitat preferences for these bats. For the $\log_{10}$ transformed average total bat activity, a one-way ANOVA indicated no significant difference between the habitat classes (F_{103}=1.29, p= 0.28; **Fig. 4**).Similar outcome was detected for the common bat species and in general no significant differences in specific bat activity between small parks, large parks and residential areas were detected (*H. savii*: F_{103}= 0.57, p= 0.57; *P. pygmaeus*: F_{103}= 2.09, p= 0.13; *N. noctula*: F_{103}=1.12, p= 0.33; *P. pipistrellus*: F_{103}=0.02, p= 0.98). The only exception was a significant difference in activity for the *P. kuhlii/nathusii* group among habitat classes (F_{103}=3.83, p= 0.03) and a pairwise ANOVA of the habitat classes revealed a significant difference between residential areas and small parks (F_{64}=6.9, p= 0.01).

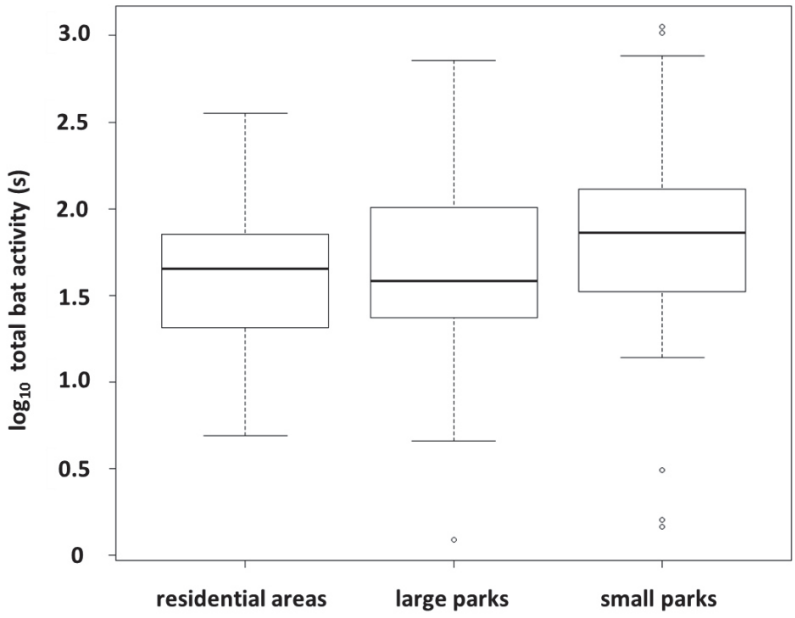

Fig. 4. Box plots represent average total bat activity (log10, s) grouped for the three habitat classes: residential areas (n=39), large parks (n=40), small parks (n=27). One-way ANOVA showed no significant differences in bat activity between habitat classes (F_{103}=1.29, p= 0.28).

Network analysis

Using the 400 m buffer size, parks were a significant predictor of total bat activity ($p < 0.0001$, $L = 19.524$). Therefore, for the first network analysis, I used parks as habitat patches and 400 m as threshold distance to link those patches. For the second network analysis, I selected the average foraging distance found for the common species *P. pipistrellus*, therefore, parks represented habitat patches and 1500 m was used as the threshold distance. Bat species with larger foraging distances were represented in the second network analysis. Given that forest was also significant in the backward selection ($p < 0.0001$, $L=19.531$), I also included forest as habitat patches and used a 1500 m threshold distance in a third network analysis.

Parks covered a total area of 25.8 km² and parks + forest an area larger than 100 km² (**Table 1**).In general, all three network analysis showed relative small overall landscape graph metrics (**Table 1**). The two graphs with parks as habitat patches and the two different threshold distances, 400 m and 1500 m, had an IIC of 0.0002 and 0.0008, respectively. The ECA(IIC) ranged between 5.8 and 11.4 km². With increasing threshold distance the NC decreased from 90 to 5 subgraphs. Highest IIC (0.0159) was obtained for the graph where the threshold distance was 1500 m and the habitat patches were represented by land cover classes parks and forest. The corresponding ECA(IIC) was 52.2 km² and all habitat patches were connected in one component (**Table 1**).

