From genes to habitats – effects of urbanisation and urban areas on biodiversity

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SUMMARY

Urban areas are landscapes dominated by built-up structures for human use. Nevertheless, nature can still be found within these areas. Urban ecosystems can offer ecological niches, sometimes only found in cities. This biodiversity in the form of genetic diversity, species diversity and habitat diversity provided the structure of this thesis.

First, we studied the effects of urbanisation on genetic diversity. We analysed the population structure of the wall lizard with highly variable genetic markers. We sampled over 200 wall lizards from nine sites with different degrees of isolation. High genetic differentiation was found between all investigated sites. The two sites most isolated showed the lowest allelic richness and the lowest observed heterozygosity. These results were combined with a GIS model to identify relevant factors of the connectivity between sites. The geographic model, based on cost-distances, showed that the dispersal on railway tracks was best correlated with the genetic data. We can therefore conclude that railway tracks provide important corridor functions for the wall lizard in the region of Basel. The allelic richness of the populations in Basel and Jura corresponded to the allelic richness of northern European populations, whereas the Mediterranean populations were allelic richer.

Second, we investigated species diversity in urban forests in the city and the surroundings of Basel. We compared the arthropod diversity, abundance and assemblages in forest patches of different sizes. Over a period of six months, we run 45 pitfall traps on nine sites and analysed three taxa: spiders, ground beetles and rove beetles. Three different methods were tested to estimate the similarity of arthropod assemblages considering under sampling. The species number of small urban forest patches did not differ significantly from large urban forests. The species assemblages however changed from smaller forest patches to larger patches. The occurring spider and ground beetle species were predominantly forest species regardless of the habitat size. In contrast, the rove beetle assemblages were not dominated by forest species. This study

demonstrates that urban groves can contribute considerably to the species diversity in an urban area. On two urban forest patches, *Leistus fulvibarbis* was found. *L. fulvibarbis* is a ground beetle, with an Atlantic-European distribution. This species has been absent from the Swiss Fauna for more than 100 years. The distribution map based on a literature review suggests that *L. fulvibarbis* is dispersing southwards since the early 90ties. The dispersion follows most likely the Rhine and its confluents.

And third, we analysed the habitat diversity, which can develop on disused railway sites. Ecologically, these sites are similar to large gravel river banks and therefore offer important habitats for threatened pioneer species. Yet, disused railway sites are of great economical interest because their reclamation costs are low and they are often located near the city centre. In an interdisciplinary study, we compared five urban development projects on disused railway sites in Europe. We identified three strategies to protect the natural sites in such railway brownfields: (1) protection of the pioneer habitats in-situ, (2) reinstallation of similar habitats on roofs (ex-situ) and (3) safeguarding of the natural process of succession. The comparison of the five projects illustrated that the current legislation varied considerably and the images of open green space differed between the stakeholders. As a consequence, the quantity and type of green space allocated changed. For future brownfield redevelopment projects, we encourage guidelines that consider the special kind of nature on such sites and guarantee planning reliability for investors.

The multiscale approach to study the effects of urban areas and urbanisation on biodiversity provided valuable results. The main effects of urbanisation and urban areas investigated in this study were habitat alteration, isolation and loss. Moderate habitat alteration had no effect on the species diversity. Moreover, typical forest species were still present in urban forests. To prevent habitat isolation, the habitat connectivity by railway tracks was important to maintain genetic diversity. Lastly, habitat loss was reduced with innovative conservation strategies and the involvement of all stakeholders. Overall, biodiversity in the city can be promoted at all levels from genes to habitats by a sustainable, scientifically based management.

GENERAL INTRODUCTION

Biodiversity in the city

Biodiversity in urban areas is important for two reasons: First, the experience of biodiversity in the form of fauna and flora shapes the perception of people who live and work in cities. The opportunities to exchange meaningful interactions with the natural world is important to gain public support for biodiversity conservation (Miller 2005). Furthermore, these encounters with nature in cities can enhance human well-being; for example by reducing the self-reported experience of stress (Grahn & Stigsdotter 2003) or by a lower probability of obesity (Nielsen & Hansen 2007).

Second, cities offer a wide potential of ecological niches and a high diversity of habitats compared to the intensively cultivated agricultural landscape (Rebele 1994). Therefore, the diversity of habitats can lead to high species richness within cities. In several studies the species numbers of plants were higher within the urban area than in the surrounding landscape (Kühn et al. 2004, Landolt 2001, Zerbe et al. 2004).

Biodiversity is defined as the variability among living organisms from all sources including, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems (UNEP 1992).

Consequently, the Swiss law requires ecological compensation areas not only in the agricultural land but also in urban settlements (Nature and Cultural Heritage Protection Act (NHG) Art. 18b Abs.2 and Nature and Cultural Heritage Protection Ordinance (NHV) Art. 15). On agricultural land, the aim is to attain 15 % of ecological compensation areas by the year 2020 (Stremlow et al. 2003). This aim is controlled by the Swiss national and cantonal governments. Farmers are financially rewarded if they implement the ecological compensation areas according to the guidelines of the government. Contrary to agricultural land, in urban areas the implementation of ecological compensation areas is neither standardised nor controlled (Eigenmann 2003). However, the fast extension of urban settlements in Switzerland gives the implementation of Article 18b an increasing priority.

Urban areas and urbanisation

Urbanisation can be defined as the growth of urban area in size and the numbers of people inhabiting urban areas. This process generates landscapes dominated by built-up structures for human use (Grimm et al. 2000). For the first time in history, in the year 2008 half of the world's population will live in cities (3.3 billion people). For the year 2030 the number of people living in cities is predicted to rise as high as 5 billion while the rural population is stagnating (United Nations 2006).

Urban areas consist of a high human population density and highly developed areas with business, industrial and residential components. The use of the term '**urban**' in a statistical context is defined by each country's government. Switzerland defines urban areas as connected built-up areas with 20'000 or more inhabitants, normally consisting of one central commune and its adjacent communes (Hotz et al. 2005).

The high increase of people living in urban areas is mainly due to processes in less developed regions (Figure 1). The main reason for the increase is the high population growth is the migration from rural to urban areas and the transformation of rural areas to urban areas. The proportion of people living in cities has risen from 18 % in the year 1950 to 43 % in the year 2005 and is predicted to reach 56 % in the year 2030.

In more developed regions urbanisation is already high: In Europe 72 % of the population is living in urban areas and this proportion is predicted to rise to 81 % in the year 2030. Here, the main process of urbanisation is the conversion of rural to urban land. In Switzerland for example, every second, 1 m^2 rural land is converted for settlements purpose (Jordi & BFS 2001). Half of the new settlement areas consist of buildings for residential purpose and almost one fifth are new transportation infrastructure. However, included in this land conversion are six percent attributed to recreational areas and cemeteries.

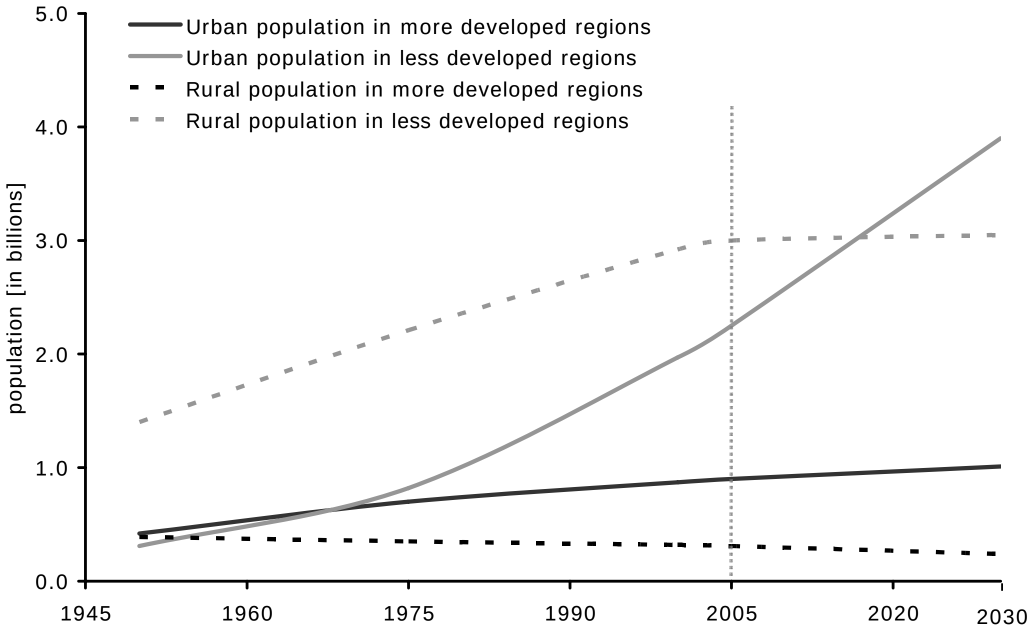

Figure 1. Urban and rural population of more developed regions and less developed regions, from the year 1945 to 2030. Figure adapted from United Nations (2006).

Effects of urbanisation and urban areas on biodiversity

Urban areas and urbanisation affect biodiversity in many ways. One of the most important consequence of urbanisation is habitat loss, habitat fragmentation and isolation or habitat alteration (Antrop 2000, McDonnell & Pickett 1990, McKinney 2002). This processes are characteristic for urban areas and lead to patchiness of urban habitats and poor connectivity among them (Trepl 1995). The patchiness of urban green areas makes dispersal a central theme of urban ecology, at least for taxa with poor dispersal ability. If dispersal and migration are reduced, the risk of inbreeding increases and the chances of survival of these species are at risk. Consequently, the extent of green areas and their connectivity is an important factor affecting species occurrence and survival in urban landscapes (Niemelä 1999b).

Another characteristic of urban areas is the high number of exotic, invasive or accidentally introduced species (Rebele 1994). If these non-native species replace the native species, they reduce biological uniqueness of a local ecosystem. McKinney and Lockwood (1999) define this process as homogenisation. The amplitude of this process and if it is a worldwide phenomena, is widely discussed in urban ecology (Kühn & Klotz 2006, McKinney 2006, Olden et al. 2006, Smith 2006). Alternatively, urban areas can be naturally species rich, because of their geographic situation (Araujo 2003, Kühn et al. 2004) or because of their high habitat heterogeneity (Klotz et al. 1984, Kowarik 1992, Pickett et al. 2001, Sukopp et al. 1993).

A further feature, typical for many urban habitats, is their early successional stage, which is maintained by disturbance (Niemelä 1999b). Examples are regularly mowed lawns of parks, regular cuts of undergrowth in forest, or extensive maintenance of railway tracks to prevent overgrowth by weed. In addition to this planned disturbance by maintenance, most of the urban habitats encounter disturbance through a high visitor pressure (Gibb & Hochuli 2002) or simply by the higher levels of pollution and noise (McDonnell et al. 1997, Müller 1977).

And finally, a distinct abiotic difference between urban areas and rural areas is the higher temperatures (+2°–3°C), especially during the night. This is the so called urban heat island effect (Arnfield 2003, Oke 1974, Sukopp et al. 1993).

Hierarchical approach for biodiversity in the city

Biodiversity has a multiscale content and can be measured on different levels going from genes, species to habitats and landscapes (Noss 1990, Raven 1992). Savard et al. (2000) recommend such a hierarchical approach for investigating urban ecosystems. In the following paragraphs of this introduction, we will discuss the first three levels of biodiversity (genes, species and habitat). For the discussion on the level of genes and species, we focus on the faunal aspects.

GENETIC DIVERSITY

The measuring of genetic diversity is only recently a theme in urban ecosystems (Shochat et al. 2006). The main focus of these studies have been species that are a health-threads to humans (Bradley & Altizer 2007): e.g. the increasing abundance of red foxes (*Vulpes vulpes*) in the city and the potential increase of infections with the tapeworm (*Alveolar* *echinococcoses*) has lead to a study on the genetic structure of the city fox in urban habitats (Wandeler et al. 2003).

One key feature of natural habitats in cities is their fragmentation and isolation. Because small and isolated populations risk loss of genetic diversity and therefore reducing their reproductive fitness and the ability to adapt to environmental changes (Frankham 1996), urban species are especially useful to study habitat fragmentation. Wood & Pullin (2002) compared the dispersal abilities of four different grassland butterflies and their persistence in a fragmented urban environment. Their results suggest that the butterfly species were more limited by the availability of a suitable habitat than by their ability to move among habitat patches. The genetic diversity can be measured with the level of heterozygosity and the numbers of alleles existing in a population.

SPECIES DIVERSITY

Most studies on biodiversity in the city have studied species diversity (Shochat et al. 2006). These studies were initiated by the discovery of high species richness within urban areas and focused on patterns of species abundance and diversity (Klausnitzer 1987, Sukopp 1983). The main approach to investigate these patterns for the urban fauna followed McDonnell & Pickett (1990), who suggested to investigate the relation of urbanisation and species diversity along an urban-rural gradient. Results from Clergeau (2006) showed that species richness of birds within the same type of habitat was unaffected by the degree of urbanisation, whereas the species richness of mammals decreased within one habitat type. If the examined sites followed the urban-rural gradient regardless of the habitat type, the number of mammal species and bird species declined with a higher degree of urbanisation, as described also by Sukopp et al. (1993).

Studies on species diversity of arthropods in urban environments were conducted mostly within one habitat type, often in urban forests. The results of a literature review are presented in table 1. An increase of species number from urban to rural habitats is reported four times; no differences in species numbers among urban, suburban and rural sites are reported five times.

Table 1. Literature review on species numbers of three arthropod taxa (spiders, ground beetles and rove beetles) in forest patches along an urban-rural gradient (u=urban, s=suburban, r=rural). Trends: species numbers increasing (↗), when the trend reported in the cited paper was significant on a 5 % level.

Reference	City	Taxa	Species number			
			u	s	r	Trend
Alaruikka et al. (2002)	Helsinki	Spiders	59	55	56	⇔
Alaruikka et al. (2002)	Helsinki	Ground beetles	18	24	17	⇔
Deichsel (2006)	Berlin	Ground beetles	17	13	14	⇔
Deichsel (2006)	Berlin	Rove beetles	29	25	30	⇔
Niemelä et al. (2002)	Edmonton	Ground beetles	21	24	28	↗
Niemelä et al. (2002)	Helsinki	Ground beetles	18	16	21	↗
Niemelä et al. (2002)	Sofia	Ground beetles	44	36	45	⇔
Ishitani et al. (2003)	Hiroshima	Ground beetles	13	21	23	↗
Weller & Ganzhorn (2004)	Hamburg	Ground beetles	7	10	14	↗

Another approach to asses species diversity in urban habitats, relates habitat size to species diversity. The underlying hypothesis for this approach is the theory of island biogeography (MacArthur & Wilson 2001), where more species are expected in larger habitat patches. Such habitat patches are regarded as islands in a hostile urban matrix. This approach was followed in several studies (Cornelis & Hermy 2004, Fernández-Juricic 2004, Gibb & Hochuli 2002, Godefroid & Koedam 2003, Guirado et al. 2007, Tilghman 1987). However, only half of these studies could confirm the hypothesis. This inconsistency can be explained by the fact that the matrix around urban habitats is not so hostile as an ocean around a island would be (Pickett et al. 2001).

HABITAT DIVERSITY

The establishment of urban ecology in Europe and particularly in Germany was accompanied by biotope mapping schemes in many cities (Schulte et al. 1993, Wächter 2003). There are several approaches to classify urban habitats, but probably the most wide spread classification follows Sukopp et al. (1993). In this classification urban habitats are categorised into eleven main groups: built-up areas of three densities, open green spaces, waterbodies, transportation installations, agricultural areas, forests, quarries or landfills, special natural relict sites and wastelands. After Kowarik (2005), we can categorises open space dominated by vegetation in urban areas into four types: (1) pristine ecosystems, like old-growth forest, (2) ecosystems shaped by silviculture or agriculture, (3) ecosystems established by urban greening like parks or boulevards and (4) ecosystems, which evolved on urban-industrial sites.

In the last few years the attention of urban planners and urban ecologists have turned to the nature of type 4 (De Sousa 2003, Kowarik & Körner 2005). These urban-industrial sites can be described as brownfields and are characterised by their former industrial use (Oliver et al. 2004). In particular, disused railway sites are economically attractive for urban redevelopment projects because the costs of decontamination are low and the sites are often near to city centres (Valda et al. 2004). From an ecological point of view, these sites are a valuable source of biodiversity in the city (Harrison & Davies 2002). As long as there is some extensive maintenance, for example in form of regular mowing, these sites are kept at an early state of succession. In this state, they offer secondary habitats to many pioneer species. If maintenance falls away, such sites will go through different phases of succession and will eventually develop into forests.

The interest in such sites by urban developers and by conservation experts is a relatively new phenomenon. There are not yet fixed rules, how to deal with this kind of nature. The protection of these habitats is therefore dependent on the current politic, the legislation and the stakeholders. Within the group of stakeholders, conservation experts play a crucial role to enhance the understanding of these sites. To find sustainable solutions, an interdisciplinary approach is necessary.

DESCRIPTION OF THE MAIN STUDY SITE BASEL AND ITS HABITATS

Geographically Basel lays in the north west of Switzerland on the border triangle with Germany and France. The city is placed at the intersection of two landscapes: the relatively narrow valley of the High-Rhine ends here and the Upper Rhine rift valley begins after the sharp bend of the Rhine northwards. The rift valley is a climatically favoured region: The mean annual temperature in Basel is 9.6°C, which is the highest mean annual temperature north of the Alps except from Geneva (9.8°C). The number of frost nights is 71 days per year; thus the fifth lowest value for Switzerland. And third, only 121 days with precipitation greater than > 0.9 mm are registered, compared to a mean of 132 days with precipitation (> 0.9mm) for Switzerland (MeteoSchweiz 2005).

Basel is the third largest city of Switzerland with 163’930 inhabitants (Table 2). With 68 inhabitants per ha Basel is more densely populated than Zurich. However, the highest population density in Switzerland is recorded for Geneva (Schweizerischer Städteverband 2007). A key-factor for the attractiveness of a city is the availability of green space for recreation and leisure (Van Herzele & Wiedemann 2003). In Basel and Geneva the recreational area per inhabitant is 12 m^2 (Hotz et al. 2005). This value is only half as high as in comparable cities of the same size like Berne or Lausanne. The low values for Basel and Geneva can be explained by the high density of inhabitants, the border situation and the political situation as ‘city-cantons’. Because of the shortage of green space in Basel compared to other cities in Switzerland, the preservation and increase of urban green spaces is of political importance and has been fixed in the political goals for the canton Basel-Stadt (Staatskanzlei Basel-Stadt 2003). To improve the green space situation, the canton Basel-Stadt has implemented a pool. This pool receives 50 % of the benefits of a private investor, when the change of the land use category by the cantonal authorities brings a gain for the investor. The money in this pool is used to create new urban green space or to improve existing urban green space.

Table 2. Cities in Switzerland with more than 100'000 inhabitants and their corresponding population density and the amount of recreational area per inhabitant (Hotz et al. 2005, Schweizerischer Städteverband 2007).

	Inhabitants [year 2005]	Population density [ha^{-1}]	Recreational area per inhabitant [m^2]
Basel	163’930	68	13
Berne	122’178	24	25
Geneva	178’722	112	12
Lausanne	117’388	28	27
Zurich	347’517	40	21

Due to its political, geographical and climatic situation Basel has always been an important trading site. To enable larger ship cargo on the Rhine and to prevent flooding in the plains of the Upper Rhine rift valley, the regulation of the Upper-Rhine started in 1817 und ended around 1872 (Allgöwer 2000). Since this large river regulation the natural flood plains of the Rhine have disappeared. Short after the completion of the Rhine regulation the railway system, another important transportation system, has been constructed. These newly built railways systems act as an important refuge for the fauna and flora originally associated with the flood plains and their large gravel river banks (Burckhardt et al. 2003). The habitats developing on these railway sites are thus a typical example for the fourth type of nature found in urban areas (Kowarik & Körner 2005). Because of changes in the logistic of railway goods in the last 20 years, the extensively managed railway areas for goods transports are commercially underused or disused. This development takes place in many European cities with large railway infrastructure for goods transportation.

Other important habitats in Basel are its urban forests and urban parks. Urban forests share unique features in relation to suburban and rural forests, including air pollution, disturbance intensity and the presence of exotic species (McDonnell et al. 1997, Rebele 1994). The urban forests in Basel represent the oldest urban ecosystem and are often found along dominant landscape features, like gorges or landscape terraces. The urban parks are found throughout the city, the oldest one dating from the mid 19th century. Others have been established only in the last few years to meet the citizens' demand for urban green space.

INTERDISCIPLINARITY IN URBAN ECOLOGY

Interdisciplinarity is essential to study urban ecology (Konijnendijk et al. 2006, McIntyre et al. 2000, Niemelä 1999a), because humans live in cities and their behaviour and decisions influence all processes in an urban environment. Furthermore, only if the residents understand the values of urban nature they accept the measures to protect it (Breuste 2004).

The integration of social and natural science is therefore a promising approach to gain more holistic knowledge on urban environments. Both fields of research have their tradition in urban ecology: In North-America urban ecology has been investigated from the beginning from a sociological point of view, whereas in Europe the natural science approach has the longer tradition (Sukopp et al. 1993).

OBJECTIVES AND OUTLINE OF THE THESIS

The overall goal of this thesis was to enhance scientific understanding of biodiversity in urban areas. In this large field of research, we chose a hierarchical approach: (1) genetic diversity, (2) species diversity and (3) habitat diversity. These three thematic fields correspond to the chapters I to III of the thesis. Within these three chapters, we present four articles which can be read independently. The four articles each deal with a different key aspect within the field urban ecology.

