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Research article

Temporal revegetation of a demolition site—a contribution to urban restoration?

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Abstract

In urban areas, open space including brownfields often became rare due to increasing urbanisation. Urban brownfields can be important for biodiversity, but especially brownfields in early successional stages seem to be refused by urban residents due to their sparse vegetation and less aesthetic appearance. The aim of this study was to revegetate a young demolition site in the city core of Osnabrück, Germany and thereby to support native plant diversity and aesthetic values. We developed two seed mixtures of native plant species and tested them in a large-scale field experiment over two growing seasons. Both seed mixtures developed towards structurally diverse and flower-rich vegetation. Establishment rates of sown species were consistently larger than 75%. Revegetation of the predominantly bare anthropogenically transformed soil by introduced species occurred fast. Vascular plant cover and vegetation height were higher on sown plots than in controls, but did not differ between the seed mixtures. Seeding did not increase plant species richness and did not reduce the establishment of a potentially invasive non-native plant species. The cover of Red-List species from the spontaneous vegetation was significantly higher in control plots. Our results indicate that not all aims can be reached on one restoration site. It has to be discussed if it is better to invest a restoration budget for measures aiming to increase acceptance of endangered pioneer plant species from the spontaneous vegetation or to introduce more attractive and more competitive species of later successional stages.

Key words: urban restoration, species introduction, urban green infrastructure, novel ecosystems, regional provenance

Introduction

In a global context, urbanisation is an increasing phenomenon. Urban residents now exceed 50% of the global population, and by 2050, there will be 2.5 billion more town and city dwellers (=79%) on the planet (United Nations 2014). In response to this global urbanisation trend, two major and opposite strategies of urban planning have emerged to meet the challenge of creating new, urban residential and commercial spaces. On one hand, many cities continue to grow and are sprawling towards surrounding areas. This phenomenon leads to massive greenfield consumption in adjacent suburban and rural areas often resulting in degradation of natural and cultural ecosystems and reducing their ecosystem services (e.g. Salvati, Ferrara, and Ranalli 2014). On the other hand, planning strategies of densification have gained popularity in many growing cities. This infill development aims to reuse existing inner-urban areas to counter urban sprawl (McConnell and Wiley 2011).

Open space in urban areas for infill development includes vacant lots, demolitions sites, wastelands or brownfields (hereafter summarized as brownfields), and both private and public green space like lawns, gardens or park areas. These different types of open space can be summarized as urban green infrastructure (Mathey et al. 2015). Elements of this urban green infrastructure are under threat in growing cities due to the actual densification policy (Haaland and van den Bosch 2015) as has been shown for urban brownfields in the UK (Dallimer et al. 2011; Lewis 2005). In shrinking cities, however, brownfields have also been transformed into informal greenspace recently (Anderson and Minor 2017; Kausch and Felinks 2012).

Open space occurring after demolition of buildings or infrastructure is often perceived as 'waste'-land with low values for recreation (Bixler and Floyd 1997; Hofmann et al. 2012; Lafortezza et al. 2008). Especially brownfields in earlier successional stages with sparse and flower-poor vegetation and large areas of bare ground seem to be rejected (Brun, Di Pietro, and Bonthoux 2018; Mathey et al. 2018; Rink 2005). On the other hand, urban residents are often inventive to improve such open space in their neighbourhood for their activities, and therefore for their well-being (e.g. urban gardening initiatives, Németh and Langhorst 2014; Rall and Haase 2011). Additionally, it has been emphasized that brownfields in cities with dense building structure and less open space can offer children the opportunity to play outdoors and interact with nature (Keil 2005; Rupprecht et al. 2015, 2016). Management approaches to enhance aesthetic values of brownfields generally include the introduction of non-native ornamental plants (Köppler et al. 2014; Kühn 2006). However, to support local and regional native plant biodiversity, it would be more suitable to use site-specific regional native plant material (Kiehl 2010; Vander Mijnsbrugge, Bischoff, and Smith 2010). Management approaches for urban brownfields have to consider potentials and risks for different species groups relevant for nature conservation purposes in order to promote local and regional native biodiversity (reviewed by Bonthoux et al. 2014; Stewart et al. 2017). On the other hand, urban areas are known as hotspots of non-native plant species (Europe: Pyšek, 1998; North-America: Clemants and Moore 2003). These seem to occur predominantly in different kinds of urban ecosystems of earlier succession stages (Kowarik 1995; Pyšek et al. 2004) and subsequently contribute to the homogenisation of European urban flora (Ricotta et al. 2014). Furthermore, urban areas can serve as a source for the spread of non-native species to suburban and rural areas, where they can become invasive and problematic for local native biodiversity (Duguay, Eigenbrod, and Fahrig 2007; Moffatt and McLachlan 2004; Padayachee et al. 2017).

Based on techniques for ecological restoration of grasslands by introduction of native plant species (reviewed in Kiehl et al. 2010), initial attempts have been made to use the potential of brownfields to promote native grassland species, which have become rare in rural areas (Kausch and Felinks 2012; Fischer et al. 2013). However, these approaches focused on long-term open space after building demolition or brownfields of mid-successional stages. Until now, projects and concepts for young and only temporarily available demolition sites in growing cities are rare (but see Kattwinkel, Biedermann, and Kleyer 2011).