Table 1. Overall landscape graph metrics of the connectivity analysis with corresponding graph element characteristics and patch attributes. Habitat patches represent the distinct land cover classes. Threshold distance (m) is the maximum interpatch distance to link patches. Total area (km²) is the total habitat compromised by the patches and n patches the number of patches involved in the analysis. IIC= Integral Index of Connectivity, ECA(IIC) (km²)= Equivalent Connected Area for IIC, NC= Number of Components

Graph elements		Patch attributes		Overall metrics		
Habitat patches	Threshold distance (m)	Total area (km²)	n patches	IIC	ECA(IIC) (km²)	NC
parks	400	25.8	907	0.0002	5.8	90
parks	1500	25.8	907	0.0008	11.4	5
parks+forest	1500	106.6	1667	0.0159	52.2	1

The relative importance of landscape patches for potential connectivity was displayed as gradual colored dIIC for the three connectivity analyses (**Figs. 5 a, b, c)**. The highest dIIC values of the two analyses for parks with different threshold distances revealed that the same two habitat patches were the most important in terms of landscape connectivity (**Figs. 5 a, b)**. For both networks more than 850 patches (>93%) had dIIC less than 1%. However, the difference between these two threshold distance scenarios was the higher importance of some smaller green space patches for the 400 m scenario between the two large parks of high importance. Furthermore, the amount of patches with a dIIC of more than 2.51% was higher for the 400 m scenario (n=21) as for the 1500 m scenario (n=11). For the connectivity analysis with a threshold distance of 1500 m and parks and forest as patches, the highest dIIC of more than 40% was received by a large forest patch in the western outskirts of the study area (**Fig. 5 c**)**.** But also for this analysis more than 93% of the habitat patches had a dIIC less than 1%.

Another difference between the three scenarios was the correltion between the relative importance of habitat patches and the patch attribute area (**Figs. 6 a, b, c**). With increasing threshold distance the relation between dIIC and patch area indicated a higher positive correlation between these variables for the two 1500 m scenarios than for the habitat network using a 400 m threshold distance (**Figs. 6 a, b, c**). This was also verified by the higher Pearson correlation indices for the 1500 m scenarios (> 0.9). With a threshold distance of 400 m some small patches tended to have a higher dIIC and were more important connectors in this network.