In chapter I, we focus on the genetic diversity of the wall lizards (*Podarcis muralis*). We present the results of the genetic analyse of the wall lizard from eight sites within the city of Basel and one control group 37 km away from Basel. The genetic analyses were based on DNA-samples extracted from 209 individuals. The individuals were genotyped with 6 microsatellite loci. We analyse the effect of habitat connectivity between the investigated sites and we report the possible factors explaining the observed population structures. Furthermore, we compare the allelic richness of our populations with other populations in Europe.

In the first part of chapter II, we analyse the species diversity in urban forest patches in relation to their size. Herby, we are particularly interested in taxa that are not directly influenced by management practise. We used the following taxa: spiders, ground beetles and rove beetles. Nine sites in urban forests were investigated with pitfall traps from April to October 2004. We analyse the differences in species richness, faunal similarity and abundances within five ecological groups between small and large urban forest patches. We compare methods for describing arthropod assemblages taking into account varying sampling effort and we identify factors influencing the arthropod assemblages.

The second part of chapter II describes a rediscovered ground beetle species. The ground beetle *Leistus fulvibarbis* Dejean has not been recorded for over 100 years in Switzerland. We illustrate the finding situation and discuss possible patterns of the recent dispersal for this species.

In chapter III, we discuss the habitat diversity on disused railway sites and the consequences of redevelopment projects on these habitats. We choose an interdisciplinary approach to analyse the effects of urbanisation on disused railway sites. We describe five development projects on railway brownfields in Europe and compare the quantity and quality of open and green space. We sketch the different views of three stakeholder groups on green space and discuss the effect of legislation on the implementation of nature priority areas on railway brownfields. In a final step, we present three strategies for the protection of habitats, which developed on underused or disused railway sites.

Finally, a general discussion with conclusions from this thesis work is presented. In this last part, we link the described effects of urbanisation and urban areas on biodiversity with the findings of our studies and discuss further perspectives.

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Population genetic structure of the wall lizard (*Podarcis muralis*) in an urban environment

Manuscript

Abstract

A key feature of urban habitats is their high degree of fragmentation and isolation. To assess the functional connectivity of habitat patches in an urban area, we analysed the population structure of the wall lizard (*Podarcis muralis*). 209 individuals were genotyped with six microsatellite loci. The subpopulations on sites connected to railway tracks were closer related than subpopulations on sites not connected to railway tracks. The two sites not connected to railway tracks also showed the lowest allelic richness and the lowest observed heterozygosity. The values of genetic differentiation were correlated with a geographic model based on cost-distances to identify relevant factors of dispersal. The model scenario, where the dispersal was facilitated on railway tracks, explained 46 % of the genetic differentiation. The isolation by distance scenario and two other scenarios were not significant. These results suggest that railway tracks in the region of Basel are important routes of dispersal for *Podarcis muralis*. If preservation of genetic diversity is a priority for species conservation, then the important corridor function of railway tracks should be considered in the green space management of cities. The allelic richness of the wall lizard population in Basel and Jura corresponded to the allelic richness of northern European populations; whereas populations from the Mediterranean region were allelically richer. This decline of genetic diversity of wall lizard populations from South to North could be explained by postglacial dispersal dynamics.

Keywords: *Podarcis muralis*, population structure, dispersal, connectivity, railway tracks, cost-distance model, microsatellite

Introduction

The expansion of cities and towns has a significant effect on natural landscapes. The conversion of rural and natural land leads to human modified urban landscapes dominated by built-up structures (Grimm et al. 2000). From an ecological point of view, urbanisation has mainly adverse effects on biotic communities, but there are also favourable impacts. A favourable effect of urbanisation is the variety of habitats, created and maintained by human influence that does not occur elsewhere. This richness of habitats often supports a high species diversity even including threatened species (Niemelä 1999).

One of the most important adverse effect of urbanisation is the destruction, fragmentation and isolation of many natural habitats (Antrop 2000). As a consequence, populations on these habitats become smaller and more isolated and therefore risk the loss of genetic diversity and eventually extinction. Furthermore, with the reduction of genetic diversity, the reproductive fitness and the ability to adapt to environmental changes is reduced (Frankham 1996). Corridors or stepping stones between habitats can mitigate such isolation and can enhance the connectivity of habitats (Anderson & Jenkins 2006, Bennett 1999). Habitat connectivity is defined as a functional linkage between habitat patches for a given species and is therefore a species-specific entity (Lindenmayer & Fischer 2007).

The analysis of the genetic population structure of urban species could thus reveal the degree of isolation between individuals caused by urbanisation in the last centuries. However, until now only few studies on population structure have been conducted within urban areas. The main focus of these studies have been species that pose a health-threat to humans (Bradley & Altizer 2007): e.g. the increasing abundance of the red fox (*Vulpes vulpes*) in the city and the potential increase of infections with the tapeworm (*Alveolar echinococcoses*) has led to a study on the genetic structure of the city fox in urban habitats (Wandeler et al. 2003). Studies on other groups, which might better inform on aspects of fragmentation and isolation, are not as well known.

An example for a protected species occurring more often in urban areas than in the surrounding rural environment is the wall lizard (*Podarcis* *muralis* (Laurenti, 1768)). Because of the heat island effect in urban areas (Arnfield 2003), the climatic situation of cities in Middle-Europe favours the occurrence of the wall lizard in urban areas (Guisan & Hofer 2003). In addition, the wall lizard uses a variety of habitats closely connected to human cultivation such as railway tracks and its ballast, ancient ruins or vineyards with dry stone walls. These habitats are substitutes for the wall lizard’s primary habitat on gravel banks of large river systems (Bender et al. 1996). Its distribution in Europe reaches from Spain in the west to Romania in the east, from The Netherlands in the North to the Peloponnese in the South (Gasc 2004, Gruschwitz & Böhme 1986). In some Middle European countries the wall lizard figures on the Red List as threatened species or is protected by national law (Böhme et al. 2005). In Switzerland the wall lizard was classified as a threatened species until recently (Duelli 1994). In the current Red List however, only the population in the North-East of Switzerland are classified as threatened (Monney et al. 2005), yet the species is still protected by national law (NHV Art. 20 Abs. 2 Annex 3) (1991). In North America, on the other hand, *Podarcis muralis* is suspected to be an invasive species (Allan et al. 2006, Deichsel & Gist 2001). For this reason, a more precise knowledge of migration and dispersal of this species is very important.

For amphibians and reptiles occurring outside of urban environments, comparable studies have assessed the geographic and genetic correlations between populations. Most studies have been based on amphibians (Andersen et al. 2004, Arens et al. 2006, Newman & Squire 2001, Rowe & Beebee 2007, Stevens et al. 2006) and these studies showed significant genetic differences at a small geographical scale. For most amphibians such heterogeneous patterns in populations probably reflects their poor dispersal ability over long distances. The question of dispersal and migration in a fragmented landscape was addressed for lizards in two experimental studies by Boudjemadi et al. (1999a) and Lecomte (2004). They found that corridor effectiveness was dependent on habitat quality for the common lizard (*Lacerta vivipara*) and that the absence of connection between habitats destabilizes population functioning.

In this study, we analyse the population structure and the connectivity of habitats for the wall lizard in an urban environment. The wall lizard is a suitable study species because it occurs on habitat patches, which show differing degrees of isolation within the city. Some habitat patches are found isolated in allotment gardens, some are found on the extensive railway network of the city and others are found along river banks. River banks and railway tracks are regarded as the two main dispersal corridors for wall lizards (Bender et al. 1996). This study should give us therefore evidence for assessing appropriate management strategies to increase or to control lizard populations. This knowledge is especially important for the wall lizard because of its threatened situation in some areas in Middle Europe, but also because of its suspected invasive status in North-America.

To analyse the population structure, we choose a molecular approach based on the analysis of microsatellites. The data describing the genetic population structure was compared with their geographic situation based on a GIS-model. The aims of the study were (1) to describe the population structure of *Podarcis muralis* in an urban environment, (2) to identify the most probable dispersal route and connectivity between the sites in the city of Basel and (3) to compare the allelic richness of the investigated populations in the canton of Basel and in the canton of Jura, with data from European populations provided by the work of Gassert (2005).

Methods

STUDY AREA

The region of Basel is one of the warmest and driest areas in Switzerland (Kirchhofer 1982) and therefore offers a suitable environment for *Podarcis muralis*. In the city of Basel (N 47°33'6", E 7°35'12") the wall lizard is found locally on a variety of secondary habitats. We identified four sites on the left side of the Rhine and four sites on the right side of the Rhine (Figure 1). The four sites on each side of the Rhine were defined by the four habitat types: railway tracks, river bank, cemetery, and allotment garden. The control group was located 37 km away from Basel, in St-Ursanne (N 47°21'52'', E 7°9'19'') in the canton of Jura (JU). The habitat on this site consisted of used and unused railway structures and a disused quarry. Most of the sites were connected by railway tracks – either still used or disused. Two sites, one at the right river bank and one on an allotment garden on the left side of the Rhine were not directly connected to railway tracks (Table 1).

Table 1. Studied wall lizard habitats and genetic variation in the studied subpopulations. Railway access: sites were considered connected to the railway network if railway tracks arrived at the site or run along the border of the site. A, Allelic richness based on a minimum of 12 individuals, H_0, mean observed heterozygosity; F_{IS}, deviation from Hardy-Weinberg expectations according to Weir & Cockerham (1984) (inbreeding coefficient); N, number of individuals genotyped; Pop.size, estimated populations size.

Site N°	Site Name	Habitat type	Railway access	A	H_0	F_{IS}	N	Pop. size
1	wolf	cemetery	yes	4.17	0.47	0.26	21	medium
2	db	disused railway site	yes	5.33	0.56	0.13	24	big
3	hoernli	cemetery	yes	4.58	0.48	0.13	24	big
4	johann	railway site	yes	5.90	0.54	0.23	23	medium
5	jrhein	river bank	yes	3.63	0.42	0.16	23	small
6	wrhein	river bank	**no**	3.53	0.39	0.11	24	small
7	rank	allotment garden	yes	5.79	0.70	0.00	22	medium
8	milch	allotment garden	**no**	2.92	0.42	0.12	22	small
9	chaux	railway site	yes	5.67	0.63	0.09	24	big

GENETIC SAMPLING

Sampling took place during summer and autumn 2005 and 2006. For the capture of the lizards, we used a modified fishing pole. This guaranteed the necessary distance to approach the lizard. At the pole's end, an adjustable noose was fixed with which to snare the lizard without hurting it. Captured lizards were measured and characterized (size, weight, age class, sex, presence of autotomy, and colour of the belly), marked, and photographed before release. This profiling of lizards was done to render evidence of phenotypic differences among subpopulations. From each site a minimum of 25 individuals were captured, in total 231 animals: 139 male, 72 female, 20 juvenile animals.

ISOLATION OF DNA AND GENOTYPING

We used three methods for collecting DNA: mouth-swabs (Poschadel & Möller 2004); loose skin from recent moulting and tissue from the tail tip.

For the buccal cells an alkaline lysis procedure was used, according to the protocol of Meldgaard et al. (2004). Loose skin was collected if available and treated like the tail tips for DNA-extraction. For the tissue, 2 mm of the tail tip was cut and stored in 96 % alcohol. For DNA-Extraction from tail tissue the QIAGEN DNeasy©-Kit was used. DNA was amplified using the polymerase chain reaction and genotyped according to six microsatellites: five microsatellites from *Podarcis muralis* (C9, B7, B3, B4, D1) described by Nembrini & Oppliger (2003) and one microsatellite from *Lacerta vivipara* (Lv3) described by Boudjemadi et al. (1999b). The length of the microsatellites B7, B3, B4 and D1 was shortened to allow faster runs on the electrophoresis apparatus. The adaptation of the primers to realize this optimisation was carried out with the software PRIMER (Marshall 2004). Amplification were performed in volumes of 12.5 µL containing 3 µL of genomic DNA, 1.25 mM_{EC} of $MgCl_2$, 1 µL of 10xbuffer (10mM Tris-HCL, 50mM $KCl)_{EC}$, 10 mM dNTP Mix (Fermentas), 0.3 units of HotMasterTM Taq DNA Polymerase (Eppendorf), 5 pmol/µl of each primer. Amplifications were carried out in a Genius thermocycler. After an initial hot start step for 2 min, 37 cycles were performed each consisting of 92° C for 30 s, 56°/55° for 20 s and an extension step of 68° for 20 s. The electrophoresis was performed on the Origins electrophoresis apparatus (Elchrom™ Scientific). The PCR-Products were analysed on Spreadex® Gels EL300 or EL400 (Elchrom™ Scientific) depending on the size of the amplicons. In total 209 individuals were genotyped.

STANDARD POPULATION GENETIC ANALYSES

Analyses were performed using FSTAT 2.9 (Goudet 2001). For analyses involving multiple comparisons the critical probability for each test was adjusted using the sequential Bonferroni correction (Rice 1989). Genetic diversity was estimated as allelic richness, where the number of alleles per locus was standardised to smallest number of individuals (Mousadik & Petit 1996) and as observed and expected heterozygosities.

Exact tests for deviations from Hardy-Weinberg equilibrium for each locus and linkage disequilibrium between loci were carried out in GENEPOP

3.4 (Raymond & Rousset 1995). No linkage was observed, and therefore all six polymorphic loci could be used in the analysis.

STRUCTURE OF GENETIC POPULATION

Population genetic structure was examined using two different approaches. First, we calculated pair-wise values of F_{ST} (Theta=θ_{ST}) using the method of Weir & Cockerham (1984) for all sites and tested for significance with FSTAT Version 2.9.

Second, we estimated the number and composition of subpopulations represented in the samples with a Bayesian model-based clustering analysis implemented in the program STRUCTURE (Pritchard et al. 2000). This program runs without prior population information or geographic knowledge and uses a Markov Chain Monte Carlo (MCMC) method that clusters individuals to minimise Hardy–Weinberg disequilibrium and linkage disequilibrium between loci. This rather new method has been revised in several publications and gives reliable results of the genetic structure (Latch et al. 2006, Manel et al. 2005, Pearse & Crandall 2004). Runs were performed with a burn-in length of 10^4 and a MCMC of 10^5. We used the admixture model and correlated allele frequency parameters for all models (Falush et al. 2003). To find the cluster size, that best explains the data, ten cluster numbers are tested: from k=1, all individuals belong to the same cluster, to k=10. These tests were run twenty times for each k in order to estimate the number of clusters. The method of (Evanno et al. 2005) was used to infer k. This procedure identifies the appropriate number of clusters using the *ad hoc* statistic Δk, which is based on the second order rate of change in the log probability of the data between successive values of k. This method is useful if the suggested model choice criterion 'Ln P(D)' of Pritchard et al. (2000) is reaching a plateau rather than a maximum.

GIS-MODELLILNG

Cost-distance modelling is based on a least-cost analysis that originates from graph theory (Bunn et al. 2000) and is recommended by Crooks & Sanjayan (2006) as a good compromise between the most basic methods based on Euclidian distances and methods of high complexity to assess the landscape connectivity. It is a standard procedure in ArcGis© (v. 9.1, ESRI) and was specially adapted for genetic questions by Ray (2005) in the extension PATHMATRIX running on ArcView (v. 3.2, ESRI).

The input for the cost-distance method consists of two maps: (1) a map grid with the investigated habitat patches (source-grid) and (2) a map grid with the matrix containing the land use types (cost-grid). On the cost grid every cell has a resistance value (cost), depending on its land use type (Adriaensen et al. 2003). Land use types that correspond with the potential habitat of *Podarcis muralis* and with potential corridors are given low resistance values. Land use types that hinder movement, get higher values (Table 2). This resistance values are based on expert knowledge. River banks, although not always in a natural condition, are potential corridors for the wall lizard because they still contain important features of the original habitat along natural river banks. Railway tracks are also regarded as potential corridors because of dispersal observations in other urban areas (Deichsel & Gist 2001, Hedeen & Hedeen 1999). The land use types of open land, forests and built-up land are attributed higher resistance.

Table 2. Costs for each class of land use for *Podarcis muralis* in the GIS study. The costs of each land use class are modified in each of the four scenarios (S1–S4). % of area: relative area of each class in the 56 km^2 study area. Built-up areas included streets.

Land cover class	% area	S1	S2	S3	S4
Wall lizard habitats	6.0	1	1	1	1
Railway tracks	2.0	1	5	5	100
Railway tunnel	0.1	1	100	5	100
Riverbanks of Rhine and inflows	3.0	1	5	100	5
Open land (meadows, fields, parks)	22.0	1	100	100	100
Forests and forest fragments	5.0	1	500	500	500
Isolated buildings and built-up areas	61.0	1	1000	1000	1000

From these two grids, three paths are calculated between each habitat patch: (1) the Euclidian distance, (2) the effective distance along the land use grid cells with the lowest resistance and (3) the least-cost path (LCP) with the accumulated costs from the land use grid cells with the lowest resistance (Ray 2005). The relation of the Euclidian distance to the effective distance is a measure of the linearity of the calculated least-cost path (Stevens et al. 2006).

MAP

The land use was modelled for a study area of 8 x 7 km (Figure 1). The land use classes were estimated from a shape file with land use classes for open space, information on the percentage of built-up area per city block (GVA 2004) and from the orthophoto for the canton Basel-Stadt (GVA 2001). The resolution of the cells was set to the 10 m as in Broquet et al. (2006). The digitalisation of the wall lizards habitats was based on the inventory of reptiles (Dušej & Müller 2000) and complemented with field observations. The linear elements like railway tracks and small rivers, or river banks in the case of the Rhine were buffered with 10 m to 30 m as suggested by Adriansen (2003).

SCENARIOS

We modelled four different scenarios. First, we analysed if the genetic data followed an isolation-by-distance pattern. In the other three scenarios, we estimate which corridor type contributes most to the connectivity of the investigated habitat patches (Table 2). The first scenario based on the Euclidian distances where all land use categories have the same costs. LCPs under this scenario corresponded thus to bee-line between two habitat patches. The second scenario based on the assumption that wall lizard habitats have the lowest resistance and thus lowest costs, followed by railway tracks and riverbanks. All other land use classes have a considerably higher resistance, with the highest resistance attributed to built-up areas. In the scenario three the resistance of the railway tracks was lower than the riverbanks and in scenario 4 the resistance of the riverbanks was lower than the railway tracks.

To assess the impacts of geographic distance on the genetic differentiation, a Mantel test was carried out (Mantel 1967). By means of a permutation procedure, this analysis tested which scenarios best fitted the genetic differentiation. The Mantel tests were performed in FSTAT 2.9.

COMPARISON OF THE EXAMINED POPULATION TO THE EUROPEAN POPULATIONS

We recalculated the value for allelic richness from the study of Gassert (2005) containing data from populations from Italy, France, Croatia, Germany and Belgium with an average of 11 to 21 individuals. These values were compared to the values of allelic richness of our data. This analyse was based on five microsatellites from the same loci. The allelic richness value was normalized (Mousadik & Petit 1996). The differentiation was evaluated with a one way ANOVA (SPSS V. 14.0, SPSS Ltd.) and a Scheffe post-hoc test on the 'ln'-transformed value of allelic richness.

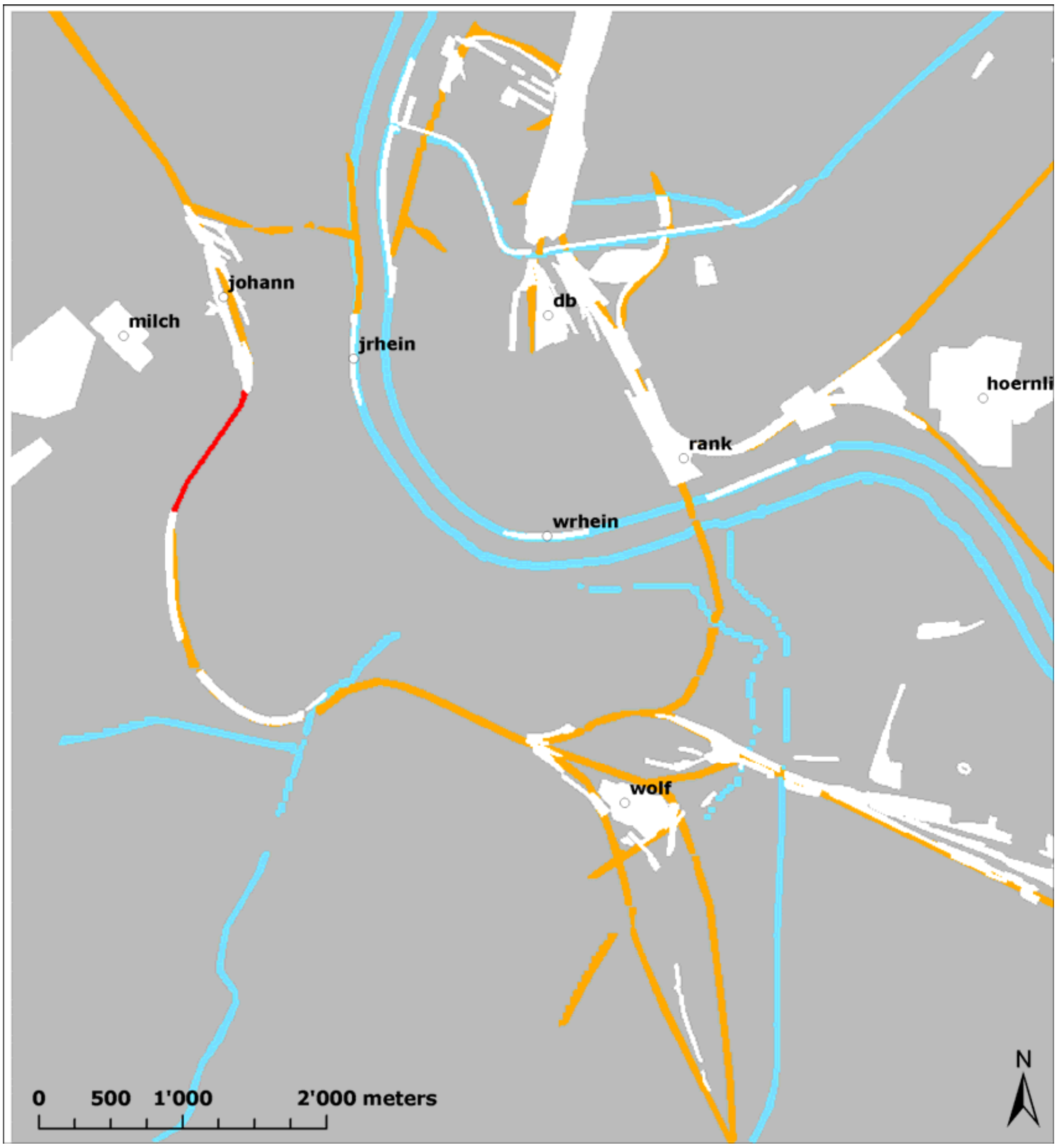

Figure 1. Sampling sites in the canton of Basel-Stadt and land use categories (white: occupied habitats; orange: railway tracks; red: railway tunnel; light blue: river banks and small rivers; grey: open land, built-up land and forests). The site of the control group is not shown (site 'chaux' in St-Ursanne, canton Jura). This site is 37 km away from Basel and is situated on the railway line from Basel to Delle.