With a focus on temporarily available open space intended to be rebuilt after a few years, we aim to develop and test restoration measures, which can enhance regional native plant diversity and minimise the spreading of non-native plant species. By introducing native plant species a rapid revegetation was intended taking into account the temporal availability of the sites. In addition, the restoration measure should improve the acceptance and recreational value of these areas by enhancing their aesthetic appearance. Thereby, we assume that especially people in the urban context appreciate flower rich (i.e. large-flowered species) vegetation types (Lindemann-Matthies and Bose 2007; Southon et al. 2017).

Specifically we asked the following questions:

1. Can the introduction of native plant species of regional provenance induce a more rapid revegetation of bare anthropogenic soils compared to natural succession?
2. Can the introduction of native plant species increase native plant species richness and suppress the establishment of (invasive) non-native species?
3. Does spontaneous vegetation include species with nature conservation value, and if so, are these species negatively affected by species introduction?
4. Do assumed heterogeneous soil conditions affect the establishment of introduced native plants and plant species from the soil seed bank?

Methods

Study site

The study area is a former military barrack located within the city centre of Osnabrück in Northwest-Germany (52°17′N, 8°01′E, 67 m a.s.l.). As part of the conversion process from former British military barracks to civilian use, buildings of the barracks were removed in 2013. Demolition waste was largely removed from the area and at the end of 2013 the soil was flattened with natural soil material (sand and loam) from the surrounding area, partially mixed with rubble. The natural soil type in the area had been a gleyic podzol. The area covers about 10 000 m^2 and was planned to be rebuilt within 3–6 years after demolition.

Treatments and experimental setup

Two native plant seed mixtures were developed for greening the area (Supplementary Table S1). The 'high-diversity' mixture (HD) contained 25 native species and the 'low-diversity' mixture (LD) 13 species (species of both seed mixtures referred to as 'target species' in the following). Seed mixtures exclusively contained local native forb species and a few archaeophytes (pre-1500 introductions), but no grasses. Selected species should be able to create colourful flowering aspects over the whole growing season (phenology based on Jäger 2011). Ellenberg indicator values (Ellenberg et al. 2001) were used for further species selection (preferably species with nitrogen value ≤ 6, moisture value ≤ 5 and reaction value ≥ 5, Supplementary Table S1) as soil pre-investigations had shown slightly acidic to alkaline pH and dry environmental conditions in summer. To support regional native biodiversity, we selected plant species, which occur in regional plant communities of flower-rich mesophytic grasslands and in early and mid-successional stages of ruderal vegetation of dry habitats (according to the regional flora of Weber 1995). In both seed mixtures, approximately 6% of legume seeds were added to the seed mixtures (HD: three species; LD: two species). To get a rapid and colourful greening at the beginning of the measure, we added a comparably high amount of seeds of predominantly large-flowering annual species occurring in regional arable vegetation (both mixtures with six species: *Cyanus segetum*, *Glebionis segetum*, *Papaver rhoeas*, *Papaver dubium*, *Matricaria chamomilla* and *Myosotis arvensis*). Due to the temporary lifetime of the brownfield, we did not sow Red-List species (exception: *G. segetum*) and chose cost-effective species. Seeds were obtained from commercial native seed production (Rieger-Hofmann GmbH, Germany). The study area belongs to seed provenance zone 1 (Prasse, Kunzmann, and Schröder 2010) and therefore, seed lots were mainly derived from seed provenance zone 1 or the adjacent zone 2.

For the experiment we established nine blocks of about 500 m^2 each (10 m length × 50 m width). Within each block seed mixtures were sown in strips of 20 m width also on the areas between the blocks. Each of the sown strips was split in two 10 m strips, one with and one without yearly mowing. Control (C) stripes without sowing and mowing had a width of 10 m. Each block contained five treatments: HD with mowing, HD without mowing, LD with mowing, LD without mowing and Control (nine replicates per treatment). Before seed-bed preparation, the study site was vegetated only very sparsely with few plant species from the spontaneous vegetation that had established between autumn 2013 and spring 2014. After harrowing the whole study site to create a suitable seed bed, sowing was done by hand in the LD and HD strips in April 2014. Seeds were not stratified before sowing. Both seed mixtures were sown with a similar low sowing density of c. 1300 seeds/m^2 (HD = 0.8 g/m^2, LD = 0.65 g/m^2) to allow also the establishment of species with high conservation value from the soil seed bank. After sowing, the study site was milled to alleviate the contact of seeds with the soil surface. Mowing treatments were conducted in both 2014 and 2015 in September. The sparse cuttings were not removed from the site.

Sampling

For vegetation and soil sampling, 45 permanent plots of 2 m × 2 m were installed on the study area, one for each treatment in each block. The permanent plots were located each in the middle of the different treatment strips to reduce dispersal from adjacent treatments into the plots (distance to adjacent treatments was at least 4 m).

To evaluate the relationship between vegetation and soil parameters, soil samples were taken from each permanent plot (mixed samples of eight sub-samples of 0–15 cm depth) in April 2014. Air-dried soil samples were sieved (2 mm mesh size) and soil pH was measured in $CaCl_2$ solution. Soil contents of calcium acetate lactate (CAL) exchangeable phosphorus (P) and potassium (K) were analysed according to VDLUFA (2002) via molecular spectroscopy (PerkinElmer Lambda 25) for phosphorus (P) and flame photometry (Eppendorf ELEX 6361) for potassium (K). After grinding, the percentage of total nitrogen (N), total carbon (C) and inorganic C (by adding phosphorous acid before analysis) was measured using an elemental auto-analyser (Carlo-Erba NA 1500). Soil organic carbon (C_{org}) was calculated by total C content minus inorganic C.