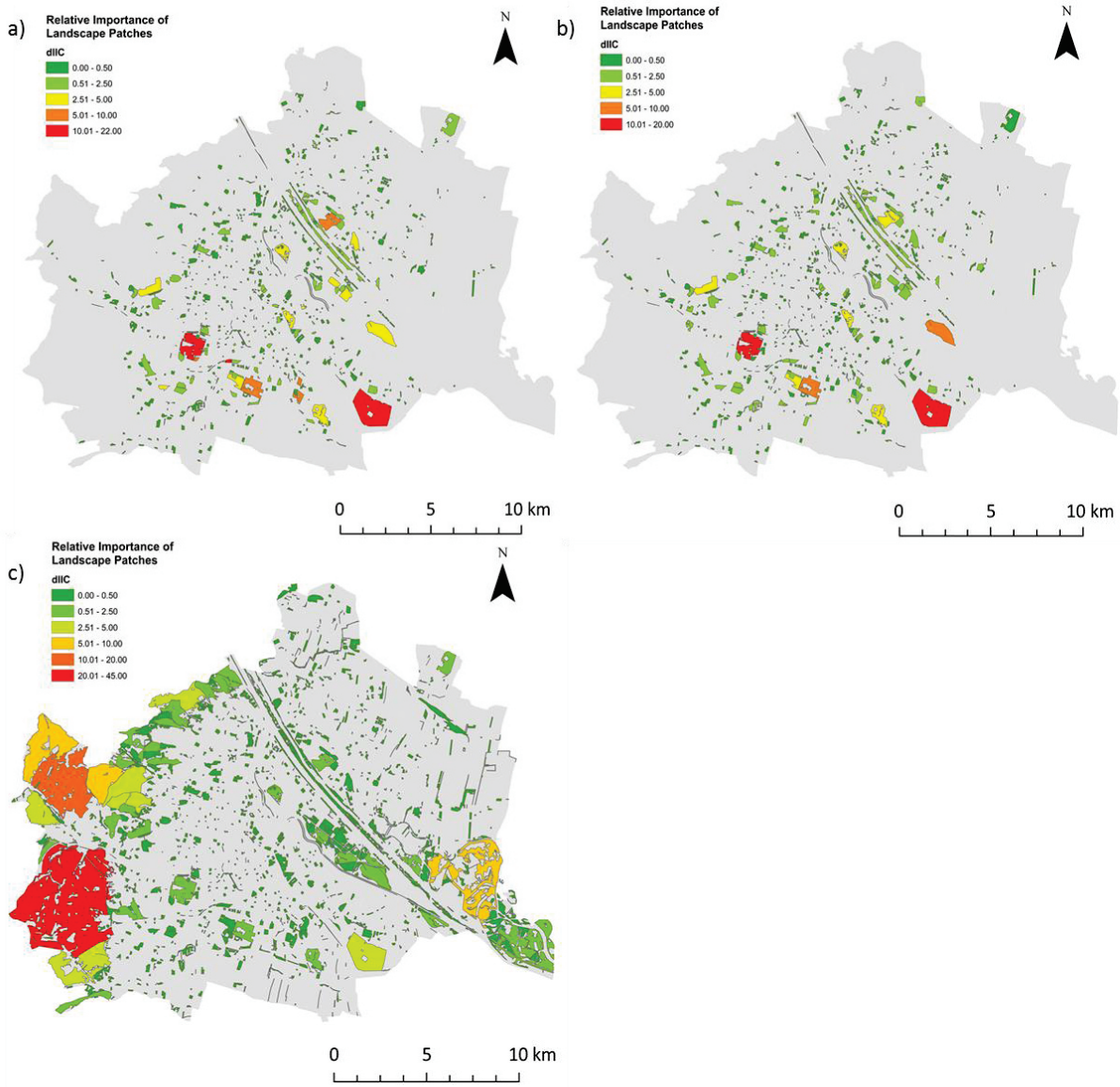

Fig. 5. Potential connectivity of urban green spaces in Vienna, Austria using different threshold distances and habitat patches: (a) 400 m, parks; (b) 1500 m, parks; (c) 1500m, parks + forest. Gradual coloration (min= dark green - max= dark red) represents the relative importance of landscape patches (dIIC), defined as the change of overall connectivity metric (IIC) on a percentage basis, if specific patches are removed.