Results

STANDARD POPULATION GENETIC ANALYSES OF THE WHOLE SAMPLE

All six microsatellites loci were polymorphic and the mean number of alleles per locus was 12.8, ranging from 8 to 18. Two loci were monomorphic: (C9 on site 'jrhein', Mu23 on site 'milch'). Genetic diversity measured as allelic richness ranged from 2.92 on site 'milch' to 5.90 on site 'johann' (Table 1). Heterozygote deficiency was significant in 4 loci but not over all observed sites. There was no significant linkage disequilibrium. Overall F_{IS} was 0.124 (smallF) and F_{ST} (θ) was 0.155.

STRUCTURE OF GENETIC POPULATION

All populations were significantly different from each other. Across all sites, pairwise F_{ST} levels ranged from 0.056 to 0.327 (Table 3). With regard to the classification of the genetic differentiation by Hartl & Clark (1997) we can identify three classes here: More than half of the pairwise F_{ST} values (19 values) where between 0.05–0.15, corresponding to a medium genetic differentiation; 14 values of the pairwise F_{ST} lay between 0.15–0.25, corresponding to high genetic differentiation and three F_{ST} values were higher than 0.25, corresponding to a very high genetic differentiation. These three high values for F_{ST} were found between the sites 'wrhein' and 'wolf', 'milch' and 'db' and 'milch' and 'whrein'. The last sites had also the lowest allelic richness and the lowest observed heterozygosity (Table 1).

Table 3. Pairwise F_{ST} values between subpopulations and the significance of differentiation on the 1 % significance level (** significant). P-values obtained after 3600 permutations (P=0.001389).

	wolf	db	hoernl	johann	jrhein	wrhein	rank	milch
db	0.139	**	**	**	**	**	**	**
hoernl	0.193	0.189	**	**	**	**	**	**
johann	0.135	0.140	0.083	**	**	**	**	**
jrhein	0.183	0.142	0.118	0.105	**	**	**	**
wrhein	0.299	0.223	0.133	0.189	0.241	**	**	**
rank	0.172	0.094	0.097	0.056	0.108	0.136	**	**
milch	0.166	0.261	0.186	0.174	0.195	0.327	0.237	**
chaux	0.132	0.110	0.086	0.103	0.085	0.163	0.063	0.186

The simulations for the optimal cluster size in the STRUCTURE program reached a plateau at an apparent k estimate of five or more subgroups for our 209 sampled individuals from 9 geographic locations. In addition, using the k estimator derived from the rate of change of k (Evanno et al. 2005) a model with 5 genetic subgroups was supported (Table 4). The five supported subgroups are represented in the cluster analysis (Figure 2) with k=5. Under the admixture model an individual's assignment probability to each cluster can be interpreted as the proportion of that individual's genome that originated in each cluster. Five geographic sites of *Podarcis muralis* ('hoernli', 'johann', 'jrhein', 'rank', 'chaux') were not clearly assigned, but all of them similar to each other having their origin in five different clusters with the colours rose, yellow, blue, green and red. The two sites 'wolf' and 'db' are stronger dominated by the red-cluster, but the other clusters still influenced the composition of these two subpopulations. In contrast, the site 'whrein' is dominated by the blue cluster and the site 'milch' by the rose cluster (Figure 2).

Table 4. The number of clusters (k) versus the second order rate of change in k (Δk). The maximum for Δk at k=5 indicates that five clusters best explain the microsatellite data for the sampled *Podarcis muralis* population.

K	Ln P(D)	var LN P(D)	ΔK
1	-3769.4	34.00	
2	-3613.7	149.16	
3	-3554.5	305.12	0.85
4	-3460.0	391.43	2.21
5	-3390.9	462.27	3.58
6	-3333.0	517.50	1.19
7	-3411.8	819.00	0.44
8	-3313.4	736.76	2.13
9	-3389.6	875.96	1.25
10	-3341.1	844.07	2.41

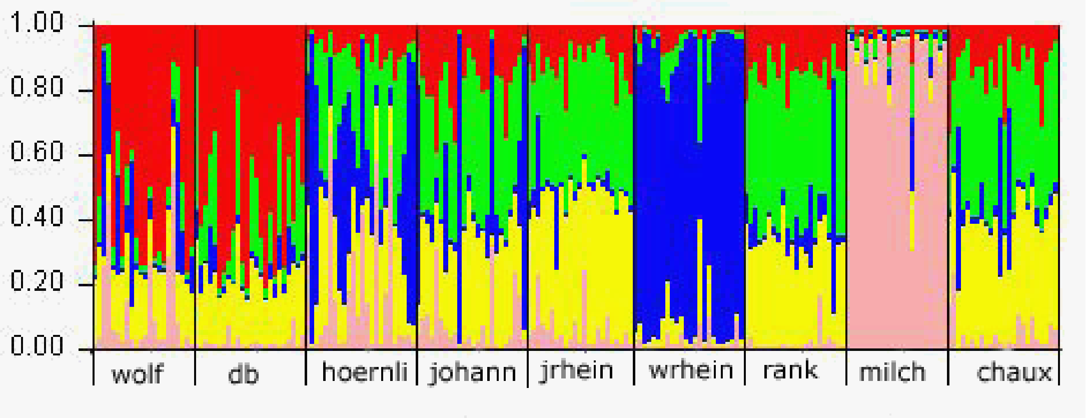

Figure 2. Bayesian assignment probabilities for 5 clusters (k=5). Each vertical line corresponds to one individual. The proportion of each colour in one bar represents an individual's assignment probability to the respective cluster with the five colours rose, yellow, blue, green and red. Individuals of different sites are separated by a black line. Sites are labelled below the figure.

GIS MODELLING

The analysis of the least-cost-path (LCD) for four scenarios gave the following results (Table 5): The Euclidian distances between the sites were not correlated with the genetic distances. Neither was this the case in the scenario two and four, where the resistance of the land use between railway tracks and river banks were equal (S2) or were the resistance of the river banks was lower than the resistance of the railway tracks (S4). However, in the scenario 3, where railway tracks including railway tunnels have the lowest resistance after the *Podarcis muralis* habitats, there was a significant correlation between genetic and geographic distance (P=0.00).

Table 5. Correlation between costs of least-cost paths (LCP) and genetic distances among the eight wall lizard subpopulations in the canton Basel-Stadt. The scenarios compare the resistance between railway corridors and river corridors. Linearity: mean linearity of LCP (± S.E.). Correlation, goodness of fit (R^2) and P values between genetic distance and LCP in the Mantel test are estimated with 2000 permutations.

case	scenario	Linearity	units	Correlation	R^2	P
S1	euclidian	1.00	meters	-0.04	0.13	0.87
S2	railway=river	0.56 ± 0.04	cost units	0.24	5.74	0.21
S3	railway<river	0.49 ± 0.04	cost units	0.68	46.00	0.00
S4	railway>river	0.52 ± 0.03	cost units	0.27	7.13	0.16

COMPARISON OF THE EXAMINED POPULATION TO THE EUROPEAN POPULATIONS

The allelic richness of the three Mediterranean populations from Italy, Croatia and France (Ardèche) was significantly higher than the allelic richness in the population of the canton Basel-Stadt and the canton Jura (Figure 3). The effect was significant with P=0.000. This difference in allelic richness was also observed between the Mediterranean populations and the North-European populations (France: Lothringen, Germany: Trier and Urft; Belgium: Anhée), also with P=0.000. The allelic richness between the northern populations from Gassert (2005) and the population from BS and JU was not significantly different (P=0.675).

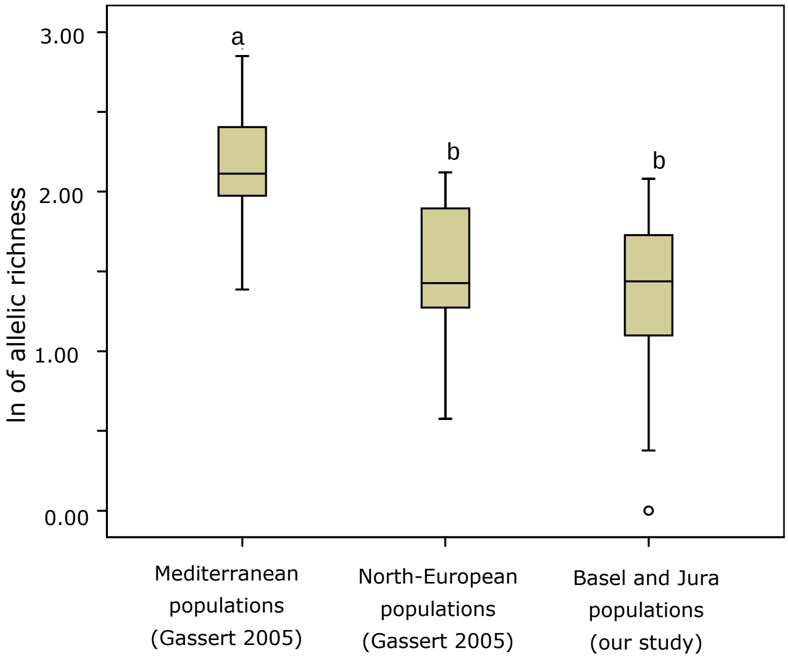

Figure 3. Allelic richness was standardised on 11 individuals. Data from Europe was split into the Mediterranean populations (Italy, Croatia, and southern France) and North-European populations (Germany, Belgium and northern France). Values were recalculated from Gassert (2005) and compared with our data. Labels a and b indicate significant differences.

Discussion

GENETIC POPULATION STRUCTURE

The genetic population structure of *Podarcis muralis* was investigated on nine sites. Eight sites were situated in the canton Basel-Stadt within a study site area of 56 km^2. One site was chosen as a control at a distance of 37 km from Basel. The genetic analyse was carried out with six polymorphic microsatellite loci.

An important first result of our study is that there was significant genetic differentiation among *Podarcis muralis*. All subpopulation pairs were significantly differentiated, even at geographical distances lower than 1 km. A second result from the analysis of the population structure is that the highest F_{ST} value (0.327) occurred between the individuals on the two sites not connected by railway tracks (Table 1). Such high and significant values for the genetic differentiation on a small geographical scale were also reported in two studies on the tree frog (*Hyla arborea*) (Andersen et al. 2004, Arens et al. 2006). Similar high values of F_{ST} were found in a study on Eastern collared lizards (*Crotaphytus collaris collaris*), although in a much larger geographical area (Hutchison & Templeton 1999). Most other studies on lizards used microsatellites to analyse mating system (Laloi et al. 2004, Morrison et al. 2002), reproductive success (Gullberg et al. 1997, Hofmann & Henle 2006) or phylogeny (Harris et al. 2002, Pinho et al. 2006, Poulakakis et al. 2005). Therefore little comparable data from other lizard populations were available.

The high F_{ST} values can be explained by the limited dispersal of the wall lizard and their territoriality. So far, migration distances of 50–90 m have been reported (Dexel 1984, Strijbosch et al. 1980) with an exceptional observation of a recaptured male in 500 m distance from the place where it has been marked (Bender et al. 1996). The observed home range lays between 25 to 69 m^2 (Boag 1973, Brown et al. 1995). The comparison of the home ranges of twenty different lizard species showed that the home range of *Podarcis muralis* is very small compared to other species of similar size. This analysis was based on the figures of home range and snout-vent length (SVL) reported in the Annex of Perry & Garland (2002). This comparison confirms the low dispersal distances of the wall lizard if we assume that the dispersal distance and home range size are positively correlated, as it is the case for mammals (Bowman et al. 2002).

A third result was that the sites 'wrhein' and 'milch' with the highest F_{ST} value, and therefore the highest genetic differentiation from the other subpopulations, showed also the lowest allelic richness and the lowest observed heterozygosity (Table 1). The different status of these two sites compared to the other seven sites was confirmed in the Bayesian analyse, where they show different cluster assignments than the individuals from the seven other sites (Figure 2).

It is therefore interesting to understand, what geographic factor best explains this genetic variance. For that reason, we combined our genetic data with a geographic model to asses the most probable dispersal path within the city of Basel. The four tested scenarios to explain the genetic variation were: 1) an isolation-by-distance scenario, where every cell has the same resistance for dispersal 2) a scenario, where railway tracks and riverbanks have the same low resistance 3) a scenario, where railway tracks have a lower resistance than river banks and 4) a scenario, where river banks have a lower resistance than railway tracks. This analysis provided the following results: The genetic data was shown not to be correlated with the Euclidian distance; therefore the genetic structure did not follow an isolation-by-distance model. The scenarios two and four were not significant either. However, the correlation of the genetic structure and the scenario 3, where the railway corridors have the lowest resistance, was highly significant. This result is supported by genetic differentiation of the control site 'chaux', 37 km away from the other sites in Basel. The individuals on this site were closer related to the individuals on the six 'railway-connected-sites' than to the individuals on the two unconnected sites (Table 3 and Figure 2). Furthermore, the higher functional connectivity of railway corridors for the wall lizard than waterways is underlined by the fact, that the site 'chaux' in St-Ursanne lays in the catchment basin of the Rhone, contrary to the sites in Basel, which lay in the catchment basin of the Rhine.

The importance of railway tracks as corridors have been described by two authors (Deichsel & Gist 2001, Hedeen & Hedeen 1999). Railway infrastructure with ballast of different sizes used in the track systems offer features and micro-climate conditions also found on the primary habitats along gravel banks of natural rivers. My own observation, confirmed by Burckhardt (2003), showed that wall lizards readily use the cable channels along the railway tracks as hiding sites. Nevertheless, the higher functional connectivity of railway tracks compared to waterways would not be predicted for two reasons. First, the initial postglacial dispersal of the wall lizard most likely occurred along riverbanks of large river systems, such as the Rhine or Rhone (Bitz & Bammerlin 1996, Gerken & Schwarz 1988). Second, the railway system seems to be quite young to have already such an impact on the genetic structure of the wall lizard.

Estimation of whether enough time has passed to generate the genetic diversity observed on the study site is difficult to assess without molecular clock estimates. What might have had a major effect on the dispersal of wall lizard was the parallel disappearance of natural flood plains because of large river regulation schemes and the creation of the railway infrastructure with suitable secondary habitats. The regulation of the Upper-Rhine started in 1817 and ended in 1872 (Allgöwer 2000). In 1844, the first railway station was constructed in Basel for trains arriving from the Upper-Rhine valley from Strasbourg. The railway line from Basel to the French border via Delémont connected the village St-Ursanne in the year 1875 to Basel. This railway line could only be constructed, once the river Birs was regulated (1865–1875) (Fridrich 2005). Most of the other major railway infrastructure in Basel were constructed until 1922 (Kunz et al. 1985, SBB 1978). In a simplified estimation, based on a dispersal rate of 0.2 km y^{-1} for the wall lizard (Hedeen & Hedeen 1999), the dispersal from Basel to St-Ursanne along the railway tracks would have taken 172 years. This corresponds to the described time frame of 132 years. Examples of how human modification of land has influenced dispersal patterns in organisms are known. Keller & Largiader (2003) showed that the construction of roads in the last 130 years had a significant effect on the population structure of ground beetles. In a study on the American martens (*Martens america*), the landscapes logging history of the last 40 years was reflected in the genetic structure of the investigated *M. america* populations (Broquet et al. 2006).

From the previous explanations, we understand that the subpopulations on sites connected to railway tracks are closer related than the sites not connected to railway sites. Nevertheless, the high F_{IS} values suggest that inbreeding occurs on all sites except from the site 'rank' and 'chaux' (Table 1). When populations are isolated, their genetically exchange is reduced and local inbreeding occurs. This inbreeding generates an excess of homozygotes, and therefore a positive value for F_{IS} (Beebee & Rowe 2004). Contrary to expectations, the values are not highest on the two most isolated sites 'milch' and 'wrhein'. Their lower inbreeding coefficient could be explained by the fact that these sites are connected to other subpopulations not sampled in this study. Future studies will need to sample other populations along river banks to better assess the connectivity of these habitats.

High inbreeding coefficients can also occur if null-alleles are present (Dakin & Avise 2004). In an other study on *Podarcis muralis* (Gassert 2005) the F_{IS} values were also high, although the investigated microsatellites were of different length. This provides evidence that our F_{IS} values are not caused by null-alleles. Null alleles in genes occur if a point mutation on one allele at a primer binding position prevents the primer from binding. Therefore, this allele is not amplified and not detectable. This leads to the false conclusion of homozygosity and therefore to a higher F_{IS} value.

A third hypothesis for the difference between the subpopulations on the railway sites and the two non-connected sites are assumptions that *Podarcis muralis* is introduced by international goods transports (Deichsel & Gist 2001, Hohl 2003). For Basel this possibility has to be taken into account because one of the investigated sites, 'db', was the main goods station of Basel and one site, 'wolf', is situated next to an active goods stations. Introduced species, such as those transported by goods, can often have a strong impact on indigenous populations by changing natural migration and dispersal patterns (Davis 2003). We did not find evidence for such imported animals when identifying individuals singled out by the analysis with STRUCTURE. To test this hypothesis more profoundly, we suggest to sample wall lizards from their main distribution centres in Europe, such as Italy, which are also centres of exportation for goods, especially for vegetables and fruits.

ALLELIC RICHNESS

The allelic richness of the investigated populations was compared with data from European populations provided by the thesis work of Gassert (2005). The mean allelic richness of the populations in North-west of Switzerland (4.4 alleles per loci) is comparable to the allelic richness of the Northern populations of Europe (4.9 alleles per loci). The Southern populations in Europe were allelic richer (9.1 alleles per loci). Rowe & Beebee (2007) identified such a pattern along a much smaller gradient from North to South in Great Britain for natterjack toads (*Bufo calamita*). This can be explained by the general dispersal pattern of species with a postglacial expansion from the southern refugia to the North and a therefore reduced allelic richness the further away from the initial centres of distribution (Hewitt 2000).

Conclusion

Overall, the cost-distance modelling approach proved to be an efficient way to assess the functional connectivity even for a simple landscape mosaic and to investigate the recent dispersal events of *Podarcis muralis*. The results suggest that railway tracks in the region of Basel are important routes of dispersal for *Podarcis muralis*. The examination of these recent dispersal events on a larger temporal and spatial scale and the comparison to the emerging phylogenetic data would be an interesting further step. The analysis of the allelic richness showed that the investigated population was similar to the Northern populations of Europe. It would be interesting to see if the gradient of allelic richness decreases gradually from South to North or if there is a sharp discontinuity before and after the Alps, a putative location of a watershed in glacial barriers. *Podarcis muralis*, although not very rare in the city, is threatened by isolation and habitat destruction in Europe. By the time of writing, one site, ‘db’, is already destroyed. If preservation of genetic diversity is a priority for species conservation, then the important corridor function of railway tracks should be considered in the green space management of cities. In the long term, the restoration of rivers is essential to provide primary habitats for the wall lizard, to guarantee stable source populations.

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How do small urban forest patches contribute to the biodiversity of the arthropod fauna?

Manuscript

Abstract

Urban forests share unique features compared to rural forests. They are intensively managed, often very small and face a high pressure from visitors. We were especially interested, how small urban forest patches contribute to the biodiversity of the arthropod fauna. Therefore, we compared the species diversity, abundance and assemblages of forest patches of different sizes. Over a period of six months, we run 45 pitfall traps on nine sites and analysed three taxa: spiders, ground beetles and rove beetles. The species number of small urban forest patches did not differ significantly from large urban forests. The species assemblages however changed from smaller forest patches to larger patches. The analysis of ecological groups showed that the spider and ground beetle species were predominantly forest species regardless of the size of the forest patch. Because of uneven sampling effort due to vandalism, species numbers had to be estimated and the species assemblages had to be controlled for under sampling. Therefore, we estimated the similarity of arthropod assemblages with three non-parametric methods. This study demonstrates that urban groves can contribute considerably to the species diversity in an urban area. They can therefore offer valuable sites for experiencing biodiversity and hence biodiversity should always be part of the management goals.