Vegetation relevés were conducted in July 2014 and August 2015. At each permanent plot, total vegetation cover, vegetation height, the cover of bare soil and litter as well as the cover of all vascular plant species in percent cover (smallest unit 0.1%) were recorded. Nomenclature of species is based on Jäger (2011).

Statistical analysis

All statistical analyses were performed using R 3.1.0 (R Core Team 2014). Analyses were conducted for each growing season separately. As no effect of mowing could be found (data not shown) data of mown and unmown plots of HD and LD, respectively, within a block were pooled for further analyses. To evaluate effects of the two sowing treatments on (invasive) non-native species, we analysed the number and cover of neophytes (post-1500 introductions, status based on Garve 2004) and the cover of *Senecio inaequidens* as this species is classified as potentially invasive species based on the German neobiota assessment (Nehring et al. 2013). To study the effects of seed mixtures on spontaneously established native plant species of high nature conservation value, we analysed the number and cover of species listed in the regionalised Red-List of vascular plant species, Lower Saxony, Germany (region: 'Hügel- und Bergland', Garve 2004). For both *S. inaequidens* and the Red-List species, we calculated their frequency of occurrence in plots and tested for differences between treatments by performing χ^2-tests.

To identify differences in short-term establishment success of target species, vegetation structure parameters, species richness, establishment of plant species groups (i.e. forbs, legumes, grasses and woody species), non-native species and Red-List species between sowing treatments (and control), we constructed generalized linear mixed models (GLMMs) with block as random factor. In each case we used treatment and soil organic carbon (C_{org}) as fixed factor (as soil parameters showed multiple intercorrelations and to prevent constructing overfitted models, we chose C_{org} as most meaningful soil parameter indicating differences between raw soils and more humus-rich soils with higher nutrient availability best). GLMMs simplification was conducted using the step function in R. In case of a non-significant random factor we used linear models (LM) without considering block as factor. We evaluated the fit of the final model for unbiased and homoscedastic of residuals visually by plotting fitted values versus residuals. Some count data needed to be square root-transformed before analysis. The goodness of GLMMs fit was evaluated by calculating conditional R^2 (Nakagawa and Schielzeth 2013). We performed ANOVAs on final models and subsequent Tukey post hoc tests in case of significant treatment effects using the multcomp package in R.

Results

Vegetation structure, establishment success of target species and species groups

Already in the first season we observed a fast vegetation establishment. There were no differences between the two sowing treatments and the control in vascular plant cover, bare ground cover and vegetation height 3 months after sowing (Table 1). During the second growing season (2015), vegetation height and the cover of vascular plants also did not differ between the two sowing treatments but were significantly higher than on control plots. Accordingly, the cover of bare ground was significantly higher on control plots (Table 1).

The total establishment rate of the sown target species reached 84% (mean: 57.1 ± 11.0%) and 92.3% (mean: 77.8 ± 10.8%) of sown species for the HD- and LD-mixture, respectively, already in the first season but declined to 76.0% for the HD and 84.6% for the LD-mixture in the second season. In the summer of the first season large-flowered annuals caused the predominant flowering aspect (Fig. 1A and B). Completely absent target species were *Heracleum sphondylium* (component only in the HD-mixture) and *M. arvensis*. The species *Hypericum perforatum*, *Tanacetum vulgare* and *Verbascum densiflorum* (only HD-mixture) showed very low establishment success growing only scattered outside the permanent plots (Supplementary Table S1). During the second season we observed a sequence of different opulent flowering aspects of target species over the season (Fig. 1B–D), while control areas appeared to be less structured and colourful (Fig. 2).

During both growing seasons 11 species of the seed mixtures also occurred in the control plots, but only three of them reached more than 50% frequency here (*Achillea millefolium*,

Table 1: Differences of several vegetation parameters (means ± 1 SD) between sowing treatments and control in the first and second seasons after sowing

