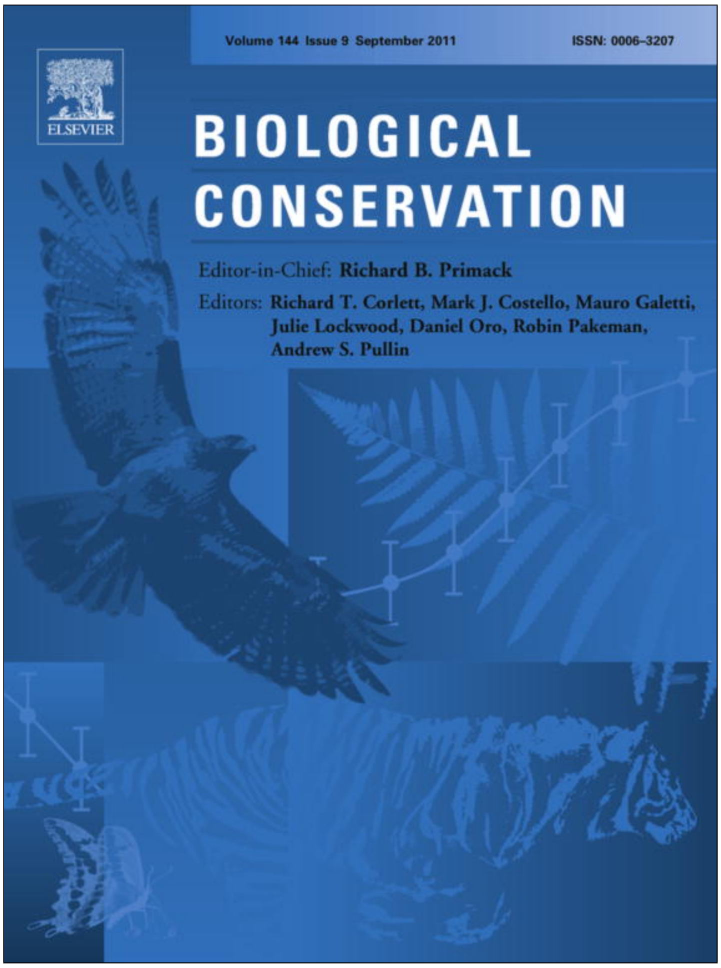
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Temporary conservation for urban biodiversity

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ABSTRACT

Urban habitats, particularly wastelands and brownfields, maintain rich biodiversity and offer habitat for many species, even rare and endangered taxa. However, such habitats are also under socio-economic pressures due to redevelopment for housing and industrial uses. In order to maintain urban biodiversity, it is currently unknown how much open area must be preserved and whether conservation is possible without complete exclusion from economic development. In this study, we applied a simulation model based on species distribution models for plants, grasshoppers, and leafhoppers to investigate planning options for urban conservation with special focus on business areas. Altogether, we modelled the occurrence of 81 species of the urban species pool and analysed settings of different proportions of open sites, different habitat turnover times, and different lot sizes. Our simulations demonstrated that dynamic land use supports urban biodiversity in terms of species richness and rarity. Setting aside brownfields before redevelopment for a period of on average 15 years supported the highest conservation value. Consequently, we recommend integrating the concept of 'temporary conservation' into urban planning for industrial and business areas. This concept requires habitat to be destroyed by redeveloping brownfield sites to built-up sites, but simultaneously creating new open spaces due to abandonment of urban land uses at other locations. This maintains a spatio-temporal mosaic of different successional stages ranging from pioneer to pre-forest communities.

1. Introduction

In general, cities possess a rich biodiversity of flora and fauna (Godefroid and Koedam, 2007; Pickett et al., 2001; Rebele, 1994), which is distributed over various types of open space, including maintained parks and gardens, as well as informal habitats such as ruderal and derelict sites (Venn and Niemelä, 2004). These landscapes form a complex spatio-temporal mosaic of different habitat types, characterised by varied and altered climatic conditions and water and nutrient fluxes (Wilby and Perry, 2006). Consequently, these areas comprise unique urban communities (Alberti et al., 2003).

Among the most valuable urban habitats are brownfield sites, composed of derelict land, abandoned railway tracks, landfills, and previously developed sites. These sites often support a rich flora and fauna that include rare species (Eyre et al., 2003; Maurer et al., 2000; Small et al., 2003). They offer heterogeneous habitats of different successional stages, which are ephemeral, mostly undisturbed, and unmanaged. Despite their ecological value brownfields are often ignored in urban conservation planning (Harrison and Davies, 2002; Muratet et al., 2007) and receive much less attention by urban ecologists than parks and gardens (e.g. Smith et al., 2006a,b). Moreover, a current paradigm of urban planning indicates that brownfield sites should be a priority over greenfield sites (i.e. sites outside cities) for new housing and industry development (DCLG, 2000; Pauleit et al., 2005). While this paradigm is certainly useful to restrict urban sprawl, it is in conflict with any goal to preserve urban biodiversity.

This study introduces the concept of temporary biodiversity and temporary building and assesses its efficacy. As a new management tool, this concept allows for both an urban renaissance and biodiversity conservation on brownfield sites. This approach views the urban habitat as a spatio-temporal mosaic of developed and abandoned sites, with recolonisation of brownfields by plants and animals from adjacent habitats and future redevelopment of brownfields for housing or industry. Currently, little is known regarding turnover rates from developed to brownfield to developed sites. However, evidence suggests that the duration of use of industrial buildings has declined in recent years and will continue in the future due to short-term, fast-moving markets and new economic trends (Hassler and Kohler, 2004).