Keywords: Arthropod diversity and assemblages, urban forests, species richness estimators, non-parametric similarity measurements

Introduction

Urbanisation is a dominant process of land alteration, converting rural to urban land (United Nations 2006). This process acts in many ways on biodiversity: in form of habitat loss and habitat fragmentation (Miller & Hobbs 2002), disturbance through traffic or people (Niemelä 1999) and biotic homogenisation (McKinney 2006). In Switzerland urbanisation is the main land conversion process: every day 7.4 ha are converted from rural to urban land (Jordi & BFS 2001), accounting for $^{2}/_{3}$ of the total land conversion.

Urbanisation means not only the initial conversion of land but also that more and more people live in urban areas. Therefore the demand for urban green space for recreational use, gathering or playing sports should rise. This is true for Switzerland where the increase of building areas and recreational areas was almost the same (16.5 % and 16.8 %) for the last 15 years (Hotz et al. 2005). Urban green spaces on the other hand are prime refugees for plants and animals in the city and can harbour a surprisingly high species richness (Gilbert 1989). This species richness can be explained by the geographic situation of the cities (Araujo 2003, Kühn et al. 2004), by the high habitat heterogeneity within urban green spaces (Hermy & Cornelis 2000) but also by the spread of alien species (McKinney & Lockwood 1999).

Urban forests share unique features in relation to suburban and rural forests, including air pollution, disturbance intensity, the heat island phenomenon and the presence of exotic species (McDonnell et al. 1997, Rebele 1994). The effect of this gradient from urban to rural on the fauna has been investigated in several cities all over the world (Alaruikka et al. 2002, Blair 1999, Ishitani et al. 2003, Kache 2001, Klausnitzer & Richter 1983, Niemelä et al. 2002). Less attention has been given to very small forest patches in urban green spaces. These patches consist of planted trees, are intensively managed, intensively used and are defined as urban groves. To analyse species richness and species assemblages, we compared these urban groves with larger forest patches defined as urban woods.

We chose three ground dwelling arthropod taxa for this comparison: spiders (Araneae), ground beetles (Coleoptera: Carabidae) and rove beetles (Coleoptera: Staphylinidae). These taxa are useful indicators for three reasons: They are known to react to environmental changes (McGeoch 1998, McIntyre 2000, Rainio & Niemela 2003), the ecology of spiders and ground beetles and to a lesser extent for rove beetles is well known and these groups are not influenced by direct management action as it would be the case for plants (can be deliberately planted) and associated herbivores, or for birds (installed nest boxes, feeding).

In this study, we aim to answer the following questions: (1) How do urban groves differ from urban woods in arthropod species number, in assemblages composition and in the reaction of different ecological groups? (2) Which measure is best suited to compare arthropod assemblages, taking into account varying sampling effort? (3) Can we identify environmental factors that have the potential to cause differences between arthropod communities, rather than just document differences between urban groves and urban woods?

Methods

STUDY AREA AND SAMPLING DESIGN

Nine sample sites in forest patches situated in public green spaces in the city of Basel, Switzerland (N 47° 33', E 7° 35') were selected. Sites were classified a priori on the basis of their size and management intensity. The management intensity is understood as the intensity of maintenance implemented in this area. In the urban groves this is normally 1 to 3 times a year, where the underwood is cleared, trees are controlled and cut and bushes and trees are planted if necessary. In the urban woods management consists of controls and cuts of trees every 4 to 10 years, the underwood is not managed. The basis of this differentiation is the definition of Kowarik (1995) for urban forest types. Five sites were classified as urban groves. These patches are smaller than 1 ha, intensively managed and dominated by Norway maple (*Acer platanoides)*. Four sites were chosen in urban woods. These patches are larger than 1 ha, extensively managed and dominated by Beech trees, large-leaved Limes and Sycamore maple (*Fagus silvatica, Tilia platyphyllos or Acer pseudoplatanus*) (Table 1).

Table 1. Description of study sites: The sites were a priori classified into two habitat types according to the size of the forest patch. The sites are characterised by management frequency (times per year trees or under story is cut), trampling intensity and microclimate (measured in mega joules per square meter and year). The last two columns show the number of samples realized (max. 95 samples) and the resulting trap days. The smallest sample size respectively the lowest number of trap days are represented in bold letters.

Name of green space	Code	Situated in	Habitat Type	Dominant tree species	Management	Size of habitat	Trampling	Radiation below canopy	Samples	Trap days
					$[y^{-1}]$	[m]		$[MJ\ m^{-2}\ yr^{-1}]$		[d]
Cecile I. Loos-Anlage[1)]	CLA	residential area	urban grove	*Sorbus aucuparia*	1.00	1'600	9.5	1'300	73	770
Jakobsberg	JAB	residential area	urban forest	*Fagus silvatica*	0.17	39'700	33.9	1'500	87	819
Kannenfeldpark	KAP	dense residential area	urban grove	*Acer platanoides*	1.00	6'000	0	1'100	93	896
Margarethenpark	MAP	urban fringe	urban forest	*Tilia platyphyllos*	0.50	18'800	15.7	1'600	86	861
St. Albanteich[2)]	SAT	residential area	urban forest	*Acer pseudoplatanus*	0.10	16'300	8	1'200	94	910
Schützenmattpark	SCP	dense residential area	urban grove	*Acer platanoides*	1.00	5'000	26.1	800	83	756
St. Johanns-Park	SJP	dense residential area	urban grove	*Acer platanoides*	1.00	2'700	17.6	1'300	**67**	**658**
Solitude	SOL	dense residential area	urban grove	*Acer platanoides*	3.00	2'000	3.3	1'300	89	903
Wolfschlucht	WOS	residential area	urban forest	*Tilia platyphyllos*	0.25	15'500	5.3	2'200	94	910

1) Young park, until now mostly dense bushes, some small trees

2) The only site not managed by the cantonal service, but by a private foundation

Table 2. This table summarises four analyses. (1) The result of the analysis of similarity (ANOSIM) with maximum 999 permutations for the *a priori* habitat types tested on different resemblance matrices. (2) The correlation of the resemblance matrices with the geographic distance between the nine sites. (3) The stress of the non-metric multi-dimensional scaling (NMDS), where low values are correlated with a small loss of information. We used 50 restarts and a minimum stress set to 0.01 for the calculations. (4) The results of the stepwise linear regression on the extracted 1st dimension of the NMDS-plots. We used three independent variables for explaining the gradient. Only the logarithmic habitat size entered the model as an explaining variable; trampling and radiation below canopy were excluded by the model. Significant values are represented in bold letters.

		(1) ANOSIM for a priori groups		(2) RELATE distance between sites		(3) stress of NMDS	(4) Linear Regression (only significant factor: natural logarithm of habitat size)				
Taxa	Method	Spearman's ρ	P-value	Spearman's ρ	P-value	2-dim	adj. R^2	F-value	P-value	β	factor
spiders	Bray-Curtis similarity	0.544	**0.016**	0.280	**0.045**	0.04	0.69	18.62	**0.004**	-6.516	0.734
	Chao-Sørensen Index	0.222	0.079	0.486	**0.001**	0.11	--	--	--	--	--
	CNESS (mbest=19)	0.409	**0.008**	0.197	0.103	0.07	0.68	17.71	**0.004**	6.457	-0.728
ground beetles	Bray-Curtis similarity	0.363	**0.040**	0.090	0.264	0.08	--	--	--	--	--
	Chao-Sørensen Index	0.350	**0.024**	0.055	0.379	0.11	--	--	--	--	--
	CNESS (mmax=6)	0.572	**0.016**	0.157	0.159	0.07	0.47	8.18	**0.024**	5.631	-0.63
rove beetles	Bray-Curtis similarity	0.244	**0.048**	0.212	0.114	0.05	0.70	19.44	**0.003**	-5.779	0.651
	Chao-Sørensen Index	-0.060	0.675	0.077	0.324	0.09	0.56	11.00	**0.013**	5.596	-0.631
	CNESS (mbest=14)	0.206	0.119	0.199	0.112	0.05	0.81	35.59	**0.001**	-5.782	0.651

Arthropods were collected using five pitfall traps per study site. The traps consisted of funnels with 11 cm diameter. The funnels were lying flush on plastic tubes of 15 cm length and ended into plastic bottles of 200 ml. The traps were installed with a minimum distance of 10 m along a thought line in the perceived core of the site. After a trial phase of three weeks with monoethylene-glycol, each trap was half filled with propylene-glycol and was covered with a plastic roof to avoid dilution by rain water. Propylene-glycol was chosen because of its lower toxicity.

The sampling period covered 19 trapping series in 6 months (19 April–18 October 2004). In the first three months the traps were emptied and refilled with new solution every week, afterwards every two weeks and in the last two series every three week. Arthropods were sorted and then stored in 75 % ethanol.

The identification to species levels for the three different taxa followed the standard literature. Spiders were identified by a specialist (T. Blick) using the key of Platnick (2004). Ground beetles were identified using standard keys (Freude et al. 1976, Müller-Motzfeld 2006), following the nomenclature of Marggi and Luka (2001). For the rove beetles the key of Freude et al. (1964, 1974) and the nomenclature of Frank & Konzelmann (2002) were used. We classified the three arthropod taxa in five broad ecological groups (eurytopic species, synanthropic species, open land species, wetland species and forest species). This classification was based on regional or national literature: for spiders Maurer et al. (1990), for ground beetles Marggi (1992) and Luka (2004) and for rove beetles Luka (2004) and Callot (2005). For rove beetles we used Buckland & Buckland (2006) in addition to the Swiss and French literature.

ENVIRONMENTAL VARIABLES

We measured *habitat size* [m] as a scalar measurement of the a priori difference between the urban woods (>1ha) and the urban groves (<1ha). For regression analyses their natural logarithm was used to normalise the variable. *Radiation below canopy* [MJ m^{-2} yr^{-1}] summarized the climatic situation. It was calculated indirectly from hemispherical canopy photography, a method widely used in forestry (Coops et al. 2004, Roxburgh & Kelly 1995). These photographs are taken through a hemispherical (fisheye) lens from beneath a canopy looking upward. This 180° pictures were taken from every pitfall trap location adjusted to the geographical North. From the pictures the total radiation below canopy was calculated with the software HemiView version 2.1 (Delta-T Devices Ltd.). The five measurements per site were pooled and the average was used. We measured the human-caused disturbance in urban forest with the intensity of *trampling* according to Grandchamp et al. (2000). Three plots of 10x10 m were randomly selected per site. Within the plots the length and width of trails was measured and multiplied by the intensity of trampling. The intensity was measured in four classes according to Lehvävirta (1999); with 1 for the lowest intensity, where the vegetation is only a little reduced in cover and 4 for the highest intensity, where bare mineral ground is visible. The highest trampling intensity was not encountered in our study.

Six further variables were measured, but omitted from analyses as they were correlated to the above-mentioned parameters: age in years (Falter 1984), altitude, clay yield, litter coverage and isolation, (all correlated with habitat size) and number of vandalised traps (correlated with trampling) (Table 1). This avoids overfitting on the linear regression model: When a correlation between two variables was significant ($P<0.05$) and highly correlated (>0.6), we excluded one of the two variables. We kept the three variables described above which were independent of each other and covered three important aspects of the habitat. The influence of these three variables was tested on species assemblages and on abundance.

STATISTICAL ANALYSIS

To account for under sampling biases in individual and species numbers, we had to standardise our data by sampling effort (see Table 1 for sampling effort per site). Abundance was measured as the number of individuals found in the traps, although we acknowledge that this is also affected by movement activity of species. The standardised abundance was estimated linearly for the lowest sampling effort of 67 samples respectively to the lowest number of trap days (658 trap days). Standardized species richness was estimated with the Mao-Tau function of Colwell et al. (2004) using EstimateS (Colwell 2006).

The potential number of the complete species inventory at sites was calculated with the 2nd order Jackknife estimator (200 random selections of sample order) following recommendations of Brose, Martinez et al. (2003). The estimation of potential species richness was applied to the complete data sets of each of the three taxa and calculated with EstimateS (Colwell 2006).

The methodological problem of incomplete sampling has received particular attention in tropical studies because communities are extremely species rich and many species are represented by only one or very few individuals (Brehm & Fiedler 2004), but problems can generally arise if assemblages are incompletely sampled (Chao et al. 2005). Incomplete sampling occurs often in an urban environment, where traps are destroyed, displaced or filled with earth, so that they are not functional during this sampling series. Although the effect of disturbance in form of trampling on arthropod communities has been investigated (Grandchamp et al. 2000, Kimberling et al. 2001, Lehvävirta et al. 2006), only Grandchamp et al. (2000) rarefied their data to account for sampling effort differences due to vandalism.

However, to compare not only species richness but also the community assemblages where sampling was biased, we used three nonparametric methods. This approach was necessary because our sampling was biased by incomplete sampling caused by vandalism and due to under sampling as revealed by the 2nd order Jackknife estimator (see Results). For complete sampling however, Legendre et al. (2005) recommends the direct analyses of raw data to achieve higher statistical power.

The faunal similarity of samples was measured with three methods: (1) the Bray-Curtis similarity (Bray & Curtis 1957), a standard similarity measure widely used in ecology (Southwood & Henderson 2000), calculated with Primer 6 (Primer-E Ltd)) which does not take incomplete sampling into account, (2) a newly developed estimator based on the Chao's Sørensen Abundance-based similarity index (Chao et al. 2005, Colwell 2006), hereafter Chao-Sørensen index, used in different fields of ecology (Cardelus et al. 2006, Yanoviak et al. 2007) and calculated with EstimateS (Colwell 2006) and (3) the Chord-distance normalized expected species shared (CNESS) (Gallagher 1998, Trueblood et al. 1994), a dissimilarity index used in tropical or benthic β-diversity analyses (Beck & Chey 2007, Snelgrove et al. 2001), calculated with the software COMPAH (Gallagher 1998).

The newly developed Chao-Sørensen index (corrected for unseen species) is a nonparametric pairwise site similarity index. This index considers the probability of individuals being species that are shared or unshared between samples, and is less sensitive to sample size. For further analysis, we rescaled the values (which take values between 0 and 1) to similarity measurement between 0 and 100.

CNESS has the advantage of not being biased by sample size and still taking the abundance of species into account. CNESS is based on the number of expected species shared in a random draw of m individuals from two samples. The choice of the sample size (m) is important: the higher the sample size (m) the smaller the influence of the abundant species and the higher the influence of rarer species in the community. The maximum for m is given by the sample with the lowest number of individuals. To determine the sample size (m) that best represent the most abundant species and the rarest species, we used the method described by Trueblood et al. (1994). By comparing the Kendall’s tau rank order correlation of distances calculated with CNESS, a sample size (m) can be found which has a correlation that is roughly the same between CNESS (m=1) and CNESS (m=max). We calculated the best m with the script findcm.m described in Gallagher (2002), referring to Legendre & Gallagher (2001), in Mathlab (MathWorks, Inc.): for spiders: $m_{best}=19$ ($m_{max}=207$), for ground beetles: $m_{max}=6$, for rove beetles: $m_{best}=14$ ($m_{max}=134$). Because of very low individual numbers on site SOL, the sampling size m for the ground beetle was fixed to 6 and left no room for an independent calculation of the best m without dropping the sites with very low individual numbers. For further analysis, we rescaled CNESS-values (which take values between 0 and $2^{1/2}$) to dissimilarity measurement between 0 and 100. We analysed the differences between the *a priori* groups of urban woods and urban groves with the nonparametric permutation test ANOSIM (Primer 6, Primer-E Ltd.).

To test for spatial auto-correlation, we calculated the relation between the faunal similarity of the sites and the geographic distance between the sites with a nonparametric Mantel test (RELATE, in Primer 6, Primer-E Ltd.).

For the ordination of the nine sites according to their faunal similarity, we used non-metric multidimensional scaling (NMDS), calculated with the software Primer 6 (Primer-E Ltd.), based on the described three resemblance indices. The values of the two extracted axes for the first two dimensions of the NMDS were used as dependent variables in a stepwise linear regression model (SPSS 14, SPSS Ltd.) with environmental parameters. All analyses were performed on three data sets: spiders, ground beetles and rove beetles.

Differences in abundance between ecological groups were analysed with a non-parametric test (Whitney U test), comparing the abundance in every ecological group per taxa with each other. Possible environmental variables were assessed on the most important ecological groups (forest species, open land species, eurytopic and synanthropic species) with a stepwise linear regression model (SPSS 14, SPSS Ltd.). For more precise estimation the abundance numbers were standardised to the minimum number of trap days instead of minimum number of samples.

Results

SPECIES RICHNESS AND ABUNDANCE IN FOREST PATCHES IN URBAN GREEN SPACES

A total of 9311 specimens representing 225 species were collected. The spiders were the richest group in species and individuals numbers with 4606 individuals representing 98 species. The second richest group were the rove beetles with 3099 individuals representing 84 species. The smallest group were the ground beetles with 1606 individuals representing 43 species. The number of singletons ranged from 23 % for ground beetles, to 28 % for spiders and 32 % for rove beetles (see Appendix for species list).

Diplostyla concolor (WIDER, 1834), a typical forest dweller, was the most abundant spider species with 1405 individuals, accounting for 31 % of all individuals. *Ocypus ater* GRAVENHORST (1802) was the most abundant rove beetle with 584 individuals (accounting for 19 % of all individuals), and has been described as a typical urban and suburban dweller (Buckland & Buckland 2006, Callot 2005). *Notiophilus rufipes* CURTIS, 1829 was the most abundant ground beetle with 543 individuals, accounting for 34 % of all individuals. *N. rufipes* is a typical dweller of deciduous forests.

COMPARING SPECIES RICHNESS BETWEEN URBAN GROVES AND URBAN FORESTS

The species richness of urban groves and urban woods was not significantly different (2-tailed exact significance tested with Mann-Whitney U: spiders: p=0.063; ground beetles: p=0.190; rove beetles: p=0.190), although species richness was generally higher in urban groves (Figure 1). The variance of species richness between the two habitats was significantly higher for spiders in the urban groves (Levene's Test for Equality of Variances, p=0.037).

The curves of estimated species numbers by the 2nd order Jackknife estimator did not reach an asymptote at all sites, demonstrating that greater sample effort is required to represent the full species complement of ground-dwelling arthropods in some sites. However, there was no correlation between the intensity of sampling at sites with a low sample size and their performance in the 2nd order Jackknife estimator (2-tailed exact significance tested with Mann-Whitney U: P=0.396). This can be illustrated for the site SAT with almost complete sampling, where the percentage of observed species compared to expected species was as low as 37 % for ground beetles or for the site SJP, where the percentage of observed versus expected species was still as high as 69 % for spiders, although this site had the lowest sample number.