	First season				Second season			
	Control	Low diversity	High diversity		Control	Low diversity	High diversity	
Parameters	$n = 9$	9	9		9	9	9	
Vegetation height (cm)	14.0 ± 7.0	14.0 ± 4.0	12.0 ± 5.0	n.s.	14.0 ± 5.0^{a}	32.0 ± 17.0^{b}	32.0 ± 11.0^{b}	***
Number (n) of								
Vascular plants	31.8 ± 10.2	35.4 ± 6.7	36.3 ± 4.1	n.s.	28.9 ± 6.0	27.5 ± 3.1	29.9 ± 3.7	n.s.
Target species	2.3 ± 1.8^{a}	10.1 ± 0.7^{b}	14.3 ± 2.6^{c}	***	2.7 ± 1.2^{a}	7.5 ± 0.9^{b}	11.2 ± 2.2^{c}	***
Forbs	22.2 ± 7.0	25.1 ± 4.2	26.2 ± 2.7	n.s.	17.6 ± 4.8	16.6 ± 2.1	18.8 ± 3.5	n.s.
Legumes	1.7 ± 0.9^{a}	3.3 ± 0.5^{b}	3.7 ± 0.9^{b}	***	2.9 ± 1.4^{a}	4.1 ± 0.6^{ab}	4.4 ± 0.9^{b}	*
Grasses	7.9 ± 2.9	7 ± 2.3	6.4 ± 1.6	n.s.	8.4 ± 2.2	6.9 ± 1.7	6.7 ± 1.1	n.s.
Woody species	0.8 ± 0.4	0.9 ± 0.9	0.7 ± 0.7	n.s.	1.7 ± 1.1^{a}	1.3 ± 0.8	0.8 ± 0.6	n.s.
Non-native species	3.4 ± 1.7	3.2 ± 1.2	2.8 ± 1.0	n.s.	1.9 ± 0.8	1.7 ± 1.0	1.6 ± 1.0	n.s.
Red List-species	1.4 ± 0.5	0.9 ± 0.9	1.1 ± 0.7	n.s.	2 ± 1.1	1.4 ± 0.7	1.1 ± 0.9	n.s.
Cover (%) of								
Bare ground	83.1 ± 13.5	87.2 ± 7.1	85.8 ± 14.2	n.s.	56.1 ± 29.8^{a}	42.0 ± 23.3^{b}	40.3 ± 26.1^{b}	***
Vascular plants	16.9 ± 13.5	12.8 ± 7.1	14.2 ± 14.2	n.s.	43.0 ± 28.9^{a}	56.7 ± 22.3^{b}	58.8 ± 25.3^{b}	***
Target species	1.2 ± 1.1^{a}	4.3 ± 2.0^{b}	5.7 ± 3.3^{b}	***	2.6 ± 1.9^{a}	39.2 ± 19.5^{b}	45.9 ± 21.2^{b}	**
Forbs	8.1 ± 5.9	6.8 ± 3.5	6.6 ± 6.1	n.s.	2.4 ± 5.4	8.4 ± 9.7	4.8 ± 6.3	n.s.
Legumes	2.4 ± 2.9	2.7 ± 1.6	3.8 ± 2.6	n.s.	33.4 ± 27.5^{a}	43.8 ± 21.1^{b}	48.7 ± 22.6^{b}	**
Grasses	6.3 ± 5.0	3.2 ± 2.5	3.9 ± 6.2	n.s.	7.2 ± 6.4	4.6 ± 4.8	5.3 ± 8.1	n.s.
Woody species	0.1 ± 0.1	0.1 ± 0.0	0.0 ± 0.0	n.s.	0.4 ± 0.7	0.2 ± 0.1	0.1 ± 0.1	n.s.
Non-native species	1.2 ± 1.2	0.6 ± 0.4	0.4 ± 0.2	n.s.	1.2 ± 1.7	0.4 ± 0.4	0.3 ± 0.2	n.s.
S. inaequidens	0.6 ± 0.7	0.3 ± 0.3	0.2 ± 0.2	n.s.	0.3 ± 0.5	0.1 ± 0.1	0.1 ± 0.1	n.s.
Red-List species	0.5 ± 0.7	0.3 ± 0.4	0.3 ± 0.3	n.s.	1.2 ± 1.3^{a}	0.3 ± 0.3^{b}	0.3 ± 0.3^{b}	*

Treatments: 'High diversity' (25 species), 'Low diversity' (13 species) and control (non-sowing). For each growing season, the differences between the treatments were tested with ANOVA and subsequent Tukey post hoc tests. Significance level: n.s.: non-significant, $^{*}P < 0.05$, $^{**}P < 0.01$, $^{***}P < 0.001$. Different letters indicate significant differences at $P < 0.05$.

Medicago lupulina and *M. chamomilla*). Treatments differed in relative target species richness in both seasons with lowest values for the control. With 39.9 ± 9.63% in the first and 37.3 ± 6.0% in the second season, the proportion of target species in relation to total number of vascular plants was about 10% higher in the HD-mixture plots than in the LD-mixture plots (Fig. 3). In both seasons, there were no statistical differences in mean number of vascular plant species among the treatments (overall mean 34.5 ± 7.19 SD, Table 1), but species richness significantly decreased from the first to the second season ($F_{(1,\ 52)} = 12.42$, $P < 0.001$, LM). Total species richness in plots was 86 (C), 90 (HD) and 90 (LD) in the first season and 82 (C), 86 (HD) and 91 (LD) in the second season.

Legume cover was similar on sown plots and controls in the first growing season. In the second season, it was about 10% larger on control plots than on sown plots but did not differ between the two sowing treatments (Table 1). Nevertheless, legumes were the dominant species group in both control and sown plots, especially during the second season, when relative legume cover made up more than 50% of total vegetation cover (Table 1). *Trifolium pratense*, *Trifolium repens* and *M. lupulina* were the dominant species in the sowing treatments (cover: HD: 76.5 ± 23.5%, LD: 74.3 ± 19.2%) and *T. repens* and *M. lupulina* in control plots (cover: 64.1 ± 29.47%). We did not detect any difference in number and cover of forb species, grass species, and woody species between the treatments in both seasons (Table 1).

Effect of seeding on non-native, invasive non-native and Red-List species

There was no difference in number and cover of non-natives and cover of the potentially invasive non-native *S. inaequidens* between the two sowing treatments and between sown plots and control in both seasons (Tables 2 and 3). The frequency of *S. inaequidens* in plots also did not differ between treatments in both seasons (first season: $\chi^2 = 0.6087$, df = 2, $P = 0.7376$, second season: $\chi^2 = 0.0909$, df = 2, $P = 0.9556$).