From a species point of view, brownfield emergence and loss result in spatially unstable habitat conditions for plants and

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animals. Additionally, undisturbed brownfield habitats are characterised by temporal variations in habitat suitability due to changes in vegetation structure during succession (Schadek et al., 2009). Consequently, habitat quality is not only a function of abiotic conditions, but is also rendered by successional change during the lifetime of an individual brownfield site. Species can only persist in such mosaic cycles if they are able to track the spatial and temporal shifts in habitat quality (Kleyer et al., 2007).

Here, we analyse the consequences of these shifts in spatio-temporal habitat availability and quality in the framework of temporary biodiversity and temporary building. We evaluate two alternative hypotheses: (1) undisturbed succession (no turnover) on brownfields provides increased biodiversity, or (2) habitat turnover due to periodic rebuilding and demolition increases biodiversity. If hypothesis (2) holds we assess the spatial and temporal turnover rate that supports the highest biodiversity. To test these hypotheses, we applied a simulation model based on species distribution models (SDMs), which have become an important tool in ecology as well as in conservation biology in recent years (Guisan and Thuiller, 2005). In this multi species approach (Garden et al., 2006) we related species occurrences (plants, leafhoppers, and grasshoppers) with abiotic soil conditions, landscape context variables, and successional site ages based on field data. We extrapolated the SDMs from plot scale to landscape scale and combined single species response to community response. Landscape scenarios of different habitat configuration in space and time (Rudner et al., 2007; Schröder et al., 2008) were assessed to derive recommendations for maintaining biodiversity of urban industrial and business areas. We subsequently provide general guidelines for practical urban conservation planning (Opdam et al., 2002) by evaluating the concept of 'temporary conservation' and discussing its implications for urban planning.

Table 1
Modelled plant (left) and insect (right) species and their rarity classification: c – common, r – rare.

Plant species	Rarity	Insect species	Rarity
Achillea millefolium	c	*Aphrodes makarovi*	c
Agrostis tenuis	c	*Arocephalus longiceps*	c
Arabidopsis thaliana	c	*Arthaldeus pascuellus*	c
Arenaria serpyllifolia	c	*Athysanus argentarius*	c
Arrhenatherum elatius	c	*Balclutha punctata*	c
Artemisia vulgaris	c	*Cicadella viridis*	c
Betula pendula	c	*Cicadula quadrinotata*	c
Bromus sterilis	r	*Cixius nervosus*	c
Cerastium holosteoides	c	*Dikraneura variata*	c
Chenopodium album	c	*Doratura homophyla*	r
Cirsium arvense	c	*Doratura impudica*	r
Cirsium vulgare	c	*Elymana sulphurella*	c
Conyza canadensis	c	*Empoasca vitis*	c
Corynephorus canescens	c	*Errastunus ocellaris*	c
Dactylis glomerata	c	*Euscelis incisus*	c
Deschampsia cespitosa	c	*Fagocyba cruenta*	c
Festuca rubra	c	*Graphocraerus ventralis*	c
Holcus lanatus	c	*Jassargus pseudocellaris*	c
Hypericum perforatum	c	*Javesella pellucida*	c
Lolium perenne	c	*Kosswigianella exigua*	c
Picris hieracioides	r	*Macropsis prasina*	c
Plantago lanceolata	c	*Macrosteles cristatus*	c
Plantago major	c	*Macrosteles laevis*	c
Poa annua	c	*Macrosteles ossiannilssoni*	c
Poa compressa	r	*Macrosteles quadripunctulatus*	c
Poa pratensis	c	*Macrosteles sexnotatus*	c
Poa trivialis	c	*Mocuellus collinus*	c
Rumex acetosella	c	*Neoaliturus fenestratus*	c
Saxifraga tridactylites	r	*Neophilaenus minor*	r
Senecio inaequidens	r	*Ophiola decumana*	r
Sisymbrium altissimum	c	*Psammotettix confinis*	c
Taraxacum officinale	c	*Psammotettix excisus*	r
Trifolium repens	c	*Psammotettix nodosus*	c
Tripleurospermum perforatum	c	*Rhopalopyx vitripennis*	c
Veronica arvensis	c	*Ribautodelphax collina*	c
Vicia angustifolia	c	*Zyginidia scutellaris*	c
Vicia hirsuta	c		
Vulpia myuros	r	*Chorthippus albomarginatus*	c
		Chorthippus biguttulus	c
		Chorthippus mollis	r
		Metrioptera roeseli	r
		Myrmeleotettix maculatus	r
		Oedipoda caerulescens	r
		Tetrix tenuicornis	r

2. Methods

2.1. Species distribution models and sampling plots

Species distribution models (SDMs) are regression models that relate species' incidence or abundance to environmental predictors (Guisan and Zimmermann, 2000). A widely applied modelling approach is based on generalised linear models (GLMs) with a logistic link function (Rushton et al., 2004). The occurrence probability (Y) of a species is given by

$$Y = \frac{1}{1 + e^{-(\beta_0 + \beta_1 \cdot X_1 + \cdots + \beta_n \cdot X_n)}} \quad (1)$$

with the predictor variables X_i, the intercept β_0, and the coefficients β_i. A bell-shaped relationship between Y and X can be described by introduction of a quadratic term.