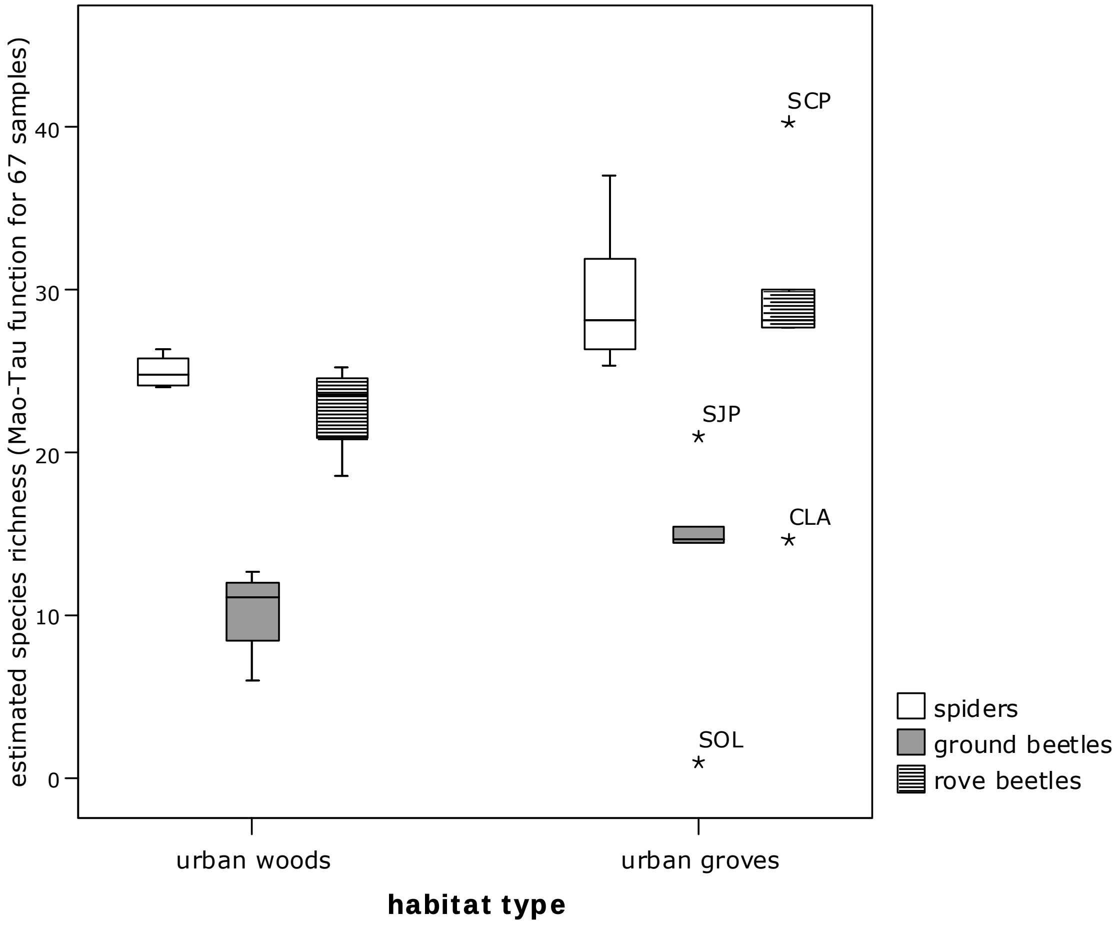


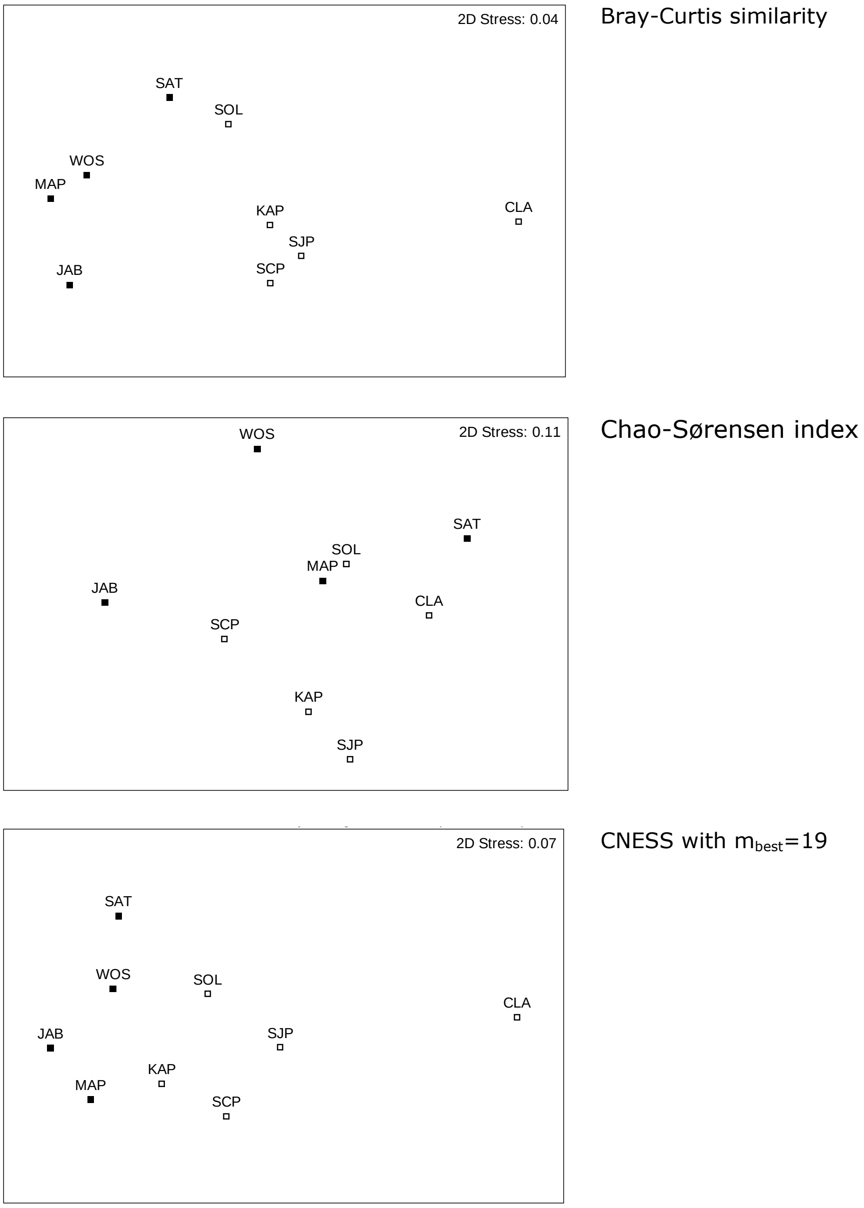
Figure 1. Rarefied species richness in urban woods and urban groves. Species numbers were estimated with the species accumulation curves of the Mao-Tau estimator and standardized to 67 samples. White box plots: spiders, grey box plots: ground beetles and striped box plots: rove beetles. The sites (SJP, SOL, SCP and CLA) with an asterisk are classified as outliners in the box plot diagram. For site abbreviations see table 1, page 50.

FAUNAL SIMILARITY BETWEEN URBAN GROVES AND URBAN WOODS

In Figure 2 we present three of the nine NMDS-plots. The three plots show the faunal similarity of the spider assemblages assessed by the three non-parametric estimators of similarity. The axes are turned, so that CLA is on the right side in the middle of the graph to enable better visual comparison. The ordination of the Bray-Curtis similarity showed the clearest picture for the separation of the two a priori habitats. The separation was also visible for CNESS, although the stress (a measure of information loss during ordination) was higher for this plot. In the visual representation of the faunal similarity calculated by the Chao-Sørensen index the two a priori groups could not be recognized any more. The inter-mixture of urban woods and urban forest was also observed for the NMDS-plots of Chao-Sørensen index in the species assemblages of ground beetles and rove beetles (plots not shown). In the other NMDS-plots (Bray-Curtis similarity and CNESS) of ground beetles and rove beetles similar patterns to the spider-plots were observed.

The responses of the three faunal resemblance estimators to different environmental parameters are summarized in Table 2. The table consists of four parts describing the results of the ANSOSIM, RELATE, STRESS and the results of the linear regression. (1) In six of nine cases the faunal similarity was significantly different (ANOSIM) in the two a priori habitat types: in all three taxa with the Bray-Curtis similarity, in spiders and ground beetles with CNESS and only for ground beetles with the Chao-Sørensen index. (2) The correlation between the faunal similarity and the geographical distance of sites was significant for the spider assemblage estimated with the Bray-Curtis similarity and with the Chao-Sørensen index. In all other cases the faunal similarity was not correlated with geographical distance. (3) The stress of the NMDS, describing the information loss due to ordination, was equal or below 0.05 for the spider and rove beetles assemblages described by the Bray-Curtis similarity and for the rove beetles described by CNESS. This gave a very good representation of the whole data according to the criteria of Clarke & Warwick (2001). The values between 0.05 and 0.11 for the estimation of similarity of spiders and ground beetles with the Chao-Sørensen index can still be regarded as reliable according to the same criteria. (4) The extracted scores of the first dimension of all ordinations correlated with the natural logarithm of habitat size. The radiation below canopy and the intensity of trampling did not explain the gradient in species assemblages and did not enter into final explanatory models. The second dimension could not be interpreted biologically for any of the three indices.

The correlation for the spider assemblages calculated for the Bray-Curtis similarity and the CNESS were very similar, confirming the visual similar pictures in Figure 3. For the ground beetles assemblages the first dimension could only be interpreted by CNESS and was also correlated to habitat size. The rove beetle assemblages were in all three estimators correlated with habitat size. The strongest correlation was observed for CNESS with adjusted $R^2=0.81$ and $P=0.001$.



Bray-Curtis similarity

Chao-Sørensen index

CNESS with $m_{best}=19$

Figure 2. Multi-dimensional scaling analysis (MDS) of spider assemblages based on three methods of resemblance measurements. Bray-Curtis was used after transforming the original species matrix with ln (x+1); CNESS was estimated with the optimal sample size m=19. Filled squares: urban woods, open squares: urban groves. For better visual comparison, the site CLA is always on the right side in the middle of the 2nd axis. For site abbreviations see table 1, page 50.

ABUNDANCE OF THE THREE TAXA AND DISTRIBUTION OF ECOLOGICAL GROUPS

In accordance to the comparison of species richness, abundances did not differ between urban groves and urban forest (2-tailed exact significance tested with Mann-Whitney U: spiders: p=0.29; ground beetles: p=0.19; rove beetles: p=0.41). Mean abundance per trap day for spiders was 0.5 individuals (s.e=0.08), 0.2 individuals for ground beetles (s.e=0.04) and 0.4 for rove beetles (s.e=0.09). The site SAT had very high individual numbers for spiders (over 1000 observed individuals), where as there were almost no ground beetles observed at this site (15 individuals). This was even more pronounced for ground beetles on the site SOL, where only 6 individuals of ground beetles were captured.

Figure 3 presents the allocation of the five ecological groups among the three taxa on the nine sites. The distribution of ecological groups shows prominent differences. The amount of forest species is high in urban forests and urban groves for the spiders and ground beetles. However, we observed only a small number of forest species for the rove beetles. Here, the species composition was dominated by eurytopic, synanthropic and open land species. The statistical analyse of the differences in abundances between the five ecological groups in urban woods and urban groves were only significant for open land spider species (2-tailed exact significance tested with Mann-Whitney U: P=0.016). For this ecological group of species the habitat size (natural logarithm) was extracted as a negatively correlated factor in stepwise linear regression. All other groups were not significantly different distributed in urban woods and urban groves.

Discussion

GENERAL SPECIES RICHNESS

The here presented findings of observed species richness for spiders, ground beetles and rove beetles are in agreement with species numbers in comparable habitats (forests and hedges) in an extensive landscape study in the suburban area of Basel (Luka 2004). In both studies, spiders were the species richest group, followed by rove beetles and then by ground beetles. Nonetheless, rarefaction curves and richness estimates showed that in our study additional sampling effort is likely to yield significantly more species. This is not an unusual outcome when sampling insect assemblages (Chao et al. 2005, Gotelli & Colwell 2001, Magurran 2004), particularly when the assemblage comprises a high proportion of infrequent species (23–32 % singletons).

DIFFERENCES BETWEEN URBAN GROVES AND URBAN WOODS

Contrary to general assumption (Gray's 1989, Niemelä et al. 2002) that smaller forest patches in densely habited areas support lower species richness, this study did not find a significant difference between species richness of urban groves and urban woods. Moreover, the highest observed and estimated species numbers in each taxon occurred in urban groves (37 spider species and 21 ground beetle species in SJP, 44 observed rove beetle species in SCP; respectively 40.23 estimated species). The difference in abundance between urban groves and urban woods were not significant either. Prior studies on arthropod species richness in urban forests with a focus on urban-rural gradients support these findings for some cities (Alaruikka et al. 2002, Niemelä et al. 2002) but not for others (Ishitani et al. 2003, Niemelä et al. 2002).

However, the differences between urban groves and urban woods of the three assemblages showed a less clear picture: The assemblages of the ground beetles were differentiated by each of the three indices, where as this was not the case for the spider or the rove beetle assemblages. The spider assemblages could be differentiated only by two indices (Bray-Curtis similarity and CNESS), albeit highly significant. Rove beetle assemblages however, were only differentiated weakly by the Bray-Curtis similarity.

The comparison of ecological groups between urban groves and urban woods in the three taxa revealed only differences between the abundance of open land species of spiders. This ecological group was more abundant in the urban groves than in urban woods. Contrary to expectations, this study did not find any significant differences in the abundance of forest species between urban groves and urban woods (Niemelä et al. 2002).

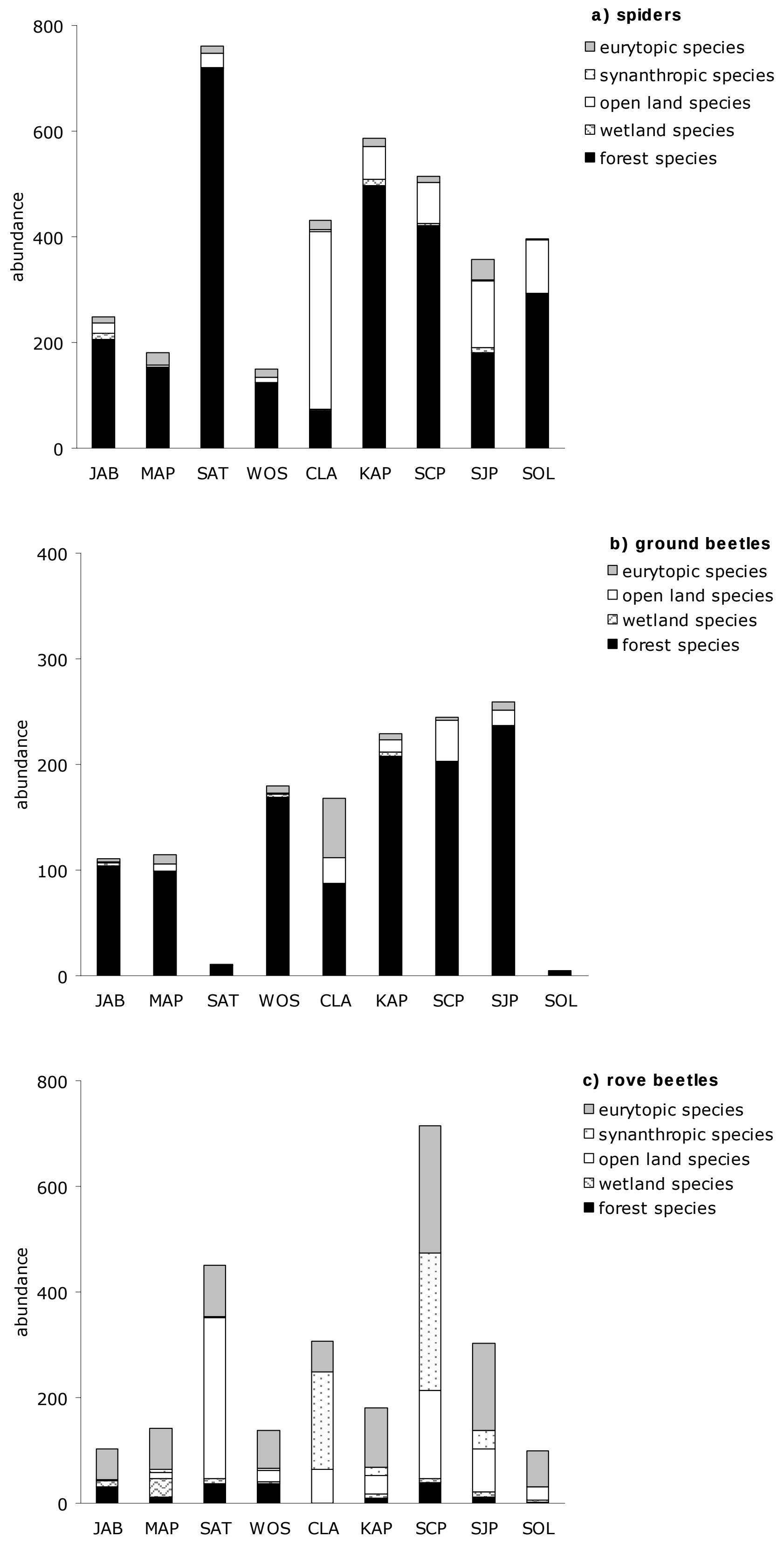


Figure 3. Abundance of ecological groups at the nine sites. Abundance was standardised to the lowest number of trap days (658 trap days). 99 % of spiders, 100 % of ground beetles and 97 % of rove beetles could be assigned to the five (four for ground beetles) ecological groups. For site abbreviations see table 1, page 50.

CHOICE OF ESTIMATORS FOR SPECIES ASSEMBLAGES

On the question of choosing the best indices to compare faunal similarity, this study found that the three investigated indices have advantages and disadvantages.

The Bray-Curtis similarity did differentiate urban groves and urban woods for all three taxa. This was confirmed by a clear visual picture and a low stress of NMDS. The disadvantage of the Bray-Curtis similarity in our case was that it does not take under sampling into account. A different approach to overcome this problem of the Bray-Curtis similarity was followed in the study of Horner-Devine et al. (2004), where the similarity was assessed with a form of community rarefaction by resampling with 1000 randomizations.

The Chao-Sørensen index did not differentiate the two *a priori* habitat types for spider and rove beetle assemblages. This could be due to the fact that the differentiation of the two habitats is weakly represented in the data, as we could suspect from our results of no observed differences in the distribution of ecological groups, except for open land spider species. On the other hand, we observed some surprising results from the number of observed shared species and expected (unseen) shared species, which are central for the calculation of the Chao-Sørensen index. For example, the observed numbers of shared spider species between the site KAP and SAT was 16, whereas the expected number calculated was 211. An even higher discrepancy was detected for the rove beetles between the sites SOL and JAB, where the observed shared species numbers was 11 and the expected shared species number 603. The ground beetle assemblages on the other hand were clearly differentiated for the two habitat types with the Chao-Sørensen index. However, in 8 out of 36 cases the numbers of expected shared species was lower than the observed shared species, which clearly points to errors in the method. Similarly unrealistic results of the estimator had also been observed in other data (e.g. J. Fahr, pers. notice).

The NMDS plots of the CNESS estimator gave a good representation of the faunal similarity with generally low stress values, although not as low as the values of the Bray-Curtis similarity. However, the estimation of faunal similarity for the ground beetles is biased by the low individual number at the site SOL. In this cases the highest possible m should be selected (Brehm & Fiedler 2004), which was done here for the ground beetles. With m=6 the most abundant species are given high priority, whereas the estimation of the best m for spiders and rove beetles finds the balance between the most abundant species and the rarer species. Therefore the estimation of the best m is not possible for studies where a site has very low individual numbers compared to the other sites.

FACTORS INFLUENCING SPECIES RICHNESS, ARTHROPOD ASSEMBLAGES AND THE DISTRIBUTION OF ECOLOGICAL GROUPS

The third question of this study was if we can identify factors that influence the investigated arthropod communities beyond the *a priori* habitats urban groves and urban woods. With the stepwise linear regression model on the extracted first axis, we confirmed the strong influence of the patch size on the species assemblage (adjusted R^2 from 0.38 to 0.81). The dependence of habitat size is connected to the variables that were correlated with habitat size, like litter, clay, age and isolation. Other factors did not contribute to the model (radiation below canopy and trampling).

Contrary to countless other studies reporting a positive species-area relationship (Cornelis & Hermy 2004, Godefroid & Koedam 2003, Tilghman 1987), we neither found a correlation between the size of forest patches and species richness, nor was the number of forest species correlated with habitat size. This outcome is supported by the review of Debinski and Holt (2000), who reported a remarkable lack of consistency in results with regard to species richness and abundance relative to fragment size.

However, these findings suggest that the communities change along a gradient of habitat size, keeping the same amount of species. Resulting antagonistic effects in the abundance of taxa were observed in open land spider species, which were significantly more abundant in smaller sites than in larger sites. An other possible evidence for this interpretation was the presence of forest indicators for ground beetles defined by the indicator species concept for forests of Baden-Württemberg (Landesanstalt für Umweltschutz 1996). These indicator species (*Carabus coriaceus* Linnè, 1758, *Abax parallelepipedus* (Piller & Mitterpacher, 1783), *Abax parallelus*

(Duftschmid, 1812) and *Molops piceus* (Panzer, 1793)) were only found in the urban woods and therefore only in the larger forest patches. The abundance of forest specialist in rove beetles was generally low; however, they were missing entirely in the two smallest forest patches.

Although management intensity was correlated to patch size (low management intensity in larger forests, high management intensity in small patches) and therefore not used as an independent factor, the consideration of the management practice (Table 1) on each site could reveal interesting points for future research: Within the urban groves two sites CLA and SOL are different in their management practice from the rest (KAP, SJP, SCP): the site SOL has a highly representative function and is therefore the most intensively managed grove of the investigated parks. This shows in accurately cut bushes and trees and a planted ground layer with exotic species. The management concept of the site SOL could therefore influence the striking lack of ground beetles. This taxon reacts maybe more sensible to extremely high management intensity than spiders or rove beetles. The site CLA on the other hand is the youngest urban grove with a lot of pioneer species in the vegetation, bushes and not yet full grown trees. The management concept of the site CLA could explain the observed high relation of open land specialist for spiders, the high number of eurytopic ground beetles and the lack of forest species for rove beetles.

Within the urban woods the site SAT is the only forest maintained by a private foundation and not by the communal services of canton Basel-Stadt. The aim of the foundation is to reach a natural forest and their management intervals are very long (every 8 to 10 years) compared to the other urban woods. How this would influence the abundance of ground beetles can not easily be understood and it can only be guessed that the extreme high abundance of spiders may form a competitive situation between the two predatory taxa.

Conclusion

This study demonstrates that urban groves can contribute considerably to the species richness in an urban area like Basel as it has been shown for urban green roofs (Kaupp et al. 2004). This is possibly influenced by the higher heterogeneity of small forest patches with usually more plant species, and more anthropogenically modified structure. However, the analysis of ecological groups shows that the occurring spider and ground beetle species are still predominantly forest species in both habitat types. On the other hand there were only a small proportion of forest rove beetles in urban groves and urban woods. What did change though, were the species assemblages along a gradient from small to large patches. To extract the environmental variable with the highest prediction potential on species assemblages in urban forests, a study design should be taken in account where the correlated factors age, littering, clay and isolation are separated. The comparison of faunal similarity needs careful consideration. In our study the CNESS performed well, even for very low sample size numbers as it was the case for ground beetles. To minimize under sampling, the study design in urban areas should take vandalism into account from the very beginning either by setting traps as buffers or by intensifying the collection interval. Overall, urban forests can offer valuable sites for experiencing biodiversity and therefore the biodiversity should always be part of the management goals.

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Annex

Table A1. Species list of spiders (Araneae) collected in the urban forest patches in Basel, Switzerland. Spiders were identified by a specialist (T. Blick) using the key of Platnick (2004). For site abbreviations see table 1, page 50.