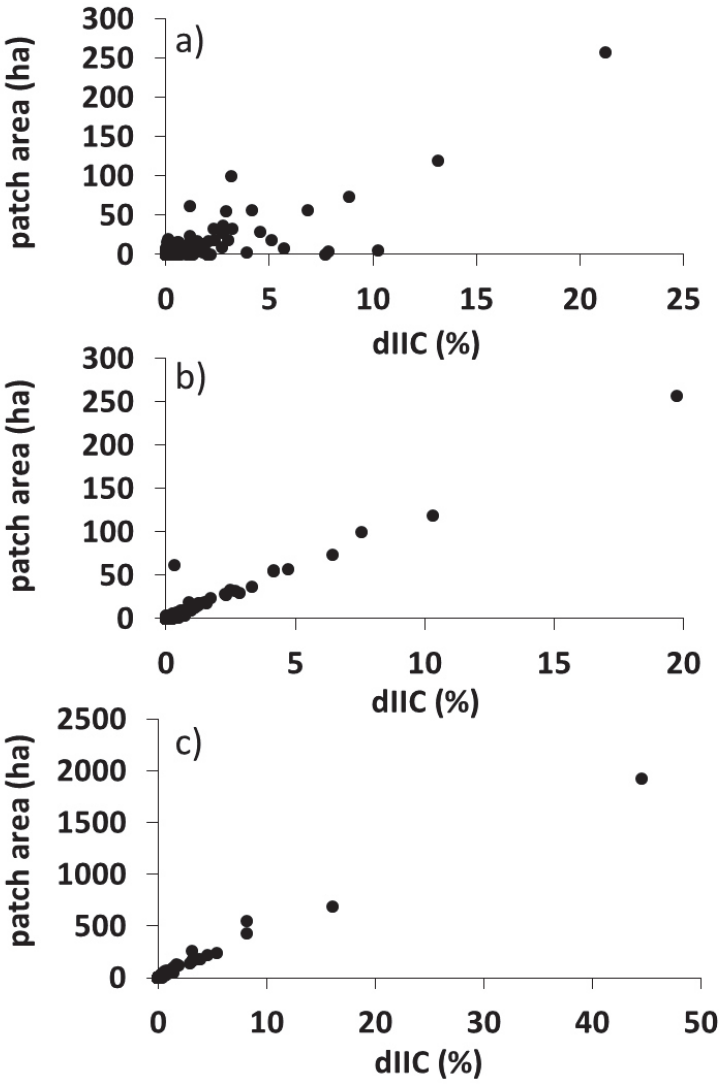

Fig. 6. Scatter plots of the relation between relative importance dIIC (%) of patches in terms of their contribution to overall connectivity and corresponding patch area (ha), divided for the three connectivity models: a) 400 m, parks; b) 1500 m, parks; c) 1500m, parks + forest. Pearson correlation indices: a) 0.82, b) 0.98, c) 0.9.

Discussion

The primary objective of this study was to determine bat activity patterns in small parks, large parks and residential areas and to assess if urban green spaces represent a habitat network for bats in Vienna using a graph theoretical approach.

In total I verified 12 bat species and three species groups in the study area. These results correspond to the findings of a recent report on the conservation status of bats in Vienna, which verified 20 species for the study area (Hüttmeier et al. 2010). There are three possible explanations why I was not able to find all bat species verified for Vienna: (1) Due to the acoustic monitoring of echolocation calls I was not able to distinguish between bats with very similar call characteristics; (2) my survey did not include forests, therefore forest dwelling species like *M*. *myotis* and *M*. *bechsteinii* were not detected; and (3) not only that some species like *Rhinolophus hipposideros* are rare, their low-intensity echolocation calls make them less detectable using acoustic monitoring. But compared to other studies on bats within European urban environments, bat assemblages tend to be quite diverse in Vienna (Gaisler et al. 1998; Guest et al. 2002; Hale et al. 2012). However, this diversity is dominated by a subset of one species group and four species (*P. kuhlii/nathusii* group, *H*. *savii*, *P*. *pygmaeus*, *N*. *noctula*, *P*. *pipistrellus*). This finding is supported by other studies on bats in urban environments, which found that some synanthropic species tend to be less affected by urbanization and therefore dominate in cities (Gaisler et al. 1998; Gehrt and Chelsvig 2004; Avila-Flores and Fenton 2005; Threlfall et al. 2012). Despite the high amount of species I identified for the study region, species-specific bat activity of rare species was too low and the variability was too high for a thorough statistical analysis identifying activity patterns between habitat classes. This variability in activity and species occurrence is often reported in studies on bats (Scanlon and Petit 2008; Skalak et al. 2012). Therefore, I focused on total bat activity as response variable for the analysis to detect differences in habitat use and additionally included the activity of common bat species. I was not able to detect a significant difference in total bat activity

among the three habitat classes (small parks, large parks and residential areas) except for the *P. kuhlii/nathusii* group. One explanation for this outcome is that total bat activity is dominated by common species and it is reported that all common bats forage in a variety habitats and are also roosting in buildings (Bihari 2004; Dietz et al. 2007; Csorba et al. 2008; Aulagnier et al. 2008, Hutson et al. 2008a,b; Hutson et al. 2015). Thus they are potentially able to exploit all three habitat classes in the same way. Furthermore, residential areas also include areas of vegetation similar to urban green spaces and previous studies have shown that residential gardens are successfully exploited by bats (Baker and Harris 2007; Hale et al. 2012). However, the results for the *P.kuhlii/nathusii* group showed a significant difference in activity between small parks and residential areas. This activity pattern indicates a higher activity of the *P.kuhlii/nathusii* group in small parks and highlights them as potential resource patches in densely built urban areas. Therefore, the high amount of small green space patches in the study area may be one explanation for the high proportion of activity of this group.

I determined overall connectivity of remaining habitat patches to evaluate the actual state of habitat availability/reachability in the study region for three different models. My results show that the connectivity concept of habitat availability/reachability can be applied in fragmented urban environments. The overall connectivity metrics of the two network models with parks as habitat patches indicate that the connectivity of these green spaces is relatively low for bats in Vienna. But due to the nature of the calculation of the IIC, these small values (IIC=<0.0009) are influenced by the size of the study area (A_L = ca. 415 km²) (Pascual-Hortal and Saura 2006). If you set the amount of reachable habitat (ECA(IIC)) in relation to the total patch area (parks=25.8 km²) it reveals that the ECA(IIC) accords to more than 22% of total patch area for the 400 m scenario and more than 44% for the 1500 m scenario, respectively. When adding forest patches to the analysis, the IIC increased from less than 0.0009 up to 0.0159. This leads to an amount of reachable habitat (ECA(IIC)) of more than 52 km², which basically corresponds to 12% of the whole study area and almost 50% of the total patch area. Thus, even if the IIC values are low, the results show that with increasing threshold distance and total patch area the relative