We built SDMs for 38 plant species and 43 insect species (leafhoppers, grasshoppers and one bush-cricket (*Metrioptera roeseli*), hereafter referred to as grasshopper; Table 1) based on species incidence data collected at 133 sampling plots on brownfield sites (derelict sites, previously developed land, and abandoned railroads) in Bremen, north-west Germany in 2003. Vegetation at the sampling plots ranged from pioneer communities to tall perennial herbaceous and shrub or pre-forest communities on relatively dry, sandy soils. All species were either natives or thoroughly naturalised neophytes. The model building was based on multi-model inference (Burnham and Anderson, 2002) and included an internal validation step. A detailed description of sampling design can be found in Schadek et al. (2009) and Strauss and Biedermann (2006), and for details on the SDM building procedure and evaluation see Kattwinkel et al. (2009) and the Electronic appendix. Predictor variables (Electronic appendix, Table A1) included soil properties and successional site age, and for the insect models, vegetation parameters. Time since succession initiation (site age) was derived from an aerial photograph time series. Additionally, landscape context variables were built into the models to account for processes affecting plant and insect species at larger scales then that of a sampling plot. These variables described the proportion of vegetation types (e.g. dense or sparse, low or high) and site age classes within different radii around each sampling plot (25, 50, 75, 100 and 200 m). Due to the interaction between vegetation and insect occurrence, vegetation structure at the plot level and vegetation type at the landscape level (i.e. surrounding the plot) were important predictors of insect occurrence (Strauss and Biedermann, 2006). In order to predict insect occurrence these parameters needed to be known. Thus, we used the predicted plant occurrences as a proxy for vegetations structure and vegetation type using PLS regressions and classification trees, respectively (Boulesteix, 2004; Venables and Ripley, 1999). Details on that procedure can be found in the Electronic appendix.

SDMs represent static statistic models relating species occurrence to the present environment (Guisan and Zimmermann, 2000). They do not account for past states unless real time series data are available. As this was not the case, we used space-for-time substitution (Pickett, 1989) to account for the temporal development due to succession. Consequently, site age was an important predictor in the SDMs. However, the predictions of SDMs were independent of those of the previous time steps because these models do not account for population dynamics.

2.2. Case study area and modelling shell

While the sampling plots for building the SDMs were situated at brownfield sites scattered all over Bremen, the case study area for the landscape scale modelling consisted of a single industrial area in the south-west of Bremen. This area covered approximately 550 ha and was surrounded by marshlands used for dairy farming (Fig. 1). The industrial area was used mainly by logistics enterprises. It was established in 1985 by successively filling in wet grasslands with sandy material. A smaller part of the area had already been developed in 1974. The sandy landfills resulted in relatively homogeneous soil properties. Approximately 45% of the area was not developed in 2006 and consisted of open spaces at various successional stages, ranging from bare soil to pre-forest vegetation.

We developed a modelling shell to upscale the SDMs from a plot scale to a landscape scale in order to analyse the effects of: (1) proportion of developed lots, (2) turnover rate of built-up lots to brownfields and vice versa, and (3) lot size pattern on single species and on biodiversity measured as species richness. These three characteristics were combined to scenarios.

In the modelling study, we assumed that the spatial ratio of open to developed sites and the temporal turnover rate remained constant over time in each scenario. Such a situation results in a constant distribution of site ages over time. It can be described by an exponential frequency distribution of site ages with many young sites and a few old ones. The turnover rate determines the ratio between young to older sites and thus also the mean site age. We assumed that random sites were abandoned or developed. Consequently, the spatial pattern of a following time step was identical to another replicate random configuration with equally proportioned land use and the same turnover rate. Thus, as both SDMs predictions and the spatio-temporal land use configuration were independent of past states, we ran the model for 1-year random replicates in 1000 iterations of artificial urban layouts composed of a certain proportion of developed land, lot size pattern and turnover rate (Monte Carlo simulations).

Simulations were carried out on a raster grid with a spatial resolution of 12.5 m by 12.5 m. The modelling shell program sequence was initiated by reading in the allocation of lot boundaries and scenario settings (Fig. 2). Land use (developed or brownfield) was assigned randomly to the lots according to the desired proportion of developed sites. Within the framework of this study, we were unable to model and predict soil nutrient and water conditions at newly established sites or following abandonment. However, the soils in our case study area comprised of relatively homogeneous, sandy, artificially filled in material. Therefore, in the next step, soil parameters were randomly assigned to the brownfield sites using a set of values representative of the sampling plots which were derived from soil analyses (Schadek et al., 2009).

Time since abandonment ('site age') was drawn randomly for every open lot from an exponential distribution (see above) with a mean value corresponding to the turnover rate of the current scenario. Subsequently, plant species occurrence probabilities were estimated within an integrated grid-based GIS by applying the SDMs to every single grid cell. Using the plant predictions as additional predictors as described above, insect SDMs were subsequently applied. From the predicted species occurrence probability maps, species incidences were computed using a threshold value based on Cohen's kappa (Fielding and Bell, 1997).