Species names	CLA	JAB	KAP	MAP	SAT	SCP	SJP	SOL	WOS
trap days per site	770	819	896	861	910	756	658	903	910
Agelena gracilens C.L. Koch, 1841			1				1		
Amaurobius ferox (Walckenaer, 1830)								2	1
Anyphaena accentuata (Walckenaer, 1802)		1	1	1			1		1
Apostenus fuscus Westring, 1851					184				
Arctosa leopardus (Sundevall, 1833)					1				
Aulonia albimana (Walckenaer, 1805)	3								
Ballus chalybeius (Walckenaer, 1802)				1					
Centromerita bicolor (Blackwall, 1833)			1				2		
C. sylvaticus (Blackwall, 1841)								1	1
Ceratinella scabrosa (O. P.-Cambridge, 1871)				7					
Cetonana laticeps (Canestrini, 1868)		1				1			
Cheiracanthium mildei L. Koch, 1864	1								
Cicurina cicur (Fabricius, 1793)									1
Clubiona comta C.L. Koch, 1839				1	5			1	
C. terrestris Westring, 1851	2			1	4	3	2	1	2
Cnephalocotes obscurus (Blackwall, 1834)							1		
Coelotes terrestris (Wider, 1834)		12		10					
Diaea dorsata (Fabricius, 1777)				1					
Dictyna arundinacea (Linnaeus, 1758)								1	
Dicymbium nigrum brevisetosum Locket, 1962							2		
Diplocephalus cristatus (Blackwall, 1833)						3		1	
D. latifrons (O. P.-Cambridge, 1863)		9	27	14	2	87	10	9	8
D. picinus (Blackwall, 1841)		4	22		2	35	13	10	
Diplostyla concolor (Wider, 1834)	1	105	436	7	555	118	30	135	18
Dysdera crocata C.L. Koch, 1838			4			9			
D. erythrina (Walckenaer, 1802)			1	1	6	1	23	31	5
Enoplognatha ovata (Clerck, 1757)						1			1
E. thoracica (Hahn, 1833)	7	1	4	1	1	1	6		
Eperigone trilobata (Emerton, 1882)							10		
Episinus angulatus (Blackwall, 1836)		3				2			6
E. truncatus Latreille, 1809	4					1	4	1	
Erigone dentipalpis (Wider, 1834)						2	3		
Ero furcata (Villers, 1789)		1			3	1	1	3	
Euophrys frontalis (Walckenaer, 1802)	1								
Eurocoelotes inermis (L. Koch, 1855)		5							11
Gonatium rubellum (Blackwall, 1841)					3				
Hahnia nava (Blackwall, 1841)	15		2		1		13		
H. pusilla C.L. Koch, 1841		4						1	
Haplodrassus signifer (C.L. Koch, 1839)							1		
Harpactea hombergi (Scopoli, 1763)						2		9	1
H. rubicunda (C.L. Koch, 1838)	1						1		
Histopona torpida (C.L. Koch, 1837)				3	10				16
Lathys humilis (Blackwall, 1855)						1			
Linyphia hortensis Sundevall, 1830		1		1					1
L. triangularis (Clerck, 1757)	2								
Macrargus rufus (Wider, 1834)		4		1					
Maso sundevalli (Westring, 1851)			7			4		1	
Meioneta rurestris (C.L. Koch, 1836)	9	1	6			11	24		
M. simplicitarsis (Simon, 1884)							6		
Metellina merianae (Scopoli, 1763)									1
Micaria pulicaria (Sundevall, 1831)	4								
Micrargus herbigradus (Blackwall, 1854)		1						4	
M. subaequalis (Westring, 1851)	1		9			58	18		1

Species names	CLA	JAB	KAP	MAP	SAT	SCP	SJP	SOL	WOS
trap days per site	770	819	896	861	910	756	658	903	910
Microneta viaria (Blackwall, 1841)		12	78	22	39	4	5	31	5
Monocephalus fuscipes (Blackwall, 1836)			2	1		7			
Myrmarachne formicaria (De Geer, 1778)	2								
Neottiura bimaculata (Linnaeus, 1767)					2	1			
Neriene clathrata (Sundevall, 1830)	4	1	2	3	2	3		1	3
Nigma flavescens (Walckenaer, 1830)									1
Oedothorax apicatus (Blackwall, 1850)									1
Ozyptila praticola (C.L. Koch, 1837)	34	4	21	50	4	130	19	42	5
O. trux (Blackwall, 1846)		15							
Pachygnatha degeeri Sundevall, 1830				1			20		
Palliduphantes pallidus (O. P.-Cambridge, 1871)	1	14	9	31	19	1	9	1	21
Panamomops sulcifrons (Wider, 1834)			15						
Pardosa hortensis (Thorell, 1872)	80	2						5	
P. saltans Töpfer-Hofmann, 2000		1	1		23			12	
Philodromus albidus Kulczynski, 1911				1					
P. cespitum (Walckenaer, 1802)	1								
P. collinus C.L. Koch, 1835					1				
P. praedatus O. P.-Cambridge, 1871								1	
Phlegra fasciata (Hahn, 1826)	1								
Phrurolithus festivus (C.L. Koch, 1835)	10		5		1		1	1	
Pirata hygrophilus Thorell, 1872									1
P. uliginosus (Thorell, 1856)		20							
Pisaura mirabilis (Clerck, 1757)	4								
Pseudeuophrys erratica (Walckenaer, 1826)								1	
Psilochorus simoni (Berland, 1911)	1								
Robertus lividus (Blackwall, 1836)				1					
Tegenaria atrica C.L. Koch, 1843	2						1		
T. domestica (Clerck, 1757)							1		
T. silvestris L. Koch, 1872									2
Tenuiphantes cristatus (Menge, 1866)		1							
T. flavipes (Blackwall, 1854)	1	61	67	72	116	64	44	104	67
T. mengei (Kulczynski, 1887)					1				
T. tenebricola (Wider, 1834)				1					
T. tenuis (Blackwall, 1852)	27	7	10		1	18	19	2	4
T. zimmermanni (Bertkau, 1890)					3				
Tiso vagans (Blackwall, 1834)			3			6	10		
Trachyzelotes pedestris (C.L. Koch, 1837)	4		19	1	4		2	11	
Trochosa ruricola (De Geer, 1778)	3	1	40			12	7		
T. terricola Thorell, 1856		5		2	28		4	1	13
Walckenaeria antica (Wider, 1834)	2						3		
W. corniculans (O. P.-Cambridge, 1875)		14		1					
Xerolycosa miniata (C.L. Koch, 1834)	1						2		
Xysticus cristatus (Clerck, 1757)	1								
Zelotes exiguus (Müller & Schenkel, 1895)	3								
Zodarion italicum (Canestrini, 1868)	271		5	1	32	6	39	120	9

Table A2. Species list ground beetles (Coleoptera: Carabidae) collected in the urban forest patches in Basel, Switzerland. Ground beetles were identified using standard keys (Freude et al. 1976, Müller-Motzfeld 2006), following the nomenclature of Marggi and Luka (2001). For site abbreviations see table 1, page 50.

Species names	CLA	JAB	KAP	MAP	SAT	SCP	SJP	SOL	WOS
trap days per site	770	819	896	861	910	756	658	903	910
Abax parallelepipedus (Piller & Mitterp., 1783)		8		6					89
A. parallelus (Duftschmid, 1812)		2		5	7				46
Agonum muelleri (Herbst, 1784)					1				
Amara aenea (De Geer, 1774)	11					3	1		
A. ovata (Fabricius, 1792)	2	2					3		
A. tibialis (Paykull, 1798)	1								
Anisodactylus binotatus (Fabricius, 1787)			1						
Anchomenus dorsalis (Pontoppidan, 1763)			1						
Anisodactylus signatus (Panzer, 1797)			1			1			
Asaphidion flavipes (Linné, 1761)		4	5	1			1		3
Badister bullatus (Schrank, 1798)	96	4	5	7	1	27	55		3
B. meridionalis Puel, 1925									1
Bembidion lampros (Herbst, 1784)				1			1		1
B. quadrimaculatum (Linné, 1761)	1		1		1	1	1		
Bradycellus csikii Laczo, 1912	1								
Carabus coriaceus Linnè, 1758									27
Calathus melanocephalus (Linné, 1758)	5						1		
Diachromus germanus (Linné, 1758)		1							
Harpalus affinis (Schrank, 1781)	6		13			13	1		
H. ardosiacus (Lutshnik, 1922)							1		
H. atratus Latreille, 1804	4		5		1	1	31		
H. distinguendus (Duftschmid, 1812)	1								
H. griseus (Panzer, 1797)							1		
H. latus (Linné, 1758)			1	1		1			
H. puncticeps (Stephens, 1828)	1						4		
H. rubripes (Duftschmid, 1812)	62		1			1	2		
H. rufipes (De Geer, 1774)	1					26	3		
H. rufibarbis (Fabricius, 1792)	1						1		
Leistus fulvibarbis Dejean, 1826				9			1		
L. rufomarginatus (Duftschmid, 1812)			1	14					
Limodromus assimilis (Paykull, 1790)									2
Loricera pilicornis (Fabricius, 1775)						1			1
Molops piceus (Panzer, 1793)		1							13
Nebria brevicollis (Fabricius, 1792)	2	80	6	94	2	61	5		46
Notiophilus biguttatus (Fabricius, 1779)		11	10	1		12	14		
N. rufipes Curtis, 1829		17	254	1	1	129	127	6	8
Parophonus maculicornis (Duftschmid, 1812)	2					2			
Poecilus cupreus (Linné, 1758)		2					1		7
Pterostichus anthracinus (Illiger, 1798)			1			1			
P. madidus (Fabricius, 1775)		6							
P. oblongopunctatus (Fabricius, 1787)					1	1	4		
Synuchus vivalis (Illiger, 1798)			3						
Trechus quadristriatus (Schrank, 1781)			3	10					1

Table A3. Species list of rove beetles (Coleoptera: Staphylinidae) collected in the urban forest patches in Basel, Switzerland. For the rove beetles the key of Freude et al. (1964, 1974) and the nomenclature of Frank & Konzelmann (2002) were used. For site abbreviations see table 1, page 50.

Species names	CLA	JAB	KAP	MAP	SAT	SCP	SJP	SOL	WOS
trap days per site	770	819	896	861	910	756	658	903	910
Acrotona parens (Mulsant & Rey) 1852						1	1		
Aleochara bipustulata (Linné) 1761							4		
Alevonota rufotestacea (Kraatz) 1856			2		1			1	
Aloconota gregaria (Erichson) 1839			1			5			
Amischa analis (Gravenhorst) 1802									1
Anotylus inustus Gravenhorst 1806					1	1			
A. sculpturatus Gravenhorst 1806					3			1	2
A. tetracarinatus (Block) 1799					1	9		1	4
Anthobium atrocephalum (Gyllenhal) 1827				1	2				3
Anthophagus angusticollis Mannerheim 1830		1		1					
Atheta aeneicollis (Sharp, 1869)							1		
A. boreella Brundin 1948							1		
A. coriaria (Kraatz) 1856						1	1		
A. elongatula (Gravenhorst) 1802						1			
A. ermischi Benick						1			
A. flavipes (Gravenhorst, 1806)		1		1		1	1	3	3
A. fungicola Thomson 1852							1		
A. gagatina Baudi 1848								3	
A. inquinula (Gravenhorst) 1802						1			
A. orphana (Erichson) 1837	3	2	3	9	23	16	6	75	2
A. paracrassicornis Brundin 1954									1
A. sodalis (Erichson) 1837		2		1		1		1	
A. triangulum (Kraatz) 1856						1			1
A. trinotata (Kraatz) 1856								1	
Bolitobius castaneus (Stephens) 1832			1			1			
B. thoracicus				1					
Bolitochara bella Märkel 1844									1
B. pulchra (Gravenhorst) 1806					1				
Callicerus rigidicornis Erichson 1839		1							1
Coprophilus striatulus (Fabricius) 1792			1						
Cypha longicornis (Paykull) 1800						7	1	5	
C. pulicaria Erichson 1839	1	1				7		4	
Dinaraea angustula (Gyllenhal) 1810	1							1	
Drusilla canaliculata (Fabricius) 1787	37		3	1	39	3	14	23	
Falagrioma thoracica Curtis 1832			20	14	382	15	52	9	8
Gabrius femoralis (Hochhut) 1851								1	
Habrocerus capillaricornis (Gravenhorst) 1806					1			1	
Homoeusa acuminata (Märkel) 1842			1		1	5		2	11
Hypopycna rufula (Erichson) 1840								1	
Lesteva longoelytrata (Goeze) 1777								1	
Liogluta longiuscula (Gravenhorst) 1802		2	11	15	6	3	1	2	3
L. microptera Thomson 1867			3	1	1	3	3		
Medon fusculus (Mannerheim) 1830								1	
Mycetoporus longulus Mannerheim 1830							1		
M. nigricollis Stephens 1835			1	12		1		1	
Nehemitropia lividipennis (Mannerheim) 1831							2		
Ocypus ater (Gravenhorst) 1802	216		21	8	1	299	35		4
O. brunnipes (Fabricius, 1781)			4						
O. compressus (Marsham) 1802		1	58	4	4	42			19
O. olens (Müller) 1764	2		17			148			5
Oligota pusillima Gravenhorst 1806			2				2	4	
Omalium rivulare (Paykull) 1789		8	29	45	58	7	1	1	8
O. rugatum Rey 1880			1						

Species names	CLA	JAB	KAP	MAP	SAT	SCP	SJP	SOL	WOS
trap days per site	770	819	896	861	910	756	658	903	910
Othius punctulatus (Goeze) 1777			1			3			
Oxypoda acuminata (Stephens) 1832		10	7	2	14	2	10	1	1
O. brevicornis (Stephens) 1832		1			3	1			
O. mutata Sharp 1871								1	
Philonthus carbonarius (Gravenh.) 1810		1		1				1	1
P. decorus (Gravenhorst) 1802		23	3	8		31			36
Phyllodrepa ioptera (Stephens) 1834						1			
Plataraea brunnea (Fabricius) 1798		13	26	15	12	178	57		53
Platydracus latebricola (Gravenhorst) 1806	42		3		1	3	57		1
Proteinus ovalis Stephens 1834		15	12	46	13	8	9	2	3
Quedius boops (Gravenhorst) 1802	1								
Q. cinctus (Paykull) 1790			2			2	1		
Q. fuliginosus (Gravenhorst) 1802	16	10	3	3	17	9	8	1	5
Q. lateralis (Gravenhorst) 1802				1					
Q. limbatus (Heer) 1839				1					
Q. semiaeneus Stephens 1833	4		1			2	5		
Rugilus rufipes (Germar) 1836		5		3	28	6	1		9
Sepedophilus immaculatus (Stephens) 1832	1		2	1	6	1		1	
S. testaceus (Fabricius) 1792								6	
Stenus flavipalpis Thomson 1860		1							
S. fuscicornis Erichson 1840		34	11	6	5	9	27	10	2
S. impressus Germar 1824					2				
Sunius melanocephalus (Fabricius) 1792						1			
Tachinus signatus Gravenhorst 1802					1				
Tachyporus hypnorum (Fabricius) 1775			1			1			
T. nitidulus (Fabricius) 1781	13					1	3		
T. pusillus Gravenhorst 1806	17	1	1			1			
Xantholinus linearis (Olivier) 1795	5	1	1	1	2	3	4		4
X. semirufus (Ritt.) Steel, 1950	5						1		
Zyras fulgidus (Gravenhorst, 1806)								3	
Z. lugens (Gravenhorst) 1802					1			1	

Leistus fulvibarbis Dejean – Wiederfund einer verschollenen Laufkäferart (Coleoptera, Carabidae) in der Schweiz

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Abstract

Leistus fulvibarbis Dejean has not been recorded from Switzerland for more than a century. Recently the species was found in intensively visited parks in the city of Basle, cantons BS and BL, Switzerland. We briefly discuss possible patterns of the recent dispersal of this species from its Atlantic-European range.

Einleitung

Leistus fulvibarbis Dejean, 1826, wurde zum ersten Mal seit mehr als 100 Jahren wieder für die Schweiz nachgewiesen. Anhand dieses Wiederfunds in Basel soll auf die Fundumstände eingegangen werden und die Verbreitung und mögliche Ausbreitungswege dieser Art diskutiert werden.

L. fulvibarbis kommt von Westeuropa über den Mittelmeerraum bis nach Kleinasien vor (Horion 1941, Turin 1981) und wird bei Jeannel (1947) als typische Art für ein atlantisches Verbreitungsmuster vorgestellt. In Frankreich ist die Art verbreitet, fehlt aber im Nordosten (Jura und Vogesen) (Deville 1921, Jeannel 1941). Im Elsass ist sie bei Callot & Schott (1993) nicht in der Faunenliste erwähnt, sondern nur als alte Meldungen ohne Belege von Bourgeois (ca. 1910) und Scherdlin (ca. 1916) vermerkt, für die keine Bestätigung durch neue Funde vorliegt. Die Angaben zu den Fundorten weisen auf die Hoheneck in den Vogesen und auf Wacken, heute ein Stadtteil von Strassburg, hin. Für Deutschland ist die Art nach Köhler & Klausnitzer (1998) für die Gebiete Rheinland, Saarland und Nordrhein gemeldet. Nach Trautner & Schüle (1996) könnte sich die Art in Deutschland in einer Ausbreitung nach Osten befinden. Neu wurde sie auch in Baden-Württemberg nachgewiesen (Schanowski & Schiel 2004) und gilt als "in Ausbreitung begriffene Art" (Trautner et al. 2005).

In der Schweiz gilt die Art als verschollen; die letzten Funde in Marggi (1992) sind: Veyrier/Petit Salève, GE/Haute Savoie (1899) und Savièse, VS (1899) gefunden von Charles Maerky sowie Biel, BE (1899) von Albert Mathey. Diese Angaben werden durch Marggi & Luka (2001) bestätigt. Weitere Schweizer Fundorte in der Literatur sind Vallorbes (NE) (Heer 1837, Stierlin & de Gautard 1869) und Basel (Stierlin 1900). Diese letztgenannten Autoren schätzen die Art für die Schweiz als selten bis sehr selten ein.

Die Art bevorzugt nach Burmeister (1939) und Valemberg (1997) Sandboden und kommt auf Feldern und in Wäldern vor. In Turin (1981) wird eine Bevorzugung feuchter, beschatteter Lebensräume auf kalkhaltigem Boden angegeben. Trautner & Schüle (1996) beschreiben die Fundstellen als extensiv genutzte Bereiche, oft an Ufern unterschiedlicher Gewässer. Nach Eyre & Luff (2004) ist das Vorkommen von *L. fulvibarbis* am stärksten mit dem Vorkommen von Laubwäldern korreliert und etwas schwächer mit Ackerland und Weiden.

Fangmethoden

Wir sammelten während der Vegetationsperiode 2004 vom 19. April bis am 18. Oktober in neun Parkanlagen in der Stadt Basel Arthropoden. Im Rahmen des MGU-Projektes Natur-Begegnungs-Stadt Basel sollte der Anteil biotoptypischer Arten in Wäldern urbaner Grünräume untersucht werden. Pro Standort wurden je fünf Barberfallen mit Trichterdurchmesser von 10 cm und Regendächern (Luka 2004) im Abstand von 10 m platziert. Die Fallen wurden in der ersten Hälfte der Fangperiode wöchentlich geleert, ab Juli noch alle zwei bis drei Wochen. Als Fangflüssigkeit wurde 100 ml Propylenglykol in 200 ml grossen Fangflaschen verwendet.

Fundortbeschreibung

In zwei Parkanlagen (Abb. 1), im Margarethenpark (Kanton BL: 610930/265700) im Süden der Stadt und im St. Johanns-Park (Kanton BS: 610720/268630) im Nordosten der Stadt wurden insgesamt zehn Individuen gefunden. Die Hälfte der Individuen trat in den Fallenleerungen im Juni, die andere Hälfte in der letzen Leerung im Oktober auf. Neun Individuen wurden im Margarethenpark gefunden; ein Individuum Mitte Juni im St. Johanns-Park.

Der Margarethenpark ist ein Naherholungsgebiet Basels mit schätzungsweise 1200 Besuchern pro Tag (Blumer 2005). Er liegt auf Basel-Landschaftlichem Kantonsgebiet, wird aber von der Stadtgärtnerei Basel unterhalten. Im Süden des Parks liegen Weide- und Ackerflächen des Bruderholzes. Der St. Johanns-Park mit ca. 900 Besuchern pro Tag (Blumer 2005) ist von drei Seiten mit Siedlungen umgeben und im Westen zum Rhein hin offen.

Beide Pärke werden nach den Grundsätzen der differenzierten Pflege unterhalten, d.h. intensiv geschnittene und gedüngte Rasenflächen im Zentrum der Grünanlage werden gegen den Rand hin durch extensivere Wiesen und Gebüsche abgelöst. In den Gehölzen werden im St. Johanns-

Park ein bis zwei Mal pro Jahr Pflegemassnahmen durchgeführt. Im Margarethenpark wird nur nach Bedarf (M. Zemp, pers. Mitt.) eingegriffen.

Abb. 1. Fundorte von *Leistus fulvibarbis* im Margarethenpark (links) und im St. Johanns-Park (rechts). Fotos: Heike Oldörp.

Die Vegetation an den Fundstellen im Margarethenpark kann als lindenreicher Parkwald charakterisiert werden. Das ganze Waldstück ist etwa 1,8 ha gross. Der Boden ist eine Braunerde mit tonigem Schluff (in den ersten 30 cm) und die Laubauflage um die Fallen herum betrug durchschnittlich 3 cm. Der Gehölzbestand im St. Johanns-Park mit einer Grösse von 0,4 ha wurde 1992 bei der Neuschaffung des Parks angelegt (Baur 2000). An der Fundstelle gehen diese Neupflanzungen in den Altbestand des Villengartens über. Der Boden ist ein Treposol mit lehmigem Sand (in den ersten 30 cm). Die Laubauflage betrug durchschnittlich 0,5 cm (Oldörp et al. in press).