In total, we detected eight spontaneously established Red-List species in both seasons. The most frequent species were *Leontodon saxatilis* and *Aira caryophyllea*. *Aphanes australis*, *Scleranthus annuus*, *Scleranthus polycarpos*, *Setaria pumila*, *Festuca filiformis* and *Ornithopus perpusillus* occurred less frequently. Treatments had no effect on the number of Red-List species in both seasons (Tables 1 and 2). The cover of Red-List species also did not differ between the treatments in the first season, but was significantly higher in the control plots (1.2 ± 1.3%) than in the sowing treatments (0.3 ± 0.3%) in the second season. In contrast, there was no difference between the sowing treatments (Table 1). The frequency of Red-List species in plots did not differ between the treatments in both seasons (first season: $\chi^2 = 5.6822$, df = 10, $P = 0.8412$, second season: $\chi^2 = 6.9441$, df = 10, $P = 0.7307$).

Effects of soil conditions on the establishment of species groups

Soil analyses indicated very nutrient poor and slightly acidic to calcareous conditions (Table 3). Soil organic carbon (C_{org}) ranged from 0.12 to 0.53%. Regarding the large standard deviations compared with the mean of each soil parameter (Table 3), the analysis indicated strongly heterogeneous soil conditions for the study site. Soil skeleton (gravel >2 mm) contained a mix of broken natural rocks, brick stone particles and mortar fragments from the demolished buildings. Total skeleton content was positively correlated with pH. Only the parameters pH, CAL

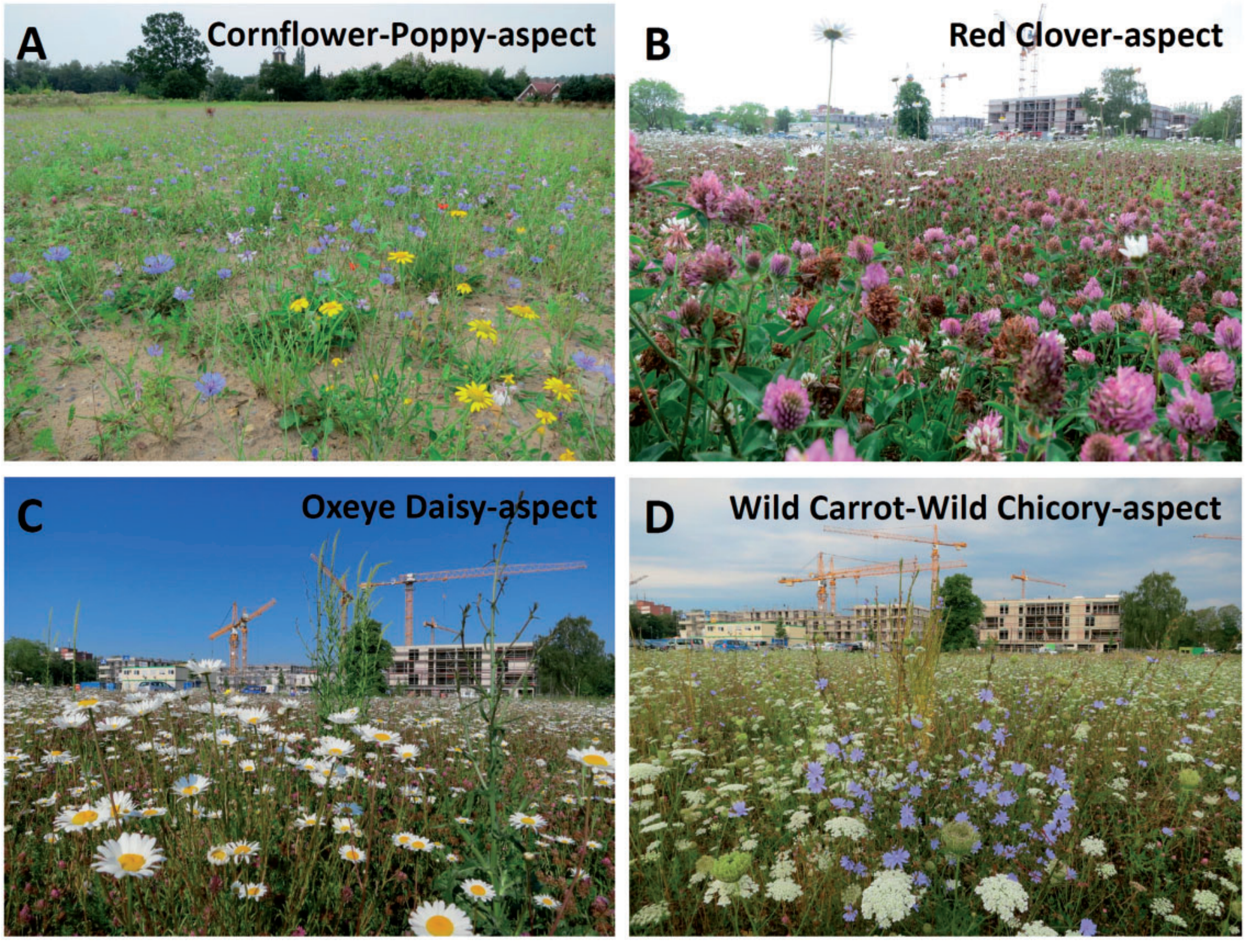

Figure 1: Changing flowering aspects in areas revegetated with the HD-mixture (25 plant species) during the two growing seasons of the study. (**A**) Aspect with dominating *C. segetum*, *G. segetum*, *P. rhoeas* and *P. dubium*, July 2014. (**B**) Aspect with dominating *T. pratense*, May/June 2015. (**C**) Aspect with dominating *Leucanthemum ircutianum*, July 2015. (**D**) Aspect with dominating *Daucus carota* and *Chichorium intybus*, August 2015.

exchangeable P and total N, respectively, were not intercorrelated (Table 3 and Supplementary Table S2).