Simulation results were evaluated from a conservation perspective by means of: (i) species richness as the proportion of plant and insect species occurring over the whole study area in relation to all modelled species; and (ii) species rarity as the proportion of rare plant and insect species occurring over the whole study area in relation to all modelled rare species. In both cases, we were interested in the conservation value of the whole study area, not of single brownfield lots. For species rarity, species were divided into regionally rare and common species. We used the plant atlas of (West-) Germany (Haeupler and Schönfelder, 1988) for the plant species where species distributions are given in raster cells of approximately 11 km $\times$ 11 km. To yield regional values, we restricted ourselves to the northwestern German region with Bremen as center. For grasshoppers, only an atlas for Bremen itself was

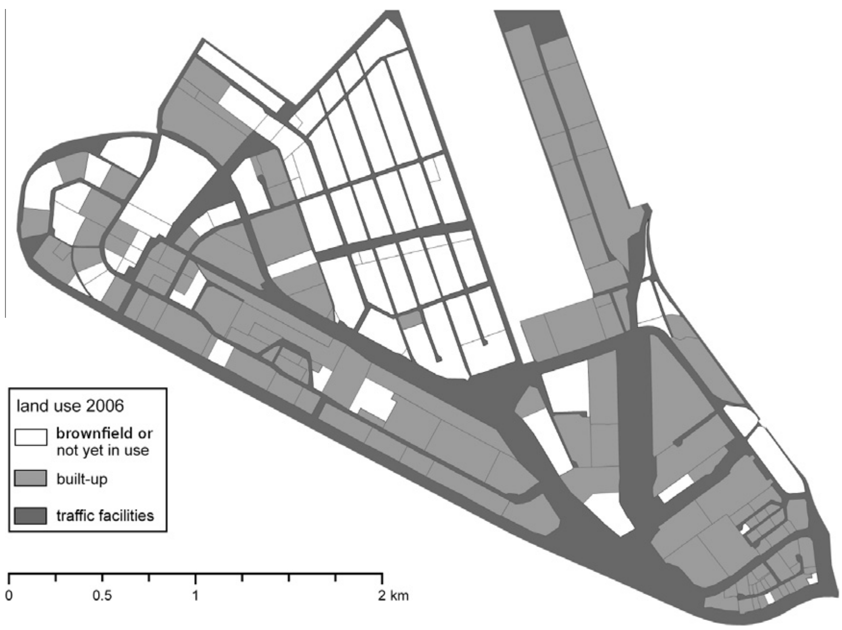

Fig. 1. Layout and current land use in 2006 of the study area.