Diskussion

Die Fundorte in der Stadt Basel weisen im Gegensatz zu den Ergebnissen von Trautner & Schüle (1996) keine erhöhte Bodenfeuchte auf. Sie liegen aber in der Nähe von Uferbereichen: der Fundort im St. Johanns-Park ist nur durch eine Uferpromenade vom Rhein getrennt; der Fundort im Margarethenpark liegt 250 m von der Birsig, einem Rhein-Zufluss entfernt.

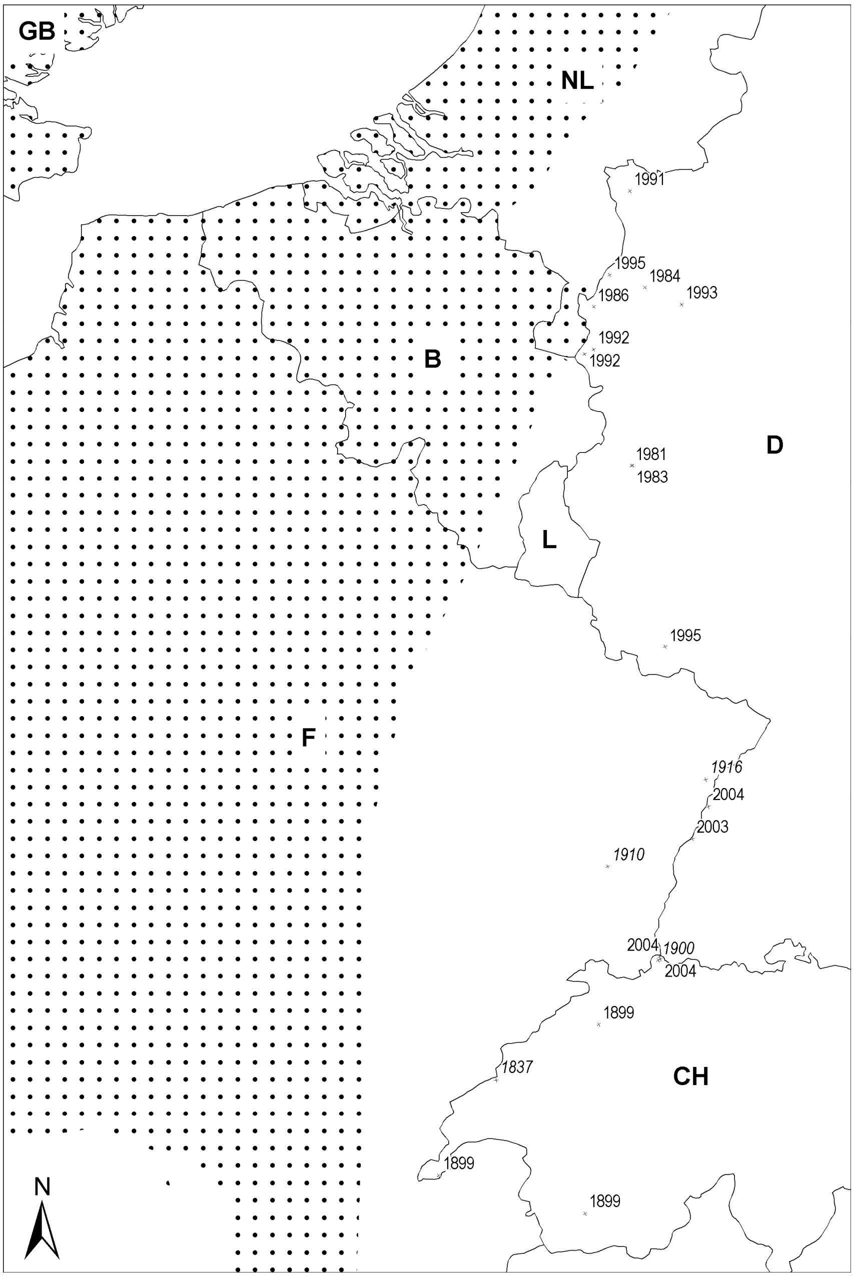

Abb. 2. Verbreitungskarte von *Leistus fulvibarbis* mit allen bekannten und aus der Literatur erwähnten Funden aus der Schweiz, Deutschland und dem Osten Frankreichs (Elsass). Normal gesetzte Jahreszahlen: sichere Nachweise; kursiv gesetzte Jahreszahlen: Nennungen ohne Belegexemplare; punktiert: bekanntes Verbreitungsgebiet (Desender 1986, Deville 1921, Jeannel 1947, Luff 1998, Mousset 1973, Turin 2000, Valemberg 1997).

Ein Widerspruch besteht zwischen der von Trautner & Schüle (1996) beobachteten Besiedlung von extensiv genutzten Bereichen und unseren Standorten auf intensiv begangenen und genutzten Flächen in städtischen Parks. Eine Präferenz für sandige Böden (vgl. Burmeister 1939) wird nur durch den Fund auf lehmigem Sand im St. Johanns-Park bestätigt. Eine

Bevorzugung von beschatteten Orten (Turin 1981) in Assoziation mit Laubwäldern (Eyre & Luff 2004) oder mit Hecken in der Agrarlandschaft (Asteraki et al. 1995) scheinen gut mit unseren Daten übereinzustimmen. Für eine verlässlichere Charakterisierung der Habitatansprüche stehen noch zuwenige Daten zur Verfügung.

Die neuen Funde in der Schweiz passen gut ins aktuelle Ausbreitungsmuster von *L. fulvibarbis* (Abb. 2). Nachdem erste Funde in der Eifel erfolgten (Koch 1990), wurde die Art Mitte der 1990er Jahre weiter südlich im Saarland beobachtet (Trautner & Schüle 1996). Die neuesten Meldungen aus Deutschland liegen noch weiter südlich, in Baden-Württemberg (Schanowski & Schiel 2004, Schanowski, pers. Mitt.). Aufgrund der neuen Funddaten aus Deutschland und der Schweiz, könnte man vermuten, dass *L. fulvibarbis* von Norden her wieder nach Basel gelangt ist. Dass die auffällige Art in einem stark besammelten Gebiet wie der Schweiz und Deutschland übersehen wurde, ist relativ unwahrscheinlich.

Weiter fällt auf, dass alle diese Funde entlang des Rheins oder nahe bei Zuflüssen des Rheins oder dessen Zuflüsse gemacht worden sind (Abb. 2), was gut mit der Vorliebe der Art für Uferbereiche übereinstimmt. Ob diese Beobachtungen eine Ostausbreitung der Art belegen, wie Trautner & Schüle (1996) vorschlagen, übereinstimmen, können wir nicht abschliessend feststellen, da in der zentralen Faunendatenbank für Frankreich keine Neumeldungen vorliegen (Muséum national d'Histoire naturelle 2003-2006). Die Überprüfung aller Neumeldungen aus den einzelnen Regionen Frankreichs wäre sicher ein interessanter Ansatz, um die Ausbreitung von *Leistus fulvibarbis* weiter zu untersuchen und anhand der neuen Fundorte mehr über die Habitatansprüche dieser interessanten Art zu erfahren.

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How do stakeholders and the legislation influence the allocation of green space on brownfield redevelopment projects? Five case studies from Switzerland, Germany and the UK

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Abstract

Disused railway sites have become a focus of redevelopment projects in many European cities. As sites that offer economically lucrative opportunities for building projects, as well as important habitats for threatened pioneer species, they are contested ground. We compared the allocation of green space in five urban development projects on disused railway sites from Switzerland, Germany and the United Kingdom. We show that distance to the city centre, the involvement of the stakeholders, and the current legislation are important in explaining the quantity and type of green space allocated. In particular, the farer away from the city centre, the higher the proportion of apartments planned and the higher the allocation of green space within the public space. Our comparison also illustrates that ‘conventional’ green spaces are of major importance for the public acceptance of urban redevelopment projects and are highlighted by developers as an important image factor. We identified three strategies to protect the valuable natural sites in such railway brownfields: protection of the pioneer habitats in-situ, reinstallation of similar habitats on roofs (ex-situ) and safeguarding of the natural process of succession. For future brownfield redevelopment projects we would encourage guidelines that take into account the special kind of nature on such sites and guarantee planning reliability for investors.

Keywords

Brownfield, legislation, nature conservation, park visions, pioneer species, railway sites, redevelopment projects, urban environment

Introduction

Disused railway sites are a focus of redevelopment projects in many European countries (Bertolini and Spit 1998). Firstly, the economically attractive location of such sites, close to or even within the central districts of cities, gives them a potentially high land value. Secondly, they often account for the largest, well-connected development areas within European metropoles (Bertolini and Spit 1998). Thirdly, such railway brownfields benefit from their relatively small reclamation costs compared to former heavy metal industry areas (Valda et al. 2004). Lastly, the economic demand to develop such sites has been intensified by the reorganisation or privatisation of national railway-groups, thereby leading to the spin-off of major real estate enterprises responsible for the development of those inner city brownfields (Michel 2005). For these reasons railway brownfields are of major interest to urban redevelopment projects.

As well as the large economic benefits, railway sites offer important secondary habitats for many rare species and especially for pioneer species (Wittig 1993; Bönsel 2000; Mattheis and Otte 1989). They provide a hot and dry microclimate due to the warmer and drier climate of cities, their large size without shade, and the dark and porous properties of ballast used in the track systems. These conditions, together with the nutrient-low substrate and the extensive maintenance of the railway tracks, form a unique habitat, comparable to the distinctive assemblages found at large gravel river banks (Allgöwer 2000). The existence of such wastelands is very important for pioneer species because the primary habitat along rivers has become so rare that all natural and semi-natural rivers are listed in Annex I of the European Union Council Directive (Council Directive 92/43/EEC 1992), the directive which regulates the protection of threatened habitats in Europe. Therefore, preservation of non-succession habitats found at railway sites is important for preservation of rare pioneer species.

If the disuse of railway sites means that maintenance and utilisation fall away, succession begins forming more common habitats. As scrubs and trees grow, the site conditions change, becoming richer in nutrients and shadow. In this process pioneer species are suppressed (Burckhardt et al. 2003). Over the years of succession, a forest will develop with characteristic species assemblages (e.g. Tree-of-heaven (*Ailanthus altissima*), Butterfly-bush (*Buddleja davidii*)). Kowarik (1995) suggests that a specific kind of nature – a '4th nature' – has been evolving within the city on brownfields. Until now, this emerging type of nature has been given little priority in nature conservation. In order to properly protect it, Kowarik encourages the implementation of a specifically '4th-nature orientated' planning law.

Urban development projects are often realised in the form of public-private partnerships, where the municipality as the 'public-partner' takes the role of the coordinator in the participation process (Blumer 2001). Experts emphasize that the acceptance of brownfield redevelopment projects on urban green space increases when citizens are given possibilities beforehand to formulate their needs and demands regarding public open space (Küry and Ritter 1998; Harnik 2000; Garvin et al. 1997).

Until now only few studies (De Sousa 2003; Harrison and Davies 2002; Burckhardt et al. 2003) show the recreation and nature conservation potentials of brownfields in cities. However, no study so far has compared the redevelopment of disused railway sites from different countries taking into account the views of some of the major stakeholder groups and the different forms of green space provided.

Here we investigate five development projects on railway brownfields by (1) comparing the quantity and quality of open and green space, (2) sketching the different views of three stakeholder groups on green space, and (3) discussing the effect of legislation on the implementation of nature priority areas on railway brownfields.

Research Methods

Information and data for the present study were gathered from railway conversion project documentation and eleven personal interviews with stakeholders and experts. The compared sites were chosen in two steps:

We first selected countries with a particularly high level of economic competitiveness and a high population density (Switzerland, Germany and the United Kingdom) because Oliver et al. (2004) show that such countries use similar definitions for brownfields. In our case, the brownfield definition of the three countries focuses on the redevelopment of underused areas, without explicitly taking into account the financial reclamation costs (for Germany and UK see Oliver et al. 2004; for Switzerland see Valda et al. 2004). In a second phase, we selected from these three countries five redevelopment projects located in five major cities (Table 1): 'Erlenmatt' Basel (CH), 'Gleisdreieck'[1] Berlin (D), 'King's Cross Central' London (UK), 'Zentrale Bahnflächen'[2] Munich (D) and 'Stadtraum Hauptbahnhof (HB)'[3] Zurich (CH). These projects fulfilled the following four conditions: they were once railway sites, they are large urban development areas within the city, their proposed use is a mixture of housing, working space, public institutions and public open space, and the implementation of the proposed projects has recently started or will do so within the next three years.

FIVE CASE STUDIES

Four of the five selected former railway sites, namely 'Erlenmatt' Basel 'Gleisdreieck' Berlin, 'King's Cross Central' London and 'Zentrale Bahnflächen' Munich, are currently the largest urban development projects of these cities (Regierungsrat Kanton Basel-Stadt 2003; www.vivico.de [28 November 2005], Bertolini and Spit 1998; Reiss-Schmidt 2002). The project 'Stadtraum HB' Zurich is not the largest, but one of considerable importance because of its location within the business district of the Swiss financial metropolis (Noser 2005). With a size of 173 hectares (Table 1), Munich's 'Zentrale Bahnflächen' is by far the largest project because it joins six sites along the old railway tracks, running in a line from the city centre to its urban fringe. This contrasts to the spatial setting of the other four cases, which are triangular in shape. Still, comparisons are allowed by the fact that

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[1] 'Gleisdreieck' in English: 'Railway Triangle'

[2] 'Zentrale Bahnflächen' in English: 'Central Railway Surface Area'

[3] 'Stadtraum HB' in English: 'City Space Main Railway Station'

the core of the ‘Zentrale Bahnflächen’ Munich, the neighbourhood ‘Arnulfpark®’ (19 ha), is similar both in form and size to ‘Erlenmatt’ in Basel.

ALLOCATION OF PLOT RATIO, PUBLIC OPEN SPACE AND GREEN SPACE IN THE FIVE CASE STUDIES

The highest plot ratio, defined as the ratio of the total floor area in a building to the area of the site on which it stands, was recorded for Berlin (max: 4), closely followed by London (max: 3.3) and Zurich (max: 2.6), while ‘Erlenmatt’ Basel, with a plot ratio of 1, is significantly lower. However, this ratio changed during the evolution of the ‘Erlenmatt’ Basel project. The first development plan for ‘Erlenmatt’ Basel foresaw a plot ratio of 1.5. In order to attract middle class residents, it was decided to lower the density to improve the quality of the new apartments to be built. The differences in plot ratios show that the less housing is planned the higher the plot ratio is (Table 1), a finding confirmed by Cabernet’s ABC-Model (www.cabernet.org.uk [28 November 2005]), which shows that high land value results in a high plot ratio.

The amount of public open space in relation to the project perimeter differs in the five projects: Four sites set aside approximately one third of the project perimeter for public open space (Table 1): the smallest amount in the ‘Stadtraum HB’ (27 %) and the highest amount in ‘Erlenmatt’ Basel (42 %). A much higher degree of open space is foreseen in the ‘Gleisdreieck’ Berlin (70 %) because a part of it must compensate for the loss of open space on the new economic centre, ‘Potsdamer Platz’ (Senat of Berlin 1994).

Regarding the allotment of green space, two groups can be identified (Table 1): The first group consists of the projects ‘Erlenmatt’ Basel, ‘Gleisdreieck’ Berlin and ‘Zentrale Bahnflächen’ Munich. In this group the allocation of green space in relation to public open space is similar, namely 75 % to 78 %. In the second group, the project ‘King’s Cross Central’ London allocates only 19 %[4] and the project ‘Stadtraum HB’ Zurich only

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[4] Own estimation based on RPS, Planning Transport & Environment (2005)

Table 1. Urban development projects on disused railway sites

City	Basel	Berlin	London	Munich	Zurich
Position of City[a]	3	1	1	3	1
Project name	Erlenmatt	Gleisdreieck	King’s Cross Central	Zentrale Bahnflächen	Stadtraum HB
Former use	Goods station and warehouses	Goods stations and marshalling yards	Mixed railway use (warehouses, sidings, marshalling yards)	Goods and container stations, marshalling yards	Mixed railway use (sidings, warehouses) and mail services
Location: distance to central business district (CBD)/city centre (CC)	Near CC	Close to CBD	Part of CBD	Gradually decreasing distance to CC	Part of CC
End of former use	2003	1945/1990	Gradually, since 1980	Gradually, since 1980	Gradually, since 1980
Realisation of development plans	2006–2020	2007–open	2007–2022	2002–open	2008–2015
Plot ratio	1	2.5–4	2.7–3.3	3.96[b]	2–2.6
Working places	2 000	No data	11 500–20 000	15 000	6 000–8 000
Apartments	700	No data	2 000	7 500	least 500
Number of working places per apartment	2.9	No data	5.75–10	2	12–16
Size (ha)	19.0	59.6	29.0	173.0	7.8
Public open space [ha]	8.0	41.6	10.5	70.0	2.1
in % to size	42%	70%	36%	40%	27%
Green space [ha]	6.0	32.5	2.0	53.6	0.34
in % to public open space	75%	78%	19%	77%	16%
Nature priority area [ha]	3.5	3.3	0.5	14.4	0.19
in % to public open space	44%	8%	5%	21%	9%

[a] Position of city in the country (population size)

[b] Plot ratio data only available for the 2nd site (Birketweg)

16 %[5] of public space to green space. In this latter group the relation between planned working places and apartments is also substantially higher than in the first group.

The vicinity to the city centre and the potential land value can explain the pattern of green space allocation. The nearer to the city centre, the higher the pressure on the public open space as a whole and on the green space in particular. The potential land value associated with such a location is a key factor in determining the pressure on the public open space and therefore the different dimensions of green space within the projects. If this pressure is very high, development projects will tend to have a high number of working places compared to apartments, which enhances the profitability of the investment. Conversely, if there is rather high proportion of apartments planned, green space gains in importance and the area set aside for it in planning is therefore also bigger.

The project ‘Stadtraum HB’ Zurich, for example, lies within the Central Business District, and although the legal background to it is similar to the legal background of ‘Erlenmatt’ Basel, it has very little projected green space. The situation of ‘King’s Cross Central’ London is comparable to that of ‘Stadtraum HB’. The site is situated within the Inner London District, near the current international centres West End and Kensington, and as the developer declares, it will be the ‘most accessible location within this world city, a major gateway and place of arrival’ (www.argentkingscross.com [14 December 2005]). On the other side, ‘Erlenmatt’ Basel, and ‘Zentrale Bahnflächen’ Munich turn out to be attractive residential areas, situated near the city centre but not within it. On these latter two sites a greater emphasis is given to green space, at least as far as the size of the allocated green area is concerned.

ALLOCATION OF NATURE PRIORITY AREAS IN THE FIVE CASE STUDIES

In addition to the surface area allotted to green space, we analysed the extent to which the flora and fauna found at each railway site was taken into account. As far as the breakdown of the green space is concerned,

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[5] Own estimation based on Stadt Zurich and SBB (2005)

three categories can be distinguished: conventional green spaces (e.g. parks and avenues), nature conservation areas and nature reserves. The latter two are distinguished by their degree of legal protection and are here defined as nature priority areas.

We expected that on all five sites a similar proportion of nature priority areas would be applied according to the presence of special habitats and rare species (see the extensive inventories and descriptions by Blattner 1985; Burckhardt et al. 2003; Langer et al. 2005; Kowarik 1995; RPS Planning Transport & Environment 2005; Bichlmeier 1990; Leutenegger et al. 2004). However, a more detailed view of the different types of green space planned on each of these conversion sites reveals remarkable differences in the extent of the green space set aside, specifically for the two categories of nature priority areas. The total surface for nature priority areas ranges from 0.19 ha in Zurich to, London: 0.5 ha, Berlin: 3.3 ha, Basel: 3.5 ha, Munich: 14.40 ha.

Only in the project ‘Erlenmatt’ Basel a nature reserve, covering a surface of 1.9 ha was allocated. The aim of this reserve is to protect pioneer species. Therefore mowing, weeding or other measures of disturbance have to be carried out in order to stop succession processes. In addition to this strongly protected nature reserve, a nature conservation area will be established. However, the final concept for this area is not yet defined as revealed in expert interviews. The whole nature priority area accounts for 44 % of the public open space. Furthermore, the law prescribes that investors have to install green roofs (Amstutz 2001) in an ecologically suitable manner (Kaupp et al. 2004; Lenzin 2006), e.g. roofs consisting of gravel that is thick enough to provide a pioneer habitat.

The biotope mapping of the project perimeter of the ‘Gleisdreieck’ Berlin declares a small forest – ‘Wäldchen’ – together with the adjacent sand lawn and ruderal meadows as the habitats most valuable for nature conservation (Langer et al. 2005). Most of the other open wasteland areas in Berlin have been gravely damaged by their use as deposit areas for construction materials during the creation of the nearby Postdamer Platz. In their recommendations, the experts propose that the forest ‘Wäldchen’ stays closed to the general public and only be opened for guided tours (Langer et al. 2005). The competition description for the landscape planning part of the project consequently requires a sensitive treatment of this area; however, it does not guarantee any further protection. According to the concept in the competition description, the 'Wäldchen' should be a 'slow' park in which contemplation and the experiencing of nature have priority (Pütz et al. 2005). In total this area allocated to the little forest accounts for 8 % of the overall open space and can be classified as a nature conservation area.