Regression analyses showed that the cover of bare ground was negatively and the cover of vascular plants positively correlated with C_{org} (Table 2). While the number of forbs increased with increasing soil C_{org} content in the first season, a negative relationship with the cover of forbs was found in the second season. In both seasons, the establishment of legumes and grasses was positively linked to soil C_{org} content. Further positive relationships with C_{org} could be detected for the cover of Red-List species in the first season and the cover of non-native species in the second season (Table 2).

Discussion

Vegetation structure, establishment success of target species and species groups

Both sowing treatments were successful concerning the rapid revegetation of bare soil. Especially legumes (Fabaceae) seem to play an important role in the revegetation of nutrient poor urban soils. As legumes with their symbiotic *Rhizobium* bacteria are able to fix nitrogen from the atmosphere, they have a competitive advantage and are known to dominate early successional plant communities (Baasch, Kirmer, and Tischew 2012; Zaplata et al. 2013). In our sowing treatments nitrogen fixation was performed predominantly by the introduced *T. pratense*, whereas legume species from the spontaneous vegetation (*T. repens* and *M. lupulina*) adopted this function in control plots. This means that a rapid greening of urban demolition sites by natural succession is possible if appropriate species (in our study, i.e. legume species) are still present in the soil seed bank or can colonize the focus area by natural dispersal from surrounding vegetation.

Interestingly, species introduction by sowing native plant species did not increase mean species richness neither by introducing 25 species (HD) nor 13 species (LD). This result shows that even a young urban demolition site can exhibit a large soil seed bank potential. Seed dispersal from sowing areas into control plot can be precluded due to the fact that target species also found in control plots predominantly did not develop seeds in the first growing season. The three most common target species also found in control plots most likely established from the soil seed bank as these species are typical ones on younger brownfields and represent historical land use types like semi-natural (dry) grassland and agricultural fields in this area (see below). Furthermore, species introduction in early stages of secondary succession patterns does not result inevitably in a higher species richness considering a time frame of two growing seasons (e.g. Baasch, Kirmer, and Tischew 2012; Edwards et al. 2007; Stevenson, Bullock, and Ward 1995). The decline in species richness from the first to the second season can be explained by the disappearance of annual species due to the lack of soil

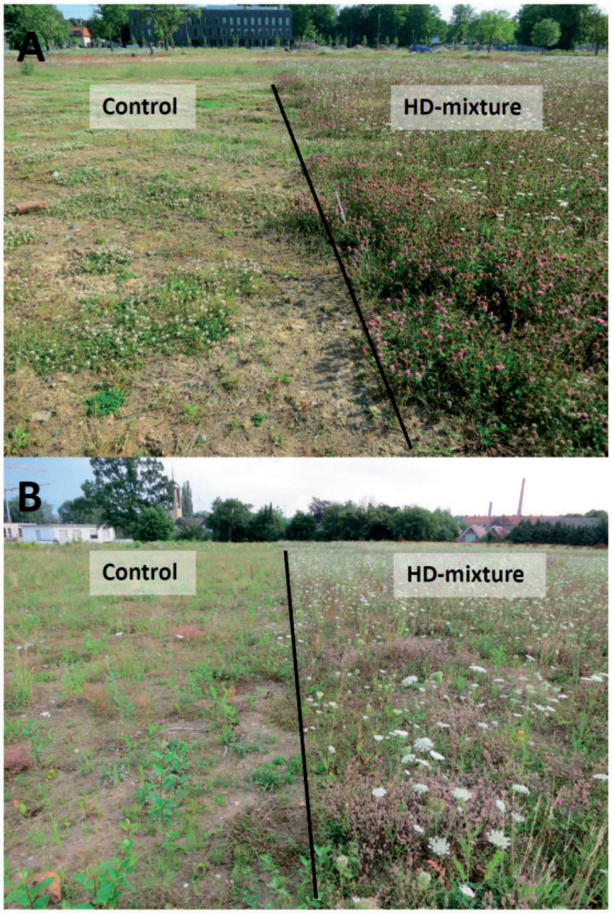

Figure 2: Flowering aspects in June (**A**) and August 2015 (**B**) of the non-sown control area in comparison to the adjacent area sown with the HD-mixture.

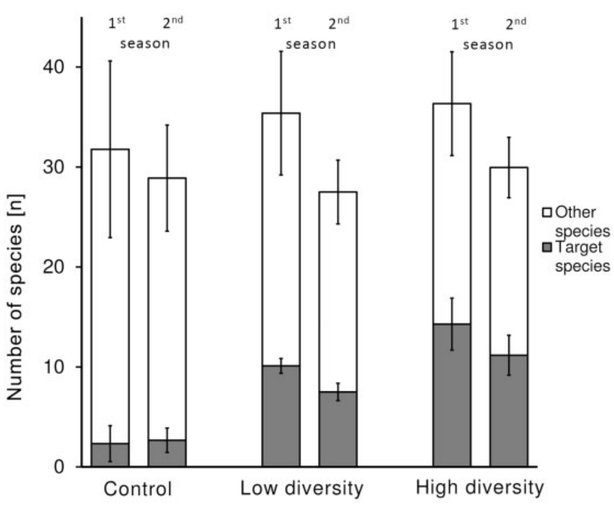

Figure 3: Mean number (± 1 SD) of sown target species and other species for the different treatments in 2014 and 2015 (for statistics see Table 1).

disturbance, both for annuals introduced by seed mixtures and from the spontaneous vegetation in control plots, as it has also been shown in other studies (Kiehl et al. 2014; Kirmer and Tischew 2014).