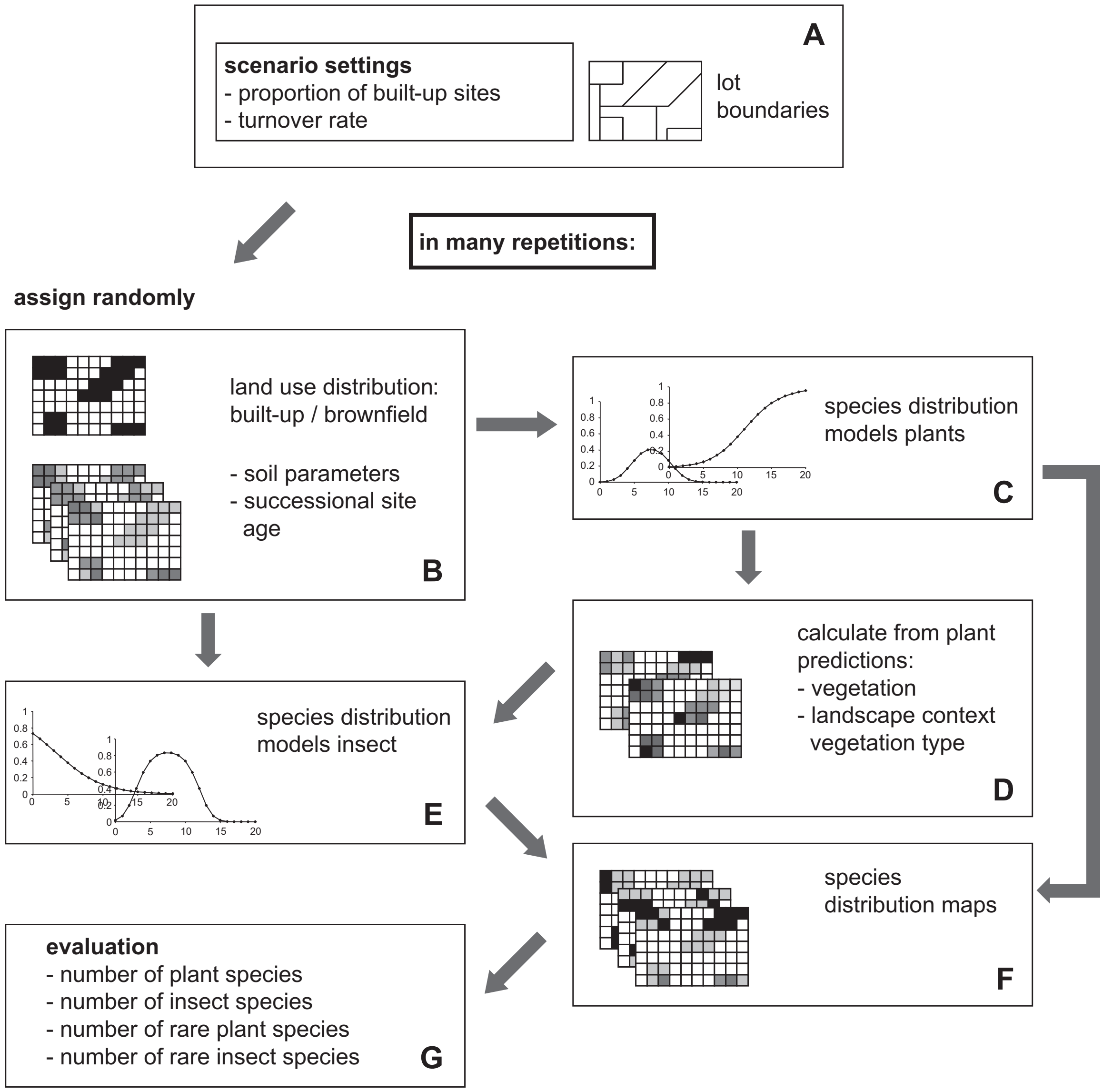

Fig. 2. Flow chart of the modelling shell. First (A), the allocation of lot boundaries and scenario settings are read in. In the next step (B), land use, successional age, and soil properties are assigned to the sites and this information is converted into raster maps. Then (C), plant occurrence probabilities are calculated by application of the SDMs to every raster cell. From these, vegetation parameters are estimated (D), and subsequently (E) the insect models are applied. The maps of species occurrence probabilities (F) are aggregated to evaluation parameters in the last step (G).

available (Hochkirch and Klugkist, 1998). In both cases, presence of a given species in 0–40% of all raster cells corresponded to the class 'rare'. All other species were classified as 'common'. For leafhoppers, rarity values were assigned based on expert knowledge provided by R. Biedermann. Mean and standard deviation of species richness and rarity over the replicate runs were calculated per scenario.

2.3. Scenario settings

According to our hypotheses, we assessed the effect of static land use (open sites remained open, developed sites remained developed and in use) vs. dynamic land use (a certain proportion of land was converted from developed to open and from open to developed every year). The static setting represented a situation where open land was set aside for conservation without any spatial dynamics. Consequently, we assigned the same site age to all open sites and tested the following age classes: 0, 3, 6, 10, 15, 20, 30, 40 and 50 years. Alternatively, the dynamic setting described situations, where the ratio of open land was the same as in the static setting, but its location shifted at different rates. To this end, different landscape turnover rates were represented by assigning exponentially distributed site age with different mean values to the brownfield lots. Mean successional age of brownfields was set to 3, 6, 10, 15, and 20 years, respectively, with the maximum age of a brownfield site restricted to 50 years, which was the oldest value found in the field. A lower mean age represented a faster turnover. The total area was kept constant in all scenarios.

To test species response to available habitat area we varied the proportion of developed lots between 40% and 90% in increments of 10%. We chose 40% as the minimum proportion because lower proportions of developed lots vs. open lots appeared unrealistic as the whole area should still qualify for an economically vital industrial site, rather than a wasteland.

To analyse the effect of lot size and spatial configuration on species occurrences, we subdivided the whole study area into lots according to four different layouts (Table 2). The first one comprised the original lot layout mapped on site ('original'); a second

Table 2
Lot sizes used in the different settings. 'Original' is the layout found in the field; in brackets the number of lots of each layout.

Lot size (ha)	Original (187)	Small (508)	Large (57)	Backup (225)
Mean	2.58	0.95	8.97	2.14
Min	0.11	0.11	4.68	0.13
Max	12.1	1.98	13.39	10.22

artificial layout was characterised by very large sites ('large'), as typically found in sites used by logistics enterprises; the third layout comprised many small lots ('small'); and the fourth was designed as a mosaic of a few large and many associated small sites ('backup'). In the latter scenario, the probability to be open was set inversely proportional to the lot size; hence it was more likely that such smaller lots were used occasionally and than abandoned again.

3. Results

3.1. Comparison of dynamic vs. static land use

Static land use had a negative effect on both plant and insect species richness (Fig. 3). As plant succession was allowed to proceed in the absence of disturbance, the number of species predicted to occur within the study area decreased for plants for site ages exceeding 3 years. For insects, species richness remained rather constant at approximately 33 predicted species to a site age of 20 years, then decreased rapidly. At a site age of 50 years for all open lots, only approximately half of all modelled plant and insect species were predicted within the study area in the static setting. In contrast, open land with shifting locations (dynamic setting) resulted in higher species richness regardless of the turnover rate.

3.2. Influence of turnover and proportion of open space

The proportion of open space and site turnover rate influenced species richness in the study area (Fig. 4): the more available open space, the higher the proportion of predicted species. Plant species richness peaked at a mean site age of 15 years, while insect richness peaked at 10–15 years. Variation over the 1000 simulation runs (expressed as a standard deviation; Fig. 4, bottom) increased with increasing density of developed sites. The lowest variance in plant species number was found at a mean site age of 15 years. For insects, the minimum variance was found at 15–20 years. In comparison to plant species, for all replicate runs in the analyses for all scenarios, an overall higher proportion of insect species was predicted with a lower variance in results. Similar to species richness, the proportion of rare species occurring in the study area was influenced by turnover rate and proportion of potential habitat (Fig. 5): the more open sites that remained available, the more rare species of both taxa were predicted to occur. Both taxa profited from a slow turnover. However, the proportion of rare plant species was much lower than that of insect species.