habitat availability/reachability is also increasing. This is also supported by decreasing NC values for the 1500m scenarios. When adding forest patches to the connectivity analysis, the habitat network represents only one single connected component and all patches are linked. However, even if the habitat network for this scenario is represented by one component, it consists of separate patches. Therefore, the IIC is still low, because the high number of links needed to move from one patch to another within a component decreased the values in the numerator of IIC (Pascual-Hortal and Saura 2006). A further aspect affecting the IIC values is the large amount of patches directly neighboring each other. These patches were analyzed as being separate from each other (**Figs. 5 a, b, c**). Thus the number of links also increased for the three scenarios and therefore resulted in lower values of the IIC.

The second approach of the network analysis is the capability to prioritize patches with regard to their relative contribution to landscape connectivity and suitability (dIIC) for the three different connectivity scenarios. My results show that for all three scenarios most patches have a relative low contribution to overall connectivity. In general more than 93% of the patches have a relative importance of less than 1%. However, the outcome for the first model (parks, 400 m) shows that with smaller threshold distance the relative importance of certain smaller patches increases and smaller patches act as stepping stones between larger patches. This corresponds to findings of other studies on landscape connectivity that small parks do not support a large amount of biodiversity but are important contributors to connectivity (Fernández-Juricic and Jokimäki 2001). For the two models with higher thresholds distance (1500 m) the high correlation indices (>0.9, Pearson) between dIIC and patch area indicate that patch area dominates spatial patch location in importance. This corresponds to the findings of Laita et al. (2011) that the IIC shows this pattern in terms of patch prioritization, meaning that with larger threshold distances small habitat patches are less important for connectivity.

Even if other studies already assessed habitat networks of green spaces within urban environments (e.g. Serret et al. 2014), my study is the first investigating the potential connectivity of

urban green spaces for bats in a city. However, there are some limitations of the network analysis. For instance, the land cover classification used a broad definition of land cover characteristics to classify urban green spaces as habitat patches and I did not account for habitat quality assuming that all green space patches of the same class are ecologically equilvalent. A solution for this limitation would be to integrate the attribute a 'habitat quality' instead of patch area for further analysis on potential habitat connectivity (Saura and Torné 2009; Decout et al. 2012). Future studies should also include local habitat characteristics to evaluate the habitat quality of certain patches for bats. Furthermore, the urban land cover class combined low density housing and highly dense urban land cover and I therefore was not able to account for the effect of residential gardens as potential green spaces and did not include their contribution to habitat connectivity of urban green spaces in the study area as shown in other studies (e.g. Rudd et al. 2002). It should also be taken into account that the outcome of the three network analyses must be interpreted differently in terms of the relevance for bat species occurring in cities. The network using a 400 m threshold distance is a multi-species approach and covers all bat species related to a threshold distances smaller than 400 m and the capability to exploit parks as habitat patches as potential foraging sites. The two scenarios using 1500 m are more species specific because the threshold distance for that model represented the foraging distance of *P. pipistrellus*.

Taken together, my findings demonstrate that the potential connectivity of urban green spaces for common bats is quite low in the study area and affected by the chosen threshold distance, the amount, size, and spatial position of habitat patches within networks. For graphs with smaller threshold distances (i.e. 400 m) spatial patch location within habitat networks becomes more important to prioritize patches in terms of their contribution to overall connectivity, because they act as important stepping stones to connect larger patches. Furthermore, I showed that there is no significant difference in total bat activity between small parks, large parks and residential areas. These findings are an important step for bat conservation in urban environments and can guide more detailed studies such as network analysis of roosts (Rhodes et al. 2006) or the assessment if functional connectivity is a predictor

for bat activity within cities like shown for birds (Shanahan et al. 2011), highlighting the importance of connectivity analysis to investigate bats habitat use in urban environments. This study showed that even general municipal information such as land use maps, combined with actual bat activity data can be applied for large scale analysis of habitat networks to identify important habitat patches which enable bats to access resources throughout the city.

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