The high similarity in vegetation parameters between the two sowing treatments can be explained by the major overlap in species composition. Target species, which differentiate between the mixtures, were the species with almost complete failure or low establishment success. The failure of *H. sphondylium*, *T. vulgare* and *H. perforatum* can be explained by its specific seed dormancy strategies requiring cold stratification to alleviate seed germination (Baskin and Baskin 2001; Pérez-García et al. 2006; Royal Botanic Gardens Kew 2008). After sowing in April no frost events occurred in the first season and birds frequently visiting the area for ingestion (especially pigeons) might have removed at least the quite large seeds of *H. sphondylium*. Another lacking species, *M. arvensis*, is known to be a facultative winter-annual species (Grime, Hodgson, and Hunt 2007) and therefore might have failed to establish after sowing in April, whereas a lack of additional soil disturbance probably limited its germination in the second year. This means that for a fast revegetation of temporal brownfields species with complex germination strategies like morphopysiological dormancy with more or less pronounced after-ripening requirements in *H. sphondylium* (Baskin and Baskin 2001) are less suitable. Otherwise, if germination requirements are exactly known, methods to overcome seed dormancy (e.g. stratification) might be applied. For a quick successful revegetation, we additionally suggest pre-investigations of at least some soil parameters (e.g. soil pH) in order to select suitable site-adapted native species (e.g. according to Ellenberg indicator values, Ellenberg et al. 2001).

Effect of sowings on non-native, invasive non-native and Red-List species

In our study, the introduction of native plant species had no significant effects on non-native species in general or on the establishment of the potentially invasive non-native *S. inaequidens*. Although most non-native plant species probably have to be accepted as new members of urban ecosystems (Kowarik 2011; novel ecosystems, sensu Hobbs, Higgs, and Harris 2009), it has been shown that some of these species can be problematic for the conservation of valuable ecosystems like urban remnants of species-rich grasslands (Morgan 1998). As *S. inaequidens* is able to dominate early to mid-successional stages of brownfields (Heger and Böhmer 2005), further studies have to show if this may also result in suppression of valuable native plant species.

The Red-List species establishing from the spontaneous vegetation were predominantly species typical for (acidic) sandy dry grasslands and arable weed vegetation of sandy areas (Jäger 2011). The occurrence of these less competitive pioneer species can be understood by considering the history of land use and geological preconditions of the study area. Gleyic podzols developed from sandy alluvial sediments distributed along the local river system passing through the city of Osnabrück (LBEG 2014a). Historical land use in this area was low-input agriculture in the vicinity of suburban grazed grass- and heathlands (LBEG 2014b) followed by use as military barracks with extensive management of open space. These conditions probably favoured the spreading of species of dry grasslands before and during the military use.

In our experiment seeding reduced the cover of Red-List species compared to control plots at least in the short run. This means that the introduction of competitive native species of later successional stages would probably not be an adequate measure if the main restoration aim would be to conserve such pioneer species potentially occurring on young brownfields. Unlike in urban restoration measures on mid-successional brownfields lacking valuable pioneer species (see Fischer, von der Lippe, and Kowarik 2013). Depending on the specific aim of the restoration measure passive restoration (= restoration

Table 2: *F*-values and level of significance for the effects of treatment and organic C on several vegetation parameters (ANOVAs of GLMMs or LMs)

		First season				Second season			
		Treatment (df = 2)	C_{org} (df = 1)	Treatment × C_{org} (df = 2)	R^2	Treatment (df = 2)	C_{org} (df = 1)	Treatment × C_{org} (df = 2)	R^2
Vegetation height		0.752	**30.971** ↑***	–	$0.53_{(a)}$	**12.650** ***	–	–	$0.64_{(c)}$
Bare ground		0.611	**29.612** ↓***	2.126	$0.54_{(a)}$	**13.357** ***	**39.240** ↓***	–	$0.87_{(c)}$
Vascular plants	Number	1.234	**6.440** ↑*	–	$0.18_{(a)}$	0.675	2.296	–	$0.02_{(a)}$
	Cover	0.611	**29.612** ↑***	0.144	$0.54_{(a)}$	**13.834** ***	**38.910** ↑***	–	$0.87_{(c)}$
Target species	Number	**57.623** ***	4.275	**6.368** **	$0.91_{(c)}$	**38.637** ***	4.415	2.932	$0.91_{(c)}$
	Cover	**12.057** ***	**8.573** ↑**	–	$0.53_{(a)}$	**6.562** **	1.402	1.989	$0.76_{(c)}$
Forbs	Number	1.548	–	–	$0.04_{(a)}$	1.297	**14.369** ↓**	0.924	$0.35_{(a)}$
	Cover	0.538	**35.733** ↑***	2.153	$0.58_{(a)}$	1.506	–	–	$0.04_{(a)}$
Legumes	Number	**24.076** ***	**9.619** ↑**	–	$0.68_{(a)}$	**4.078** *	–	–	$0.19_{(a)}$
	Cover	1.760	**8.888** ↑**	1.384	$0.66_{(c)}$	**6.671** **	**16.522** ↑***	–	$0.76_{(c)}$
Grasses	Number	1.450	**19.394** ↑***	0.005	$0.34_{(a)}$	2.563	**6.655** ↑*	0.068	$0.21_{(a)}$
	Cover	2.207	**24.274** ↑***	3.088	$0.53_{(a)}$	0.926	**31.193** ↑***	3.060	$0.57_{(a)}$
Woody species	Number	0.145	0.123	1.074	$0.10_{(a)}$	2.228	3.422	–	$0.16_{(a)}$
	Cover	0.746	0.756	3.082	$0.64_{(c)}$	1.286	–	–	$0.29_{(a)}$
Non-native species	Number	1.470	–	–	$0.61_{(c)}$	0.317	1.537	0.520	$0.07_{(a)}$
	Cover	3.195	–	–	$0.14_{(a)}$	2.758	**13.191** ↑**	**3.468** *	$0.44_{(a)}$
S. inaequidens	Cover	1.542	–	–	$0.04_{(a)}$	2.099	2.012	–	$0.12_{(a)}$
Red-List species	Number	2.355	3.225	–	$0.16_{(a)}$	2.464	–	–	$0.10_{(a)}$
	Cover	0.876	**22.830** ↑***	1.490	$0.46_{(a)}$	**3.630** *	2.178	–	$0.20_{(a)}$