3.3. Influence of lot size

For the layout with large lot sizes, fewer species were predicted to occur over all scenario settings than for other lot size layouts ('original', 'small', and 'backup') (Fig. 6, left). Furthermore, variation over the 1000 simulation runs per setting was highest for the largest lot sizes (Fig. 6, right). Results for the other three layouts were similar. However, the differences were rather small (e.g. for a developed proportion of 60% and a mean site age of 10 years, approximately 41 insect species were predicted for large lot sizes and approximately 42.5 species for small lot sizes), but the values increased with a decreasing proportion of open sites.

4. Discussion

4.1. Factors influencing urban biodiversity

The results of our model showed that with respect to the two alternative hypotheses, i.e. (1) no turnover vs. (2) turnover from brownfield to developed sites and from developed to brownfield sites, the second hypothesis results in increased urban biodiversity. Thus, dynamic land use, based on the repeated turnover of brownfields to development and vice versa, maintained and even enhanced the conservation value of an industrial area in terms of species richness and rarity with respect to the urban species pool analysed here. This dynamic landscape facilitated different successional stages and consequently provided habitats for a range of different species in the study area (Flores et al., 1998), whereas a static landscape where habitats were not managed and characterised by spontaneous succession yielded late successional stages in every brownfield, thus excluding the species pool of earlier successional stages from the whole area (Kattwinkel et al., 2009). The relevance of patch dynamics and mosaic cycles for species occurrences in landscapes has been shown for many ecosystems, including forests and grasslands (Kleyer et al., 2007). The ecological

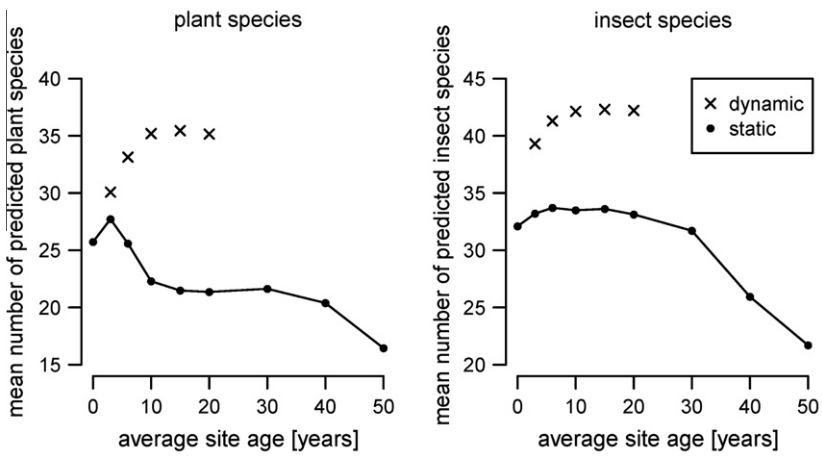

Fig. 3. Plant (right) and insect (left) species richness for dynamic land use (turnover) and static land use (same age for all sites). Lot boundary setting as found in the field ('original') and proportion of open sites is 0.4. Site age (*x*-axis) gives the age of all open lots for the static setting, but the average site age for the dynamic setting.

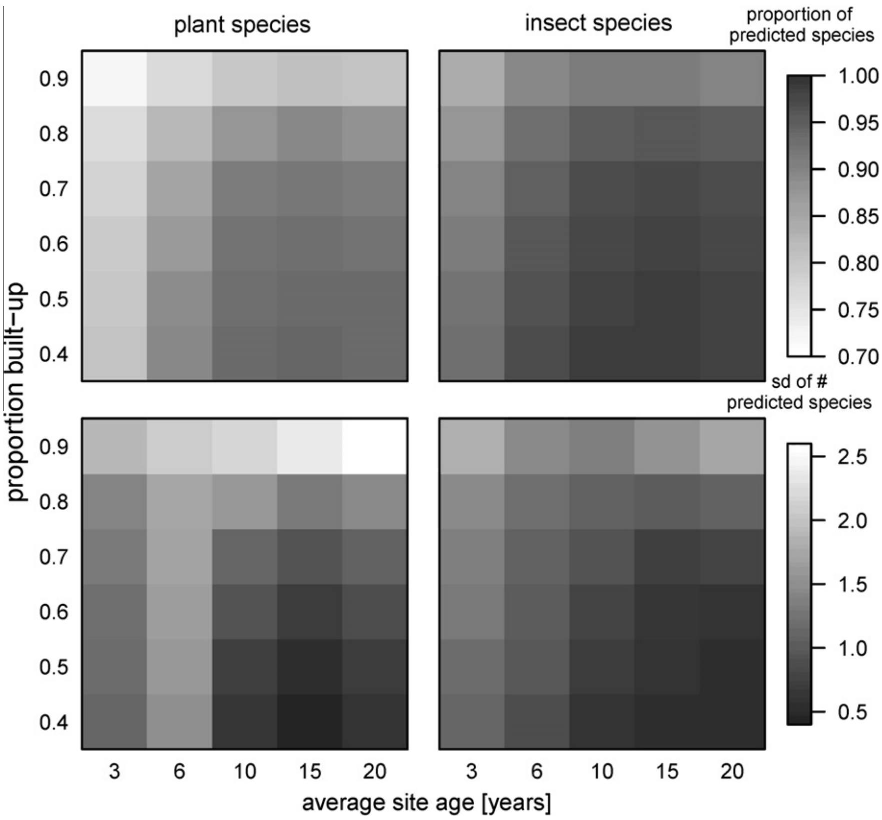

Fig. 4. Species richness (top) and its standard deviation (bottom) of plants (left) and insects (right) in relation to mean site age and proportion of built-up area. The top graph shows the mean number of predicted species in relation to the number of modelled ones (37 plant species and 43 insect species, respectively), the bottom one shows the standard deviation of the number of predicted species over the 500 simulation runs per scenario setting.