According to the project documents for 'Zentrale Bahnflächen' Munich, the mosaic of typical railway habitats should be conserved. Twenty-four hectares of the perimeter have been identified as valuable railway biotopes and have been mapped to document the state of these areas (Kleyer 2002). Out of these mapped 24 hectares, 60 % have to be conserved within the project perimeter and 40 % will be realised next to the project perimeter (Hutter-v. Knorring et al. 2005). On the second site of the 'Zentrale Bahnflächen' a so called 'Pioneer Park' (1.8 ha) will be implemented to protect pioneer species with special measures such as mowing and grubbing. All these areas shall stay accessible to the public, protected only by visitor's guidance. These extremely large railway habitats can also be classified as nature conservation areas and account for 27 % of the total green space.

The projects 'Stadtraum HB' Zurich and 'King's Cross Central' London are rather similar. In the 'Stadtraum HB' the nature conservation area will be a large strip following the existing railway tracks (Baumgartner et al. 2004). This line should serve as ecological corridor for typical species of railway habitats, like the wall lizard (*Podarcis muralis*) or wild bees. The estimated contribution of this nature conservation area to the public open space is 9 %. With the redevelopment of this site, large green and brown roofs on the old warehouses will be lost. Whether the new houses have to be greened is not yet clear, and how this will increase the percentage allocated to nature priority is still open.

Bordering 'King's Cross Central' is the nature reserve 'Camley Street Natural Park', a forest that developed naturally on a former coal store site. It was established as a protected area in 1988. It is excluded from the urban development project and therefore does not show in our statistics but is important when considering green space allocation in this site. The canal on the perimeter of the 'King's Cross Central' is classified as a nature conservation site. This part of Regent's Canal constitutes a nature conservation area of 0.5 ha (5 % of the public open space). Furthermore, for green space allocation the RPS Planning Transport and Environment (2005) stipulates that 15 % of the roof surface has to be green or brown roofs. This should compensate for the wastelands on 'King's Cross Central' destroyed during the construction work of the new Eurotrain Terminal.

VIEWS OF THREE STAKEHOLDER GROUPS

Stakeholders in redevelopment projects hold various and changing perspectives on the issue of green space and its various sub-types. Such views are closely related to the motives of the different stakeholder groups. We considered here three groups: firstly, developers, secondly citizens and residents, and finally nature conservation experts from the administration or non-governmental agencies.

Developers generally draw attention to the specific urban qualities of the green spaces provided within the development projects and thereby underline the value added for the planned housing, office and retail estate. In the project documentations they describe green spaces as a major contribution to the overall image of a project. For example, in London the developers accentuate that they are 'keen to capitalise upon the canal's positive contribution to 'King's Cross Central' and see redevelopment as an opportunity to bring life to the canal, enhance its character, wildlife value and recreational use and improve access and safety' (www.argentkingcross.com [14 December 2005]). The slogans created for the advertising of the sites in Basel, Berlin and Munich by Vivico Real Estate address the symbolic value of green areas even more directly: 'Erlenmatt' Basel is promoted as the 'Green Lung in the Heart of the City', whilst in Berlin the motto is 'Living around the Park' and in Munich it is 'Munich is Moving into the Park' (www.vivico.de [28 November 2005]). Urban green spaces therefore obviously seem to matter as a pull-factor for the different clients developers have in mind. In their research on urbanisation processes of western post-fordist cities, Keil and Graham (1998, p. 119) came to the conclusion that ‘nature has become a major discursive element of the production of urban space’. However, the images that are foregrounded in the compared projects refer to a certain kind of urban parks: public green spaces with a highly representative character, dominated by intensively cared-for green lawns, footpaths and avenues. In interviews experts from NGOs and the administration have questioned these aestheticized images that developers use to promote ‘Erlenmatt’ Basel. Because 58 % of the park will serve as a nature priority area, it will by no means meet the green and colourful renderings, displayed on the websites of the real estate agency (Figure 1).

Figure 1a. Present situation on the project site ‘Erlenmatt’ in Basel, with the former DB-Administration-Building in the background. Photo: W. Altherr.

Figure 1b. Visualisation of stakeholder view of the redevelopment project ‘Erlenmatt’ in Basel with the former DB-administration-building in the background Photomontage: Vivico Real Estate www.vivico.de [21 December 2005].

The second group of stakeholders, namely citizens in general and local residents, stresses the immediate and daily use and therefore the practical value of the green spaces. These stakeholders’ concerns are brought to public mainly through different forms of citizen participation. For example, in Berlin the framework for the landscape design contest ‘Gleisdreieck’ (launched mid December 2005) largely rests on the residents’ demands regarding the future park (Pütz et al. 2005). Citizens’ ideas for the park were collected by the municipality through a representative survey with residents of the neighbouring areas, group-interviews, and a three-week long online-discussion (www.gleisdreieck-dialog.de [8 December 2005]). The results show that even though nature is highlighted as an important value, interviewees' concerns mainly relate to questions of daily use. This includes having places to go for a walk, play with the children, to read and rest, meet people for a picnic and play sports (Klaphake et al. 2005). Furthermore, the result of the latter study emphasised that the demand for green space is very high due to the lack of sufficient green areas in the surrounding neighbourhoods. Accordingly, the park 'Gleisdreieck' is likely to face intensive use.

A similar picture to that in Berlin can be drawn for 'Erlenmatt' Basel. Our representative survey on the use of green spaces in Basel carried out in 2004 (Oldörp et al. in press) has come to conclusions very similar to those of the municipality survey conducted in Berlin. This was reflected in the run-up to an official public poll on the redesignation of the area in which the image of a conventional neighbourhood park allowing intensive use was widely promoted in the press and on campaign posters. Furthermore, it was emphasised that 'Erlenmatt' will offer a major relief for the nearby neighbourhoods because those areas offering only 3 sqm public open space per resident by no means meet the standard 9 sqm recommended by Basel's Planning Department (Schwarze and Abegg 2004).

The projects in Zurich and London give a slightly different picture. Here the debate to date does not focus on questions concerning public open or green space. Instead, questions of social housing or the support for small local businesses are emphasized. The reason for this is that restructuring measures within city centres often have a major impact and direct influence on the living conditions of the residents and local businesses on or close to the redevelopment site (Hamnett 2003; Vicario and Martínez Monje 2003). In the case of 'King's Cross Central', the reason why green space is not a major issue furthermore seems to derive from the fact that the most important and most-valued green space, the 'Camley Street Natural Park', does not lie within the project's boundaries and therefore does not face major threats. In Zurich the explanation for the little attention to open public and green space might be that the 'Stadtraum HB' lies close to the park ‘Platzspitz’ and to the green space ‘Kasernenwiese’, both major green spaces in the city centre.

The images with which both stakeholder groups deal show that green space is an important issue, but that questions of nature-orientated design do not necessarily occupy the developers or the residents. Kowarik’s (1995) general stipulation, namely that the potential and value of new forms of urban nature found on disused railway sites should receive major consideration in the redevelopment of brownfields to greenfields, seems of minor priority to both groups.

Evidence from interviewed experts confirms that it is largely the attitudes of the third stakeholder group, namely nature conservation experts from the administration or non-governmental agencies, which favour views such as Kowarik’s and highlight the value of new forms of urban nature in the redevelopment of railway brownfields. If the significance of such new forms of urban nature is addressed within participation processes – as in Basel and Berlin – this was mostly done by experts or specific environmentalist groups engaged in such participatory processes. De Sousa (2003, p. 192), in his research on Toronto’s (CA) conversion of urban brownfields into greenfields, comes to similar conclusions regarding the attitudes of this third group of stakeholders. He notes that ‘Support for ecological restoration projects typically came from established community-based environmental groups (...) while support for green space in under-serviced neighbourhoods typically came from smaller, ad-hoc groups that were united by a community leader (or leaders).’ De Sousa therefore adds a cautionary note to the importance of public participation, warning that ‘the downsides of this extensive community involvement at many sites, however, was the emergence of a debate over what type of green space was to be implemented. (...) several groups pressured the city to turn the site into an ecological habitat, while others lobbied for soccer fields, baseball diamonds, a marina and other recreational use.’ (De Sousa 2003, p. 192).

THE ROLE OF LEGISLATION

A comparison of the five sites shows that legislation proves to be the most important instrument for the implementation of such environmental demands. This is the case even though none of the three countries (Switzerland, Germany or UK) have national legislation, which specifically regulates nature conservation on brownfields formerly used by the railway. Significant differences in the legal situation can be found between the different countries, leading to strongly divergent approaches regarding the degree and type of conservation.

In Switzerland, the occurrence of red list species turns out to be of decisive importance. Red list species such as the Spotted Knapweed (*Centaurea stoebe)* or the Blue Sand-Grasshopper (*Sphingonotus caerulans)* within the boundaries of 'Erlenmatt' Basel have been the basis for the establishment of a nature reserve. Similarly, with 'Stadtraum HB' Zurich, the occurrence of red list species led to the creation of a nature protection corridor along the railroads.

In Germany, 'Gleisdreieck' Berlin hosts a significant number of red list species too, which in Switzerland could lead to vast protection measures. This, however, will not be the case in Berlin because protection measures in Germany relate to the appearance of protected habitat types instead. In Germany red lists are merely guidelines and not legally binding instruments. On 'Gleisdreieck' Berlin, the damage caused during the development of the 'Potsdamer Platz' has left only one protected habitat type, next to the aforementioned 'Wäldchen'. Therefore the framework to the landscape competition did not prescribe any additional protected areas.

The 'King's Cross Central' London redevelopment project turns out to be the only site where a legally binding law related to the transformation of brownfields formerly used by the railways applies. This is due to the London Biodiversity Action Plan (LBAP) proposed in 2001, which promotes 'the survival of rare and valued species associated with some 15 London habitats such as woodland, heathland, railway linesides, private gardens, cemeteries and wastelands' (Harrison and Davis 2002, p. 95). However, in contrast to sites such as 'Zentrale Bahnflächen' Munich, in the case of 'King's Cross Central' the natural habitats required by the legislation will not be maintained or reinstalled at their original site. Instead, the legally prescribed natural habitats will be located high above ground. The immense pressure on potential usable land, which is responsible for the minor share allocated to open public space and green space at ‘King’s Cross Central’, has left the rooftops as remaining sites for natural conservation measurements. These roofs are planned as ‘brown roofs’ – a term referring to the concept of ecological greening of roofs developed in Basel (Brenneisen 2003).

Conclusion

Urban brownfields formerly used by railways represent important urban redevelopment sites in Europe. *In size*: four out of the five sites analysed represent the largest urban developments within the city; *in economic terms*: due to their central position and low reclamation cost, these brownfields have a high land value and are subject to building or planning processes; *in ecological terms*: the disused railway sites do not only contain many valuable species but also a specific form of urban nature and therefore provide a new kind of green space in the city.

Public open space and urban green space are differently allocated in our five case studies. The allocation of green space within the public open space was considerably lower on redevelopment sites in the city centre, where a high proportion of working places is evident. Therefore, we can conclude that on these sites green space is given a lower priority than on sites with a more residential character, where the higher proportion of green space enhances the attractiveness for living there.

We have illustrated that the priorities for the use of green space differs markedly between the stakeholders: To *developers* green space is of major importance in the marketing of the real estate. To *citizens and residents* they provide the possibility for leisure and for intensive use, such as playing sport or social gathering. The supply of such green spaces leads to higher acceptance rates of large urban development projects. The type of green space the developers as well as the citizens and residents have in mind is most often a conventional green park. *Nature conservation experts* therefore seem to be the main stakeholders concerned with the conservation of brownfield habitats.

Even among environmentalists the appreciation of this 4th nature (Kowarik 1995) is a relatively new phenomenon and the management practices are not yet well established. Here, we identify three management approaches: One option is protecting the habitats for pioneer species on existing railway habitats (Basel, Munich). This requires a high level of maintenance to prevent succession. A second option is installing roofs with a local gravel substrate for pioneer habitats (Basel, London). This reduces maintenance intensity because of a low substrate thickness and a low re-colonisation rate of species. However, such roof habitats have the disadvantage of inaccessibility because of their height above ground, and are therefore difficult for citizens to view or encounter. A third option is to allow the process of succession to change the character of the habitats. Most often the vegetation on such sites develops into an urban forest (Berlin, London). This has the disadvantage of losing threatened pioneer species. All three conservation strategies are being implemented through the use of legal policies, however the level and degree of protection and the type of habitat protected is not the same.

In the future, specific planning laws and strategies should be considered. They could range from laws that legislate the strong protection of nature reserves, which includes barring any public access, to guidelines that regulate a more relaxed status as nature conservation areas, where the public are invited to interact with nature. However, we would like to highlight the importance of a mixed approach, which will be followed in the case of ‘Erlenmatt’ Basel and the ‘Pioneer Park’ on the ‘Zentrale Bahnflächen’ in Munich. For people living in cities the implementation of such policies in the allocation of park areas could provide important sites for an encounter with nature on closer and more direct terms than conventional policies would permit. For investors such policies guarantee planning reliability and for nature conservation experts they give a legally binding protection to the pioneer habitats.

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GENERAL DISCUSSION AND CONCLUSIONS

Effects of urbanisation and urban areas on biodiversity

In this general discussion, we would like to illustrate the main effects of urbanisation and urban areas on the three biodiversity levels outlined in the introduction. The consequences of habitat alteration, isolation and loss are discussed in the studies on genetic diversity, species diversity and habitat diversity. The effect of succession was investigated in an interdisciplinary study on railway redevelopment projects in five cities in Europe. Management practice and the associated level of disturbance were an important factor in the study on arthropod diversity in urban forest patches. The influence of introduced species on diversity and possible effects of homogenisation are discussed here. We look back at the role the urban heat island effect played in our studies. To finish this discussion, we look briefly at the observed interaction between biodiversity and the inhabitants and at possible political measures to implement biodiversity in the city. Finally, we present the conclusions of this thesis.

EFFECTS OF HABITAT ALTERATION, ISOLATION AND LOSS

The effect of habitat alteration was measured in the study on arthropod diversity in the form of trampling intensity, but this factor did not affect arthropod diversity or assemblages. This result was confirmed by two studies on the effect of disturbance through trampling on ground beetles (Grandchamp et al. 2000, Lehvävirta et al. 2006).

The result from the study on genetic diversity showed that the habitat connectivity for wall lizard can be provided by railway tracks and therefore prevent isolation. The connectivity of ‘human-made’ corridors, like railway tracks, can even be higher than more natural structures, like river banks. This result confirms the value of a species-specific analyse of the habitat connectivity to consider the effectiveness of corridors (Lindenmayer & Fischer 2007). On the other hand, corridors can also be means of dispersal for introduced species, as it is the case for the wall lizard in North-America

(Hedeen & Hedeen 1999). If such species become invasive, it is important to know its dispersal patterns (Allan et al. 2006).

The reality of habitat loss is emphasized by the fact that on one investigated site 75 % of the former wall lizard habitat is destroyed, due to an urban redevelopment project, by the time of writing. Such urban redevelopment projects are often planned on disused industrial sites (Breuste 2004). We demonstrated that the allocation of green space were different; although, the ecological situation on all investigated sites were similar. To reduce the complete habitat loss on these disused railway sites, specific planning laws and strategies should be considered. They could range from laws that legislate the strong protection of nature reserves, which includes barring any public access, to guidelines that regulate a more relaxed status as nature conservation areas, where the public is invited to interact with nature.

SUCCESSION AND MANAGEMENT OF URBAN HABITATS

The thematic of succession and disturbance is of major importance in the conservation of the described brownfields. Their ecological particularity developed because of regular disturbance in form of mowing or weeding. If the disuse of railway sites means that maintenance and utilisation fall away, succession begins forming more common habitats. In this process, pioneer species are suppressed. If we want to preserve these pioneer habitats, a mean of maintenance has to be developed. Because maintenance of any kind is expensive, other measures, like ex-situ on roofs or the natural succession can be alternatives for these urban ecosystems.

The level of disturbance was a key factor in distinguishing the urban forest patches. The small urban forest patches were characterised by their high intensity of management. Every year or every second year shrubs and underwood are cut, whereas the larger forest patches were only maintained every four to ten years. Although, this considerable different treatment, species richness did not differ. In contrast, the species assemblages changed significantly between small, intensively managed forest patches compared to large, extensively managed forest patches. To distinguish the effect of size and management intensity, these two effects should be separated in a future study.

INTRODUCED SPECIES AND HOMOGENISATION

One focal effect of urbanisation identified in the general introduction was homogenisation, which replaces native species by non-indigenous species introduced by humans (McKinney & Lockwood 1999). However, the species list from urban forest patches (Chapter II, part 1, Annex) did not reveal exotic, invasive or accidentally introduced species. Furthermore, the findings of e.g. *Carabus coriaceus* were rather indicators for a well adapted forest fauna. We can therefore exclude this effect for the urban forests investigated.

Nevertheless, the accidental introduction of species was discussed in the wall lizard study because some authors suggested that this species is introduced by railway goods transport of vegetables and fruits (Deichsel & Gist 2001, Hohl 2003). Although, we could not detect introduced species in the genetic population pattern, the question remains interesting. To test this assumption more profoundly, we suggest sampling wall lizards from their main distribution centres in Europe, which are also centres of exportation for vegetables and fruits, and compare these samples with our samples.

URBAN HEAT ISLAND EFFECT

The urban heat island effect is one important factor for the establishment of wall lizards in the city of Basel. Combined with suitable habitats, wall lizards can form large populations. The high frequency of wall lizard in urban areas has also been reported from Zurich, where the population of wall lizards on the railway tracks of the main station is estimated to be the largest population North of the Alps (Felten et al. 2007).

Urban forests and urban green space, on the other hand, are habitats with a considerably lower mean temperature than the rest of the city (Shashua-Bar & Hoffman 2000). This general observation was confirmed for Basel (Lehmann 2006) in the frame of our project ‘Natur-Begegnung-Stadt

Basel'. The mean temperatures were significantly reduced in urban green spaces (-0.2–2° C); on one site, a reduction of 6°C compared to the surrounding environment was observed. The lower mean temperatures in urban forests confirmed their importance for the climate of cities. These findings are reflected in the high amount of forest species in the observed taxa of spiders and ground beetles.

BIODIVERSITY IN THE CITY AND THE INHABITANS

Urban ecosystems can be an important source for experience and education of the inhabitants of cities (Miller 2005, Niemelä 1999, Schemel 2001). Furthermore, their function for the well-being of the citizens is more and more recognised (Chiesura 2004, Grahn & Stigsdotter 2003). The importance of urban green space was confirmed by the results from the study on brownfield development and the results from a representative survey in our project 'Natur-Begegnung-Stadt Basel' (Oldörp et al. in press). People appreciated 'conventional' urban green space e.g. a classical landscape park. They did not associate the nature of the fourth type, which evolved on urban-industrial sites, as green space. These observations were confirmed by a study from Eastern Germany (Breuste 2004), where people preferred conventional parks to nearby open space that developed on disused industrial sites. Therefore, a higher amount of popular information on this fourth type of nature may be necessary along with the accessibility of these sites for the public to enable encounter with this type of nature. The habitat closest associated with nature in the urban environment was the urban forest (Oldörp et al. in press). Therefore, urban forests have a high potential for education and experience with nature. This potential was also identified in the study of Alvey (2006) and Konijnendijk (2005).

BIODIVERSITY IN THE CITY AND THE POLITICAL SITUATION IN SWITZERLAND

Conversion of rural land to urban settlements is a fast and world wide process. In Switzerland, the amount of converted land is particularly high with a rate of 1 m^2 per second of converted land. The importance of the topic in the Swiss politic is underlined by the launch of a political initiative 'Landschaftsinitiative – Raum für Mensch und Natur' (Initiativkomitee

2007). The goal of this initiative is the stronger implementation of a sustainable land use and the prevention of further conversion of rural land to urban settlement areas.

However, where the process of land conversion can not be stopped or reduced, the urban nature in cities and suburbs of cities becomes more and more important (Niemelä 1999). In Switzerland, a high quality of nature in urban areas is a key concept of urban development (Stremlow et al. 2003). The implementation should be realised on different levels. In many publications from the urban authorities, inhabitants are encouraged to follow ecological consideration for their private green and in their allotment gardens (Eigenmann 2003). The private foundation ‘Natur & Wirtschaft’ certifies companies if the design and maintenance of the companies area follow ecological guidelines. On a cantonal level, Basel-Stadt has established a foundation to enhance the quality and quantity of open green space. An important legal instrument on the national level is the ordinance for ecological compensation in settled areas (Nature and Cultural Heritage Protection Ordinance Act 15). The consequent application of this law could be a good measure to secure nature in urban areas.

Conclusions

The high rate of land conversion from rural land to urban settlements brings the question of urban nature into focus. Urban areas harbour diverse types of nature from almost natural habitats, urban forests and parks, to habitats which have evolved on previously urban-industrial sites. All these ecosystems are associated with characteristic species assemblages. To maintain this urban biodiversity for the inhabitants and for its own biological value, better understanding of ecological processes is necessary. The multiscale approach to study the effects of urban areas and urbanisation on biodiversity outlined in this thesis has provided important insights. The main effects influencing biodiversity were habitat alteration, isolation and loss. As demonstrated in this thesis, biodiversity in the city can be promoted at all levels from genes to habitats by scientifically based management strategies. Humans live mainly in cities and their behaviour and decisions influence all processes in an urban environment, including urban nature. Therefore, the quality and quantity of urban nature can only be enhanced with an interdisciplinary approach, where the social sciences, economy, law and ecology work together. This approach would be not only beneficial for the biodiversity in urban areas, but also for the inhabitants of cities.

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Population genetic structure of the wall lizard (*Podarcis muralis*) in an urban environment

Other publication

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3rd poster award at the ‘Jahrestreffen des Arbeitskreises Biogeographie’ im VGDH, 18.–21. Mai 2006

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