(a) = adjusted R^2, (c) = conditional R^2 according to Nakagawa and Schielzeth (2013). Bold letters indicate significant effects. Arrows show directions of significant effects. Significance level: *$P < 0.05$, **$P < 0.01$, ***$P < 0.001$.

Table 3: Means ± 1 SD of soil parameters for the study site ($n = 45$)

	Mean ± SD
pH[a]	6.87 ± 0.96
P_2O_5 (mg/100 g soil)[b]	4.36 ± 3.15
K_2O (mg/100 g soil)[a]	3.09 ± 1.42
C_{org} (%)[b,c]	0.45 ± 0.28
Total N (%)[c]	0.02 ± 0.01
Gravels > 2 mm (%)[a]	7.44 ± 6.24

Similar lowercase letters indicate intercorrelated parameters (see Supplementary Table S2).

without active human intervention; e.g. Albrecht et al. 2011; Kirmer et al. 2008; Prach and Pyšek 2001) may be possible if species of nature-conservation interest are present in the soil seed bank or in the surrounding vegetation. A distinct urban planning strategy should develop a spatio-temporal network of such open habitats (Kattwinkel, Biedermann, and Kleyer 2011) based on a metapopulation approach (Eriksson 1996) considering the regional species pool and habitat connectivity (Johnson et al. 2018). In addition, management strategies have to include disturbance regimes in brownfields of older successional stages (Schadek et al. 2009) if the aim is to promote threatened pioneer species. For brownfields with low soil-seed bank potential, however, native species introduction might be an effective measure to increase aesthetic values as well as to increase values for regional native plant diversity (Fischer et al. 2013).

Effects of soil conditions on the establishment of species groups

We found heterogeneous physical and chemical conditions of soils in our study area, which is a common phenomenon in urban areas (Gilbert 1989; Godefroid, Monbaliu, and Koedam 2007). The low contents of N, P and K indicate that the soil in the study area can be classified as immature soil (without distinct topsoil development), which is typical for young stages of urban demolition sites (Gilbert 1989). In our study, soil development indicated by C_{org} content as a proxy had a strong effect on vegetation structure. During successional processes C_{org} generally correlates with soil total N (e.g. Gleeson and Tilman 1990), and in our case also with exchangeable P content. Increasing vegetation cover and plant height with increasing soil fertility was also found in other studies on brownfield succession (Bornkamm 1986; Schadek et al. 2009). In our study, especially legumes and grasses benefitted from higher nutrient availability in more humus-rich soils. Potentially negative effects of dominating legumes and grasses at these more productive sites on Red-List species' populations have to be studied in future.

Aesthetical values

In our study the sowing treatments were obviously more colourful during the seasons (although not analysed in detail!) and showed a significantly larger vegetation height and structurally more diverse vegetation as compared to controls. It is known that at least Central European urban residents perceive areas containing diversely structured vegetation with large and colourful forbs imbedded in a matrix of green shaped grasses as particularly beautiful (Lindemann-Matthies and Bose 2007; Southon et al. 2017). Our sowing treatments met these criteria (Figs 1 and 2). We observed that the sites were frequently visited by pedestrians, who obviously appreciated the flowers (personal communications). As the 'interviewed' urban residents also noted, the vegetation of the sown plots seemed to hide urban waste, which often accumulates in 'neglected' brownfields, and hence appeared to be more aesthetic than the sparse pioneer vegetation of unsown plots. Further studies are required to study this assumed benefit for aesthetic appearance in detail. In addition, the restoration of urban brownfields might be regarded in a broader sense beyond the traditional restoration of historical conditions of an ecosystem (Standish, Hobbs, and Miller 2013). By creating structurally diverse and flower-rich aesthetic vegetation (restoration site as eye catcher), it may be possible to enhance urban residents' interest in nature, especially when guided tours and environmental education will be involved.

Conclusions

Our study showed that it will not be possible to reach all aims in the same area, which means that both sites for native species introduction and for spontaneous succession should be included in restoration strategies of urban open space. Using the natural potential of developing urban brownfields might be more cost effective, especially for sites which are only available for a short time, than the introduction of native plant seed mixtures. If valuable species from the spontaneous vegetation like threatened pioneer species occur on brownfields, ecological restoration might invest more cost and effort in lobbying activities for the acceptance by urban residents of these inconspicuous species rather than in introducing opulently flowering plant species of later successional stages.

Supplementary data

Supplementary data are available at *JUECOL* online.

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Data availability

Data sets used for analyses in this study are available upon request to the corresponding author.

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