Fig. 5. Occurrence of rare plant (left) and insect (right) species in relation to mean site age and proportion of built-up area; the graphs show the mean number of predicted rare species in relation to the number of modelled ones (6 plant species and 10 insect species, respectively).

processes that govern dynamic landscapes can be viewed on temporal and spatial scales.

Therefore, two factors can be accessed by urban conservation planning that strongly affect the plant and insect community and consequently the nature conservation value of urban green spaces: (1) the proportion of open space; and (2) the rate of turnover from open to developed sites. Our results showed that the two factors complement each other because a higher proportion of developed sites can be partly balanced by a slower turnover. By extrapolating SDMs through space and time, our modelling approach provides recommendation of how to integrate biodiversity research into urban planning applications (Opdam et al., 2002; Wintle et al., 2005). When synthesising all scenarios, we recommend a 50–60% proportion of open space with an average site age of 15 years as the best combination of spatial and temporal habitat availability.

Regarding the spatial aspect, initially, a proportion of open space of 50–60% might seem high for industrial areas, as German law allows a site occupancy index of 0.8 (i.e. leaving only 20% open space; (BauNVO, 1990)). However, this regulation was not intended to account for the conservation of urban biodiversity when it was passed. Incorporation of conservation in urban planning requires a lower lot cover index in order to offer habitats for urban species. The new aspect we stress here is that conservation should be integrated into urban land uses rather than separated (Pedersen et al., 2004) and that it should be temporary rather than permanent.

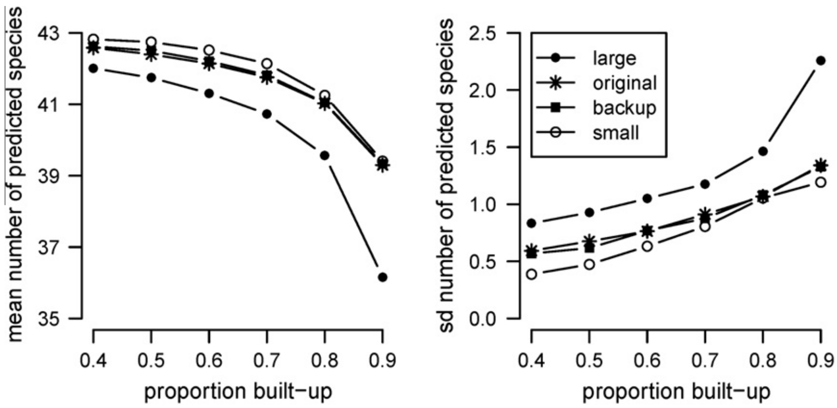

Fig. 6. Insect species richness (mean number of overall occurring species; left) and its variation (standard deviation; right) at different lot sizes as a function the proportion of built-up area. 'Original' refers to the lot layout as found in the study area; 'large' to fewer, but larger sites; 'small' to many small lots; and 'backup' to several large lots with associated small expansions sites. Mean site age set to 10 years.

By comparing different spatial layouts (lot sizes) for a given overall proportion of open space, layouts with many smaller sites ('small', 'backup', and 'original') were rated better than layouts with fewer, larger sites ('large'): Many smaller sites offered more variation in age classes at the same proportion of open space, thereby leading to a higher diversity of successional stages (see also Deutschewitz et al., 2003).

Regarding the temporal aspect, at first glance, the recommended turnover rate that results in an average age of 15 years for the brownfield sites may seem slow. However, it is well in line with the results of a land use distribution analysis of industrial and business sites of six German cities (including Bremen): The analysis of aerial photograph time series (1951–2003) revealed that the mean age of these open patches was 15 years averaged over the six cities and 10 years for Bremen (Empter, 2006). In both cases the distribution of site age classes of the open lots showed the pattern that result from random turn over of lots as assumed in this study (exponential distribution). On average, 40% of the area consisted of open land including brownfields, storage ground and unpaved traffic areas.

The applied modelling approach includes some simplifications: As we set soil properties to be constant within one lot in the model, we did not account for the larger heterogeneity within larger patches (Ouin et al., 2006). Therefore settings comprising a higher number of small patches might have resulted in higher conservation values. Additionally, we did not consider dynamic population processes, which could result in decreased risks of extinction for populations inhabiting larger patches and higher risks of species failing to colonise smaller, isolated patches (Hanski and Thomas, 1994). Nevertheless, by including landscape context variables that describe the landscape at different spatial scales as predictors for species occurrence (Dauber et al., 2005), the SDMs implicitly account for connectivity and patch area. However, only the landscape context of the current year was available in the statistical analyses. It reflects an indirect relationship to species occurrence opposed to the landscape configuration of previous years, which might reflect a direct causal relationship. Still, the statistical analyses indicated that the current landscape context was an important driver for species occurrence (Electronic appendix, Table A1), which was also shown in other studies investigating riparian birds (Martin et al., 2006), butterflies (Cozzi et al., 2008), epiphytes and ground-living plants (Paltto et al., 2006), and plants on abandoned railway areas (Westermann et al., 2011). Additionally, if landscape turnover is not too high (mean site age larger than 3 years) the landscape context does not change extremely from year to year.

Our results suggest that the permanent protection of single sites from land use as a traditional concept of nature conservation is not feasible for ruderal communities in unused brownfields and wastelands within urban areas. In general, stationary, isolated protected reserve designs have been questioned to be protective in landscapes that are characterised by anthropogenic disturbances (Bengtsson et al., 2003; von Haaren and Reich, 2006), and new concepts of dynamic nature conservation are currently being proposed and tested (Drechsler et al., 2009; Pressey et al., 2007; Rayfield et al., 2008). Additionally, as species richness was driven both by local factors (e.g. soil properties and successional age of a patch) and by landscape factors (landscape context variables), planning for urban conservation has to focus on the landscape scale, rather than on the scale of a single patch. This fact has been recognised (Mörtberg et al., 2007) but is nevertheless often neglected.

4.2. Integrating temporary conservation into urban planning

Dynamic landscapes and disturbance ecology have been studied in the context of conservation for a long time (e.g. Fahrig, 1992; Snäll et al., 2005). In this context, the intermediate disturbance hypothesis, stating that species diversity will be highest at intermediate frequencies of disturbance, has been successfully applied to urban environments (Zerbe et al., 2003). The dynamic nature of open space in urban areas matches the concept of temporary conservation, which builds on the fact that successional stages providing habitat for certain species only exist for relatively short periods of time. Planning and managing such dynamics to optimise conservations value could certainly be accomplished by management schemes including mowing or sod-cutting. However, we suggest integrating economic use and biodiversity management in order to open new opportunities in urban conservation planning.

For instance, German law requires ecological compensation during planning and construction processes. Similar regulations are in place for other European countries, although often less stringent and/or extensive (Peter et al., 2002). Compensation areas within industrial areas under construction would supersede the need for adequate compensation at other locations. The size of such green industrial parks will be larger, but their ecological value will be substantially higher than without the in situ compensation and can even be enhanced compared to former land use practices (e.g. agricultural). Thus, future developments can be viewed as business parks of lower density, which allows an increase in ecological value but also adds to leisure and recreational opportunities. Such green, wild spaces with open public access can support human well-being while simultaneously increasing the economic value of urban areas (Hobden et al., 2004; McGranahan et al., 2005). Additionally, they provide important ecosystem services such as micro-climate regulation, air filtering, and water regulation.

Furthermore, widespread changes will be observed in commercial construction, shifting the focus to short-term, largely due to an unpredictable economy and its influence on architecture (Hassler and Kohler, 2004). Nevertheless, a study by Dissmann and Hopp (2002) showed that 80% of industrial construction facilities remain on-site longer than 20 years. In the context of temporary conservation, some interspersed short-term temporary buildings will allow for rapid enough turnover rates to create adequate newly opened sites. These buildings should be of high architectural and building quality to be more than provisional solutions, i.e. they should be reusable and dismountable, as well as ecologically and economically efficient (see examples in Draeger, 2010). The intentional destruction of some habitat (combined with the creation of new open space at other locations) can reduce the reservations of certain stakeholders against nature conservation. If the public perception of brownfield sites is improved by demonstrating their ecological value instead of continuing to view these areas as wasteland creating social problems (Herbst and Herbst, 2006), even during periods of low economic development, brownfields will convey a positive effect on the overall appraisal of business parks.

Urban development contracts between city administration and property holders can be a means of regulating temporary open spaces. The city of Leipzig, for example, offers legal advice to arrange agreements between temporary users of abandoned sites and owners, which regulate the duration and kind of use while preserving the development rights of the owner and even exclude the lot from real estate tax (Stadt Leipzig, 2005). This way, negative effects of city shrinking (including population value decline and house and property abandonment) can be attenuated while the positive implications are enhanced (Haase, 2008). This innovative approach, developed to moderate the consequences of economic decline could be expanded to include the concept of temporary conservation.

5. Conclusions

Urban biodiversity depends on a variety of different habitats. Thus, if biodiversity is to be maintained within urban areas, temporary conservation offers an opportunity to facilitate both conservation management and an urban renaissance. Preserving a proportion of land for conservation in a static setting results in much lower species richness and rarity than in a dynamic setting. Therefore, a certain proportion of developed land is essential to cause habitat turnover due to redevelopment at one lot and abandonment at another, which is necessary to support species persistence and viability. Our results suggest that 50–60% of the area should be left open for in average 15 years to support urban biodiversity. Such temporary conservation overcomes the traditional concept of protecting isolated habitats, and redefines urban green spaces in a dynamic and flexible context. Moreover, temporary buildings represent a contemporary innovation in keeping with the short-term, fast-moving markets, and a means to track new economic trends. This combination of spontaneous, open green spaces and modern architecture can increase the ecological as well as economical value of business areas.

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Supplementary material

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.biocon.2011.06.012.

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