Landscape and Urban Planning 170 (2018) 256–265

Contents lists available at ScienceDirect

Landscape and Urban Planning

journal homepage: www.elsevier.com/locate/landurbplan

Research Paper

‘Wild’ in the city context: Do relative wild areas offer opportunities for urban biodiversity?

Anke Müller[a,c,*], Peder K. Bøcher[a,b], Christina Fischer[c], Jens-Christian Svenning[a,b]

[a] Section for Ecoinformatics & Biodiversity, Department of Bioscience, Aarhus University, Aarhus, Denmark
[b] Center for Biodiversity Dynamics in a Changing World (BIOCHANGE), Aarhus University, Aarhus, Denmark
[c] Restoration Ecology, Department of Ecology and Ecosystem Management, Technical University Munich, Freising, Germany

ARTICLE INFO

Keywords:
Biodiversity
Urban greenspace
Urban habitats
Bioscore
Wildness mapping
Urban wildscapes

ABSTRACT

Urbanization is increasing worldwide, making it essential to improve management of urban greenspaces for better provisioning of ecosystem services and greater biodiversity benefits. At the same time, societal interest in reduced intensity management regimes is growing for a range of practical and normative reasons. We assessed if relative wild urban greenspaces, under little or no management, are associated with increased levels of biodiversity. We conducted a GIS-based relative wildness mapping for the Danish city Aarhus, and compared relative wildness to field-measured perceived biodiversity at 100 randomly placed sample sites in the city centre. Perceived biodiversity was estimated using the bioscore methodology. The results show a positive relationship between mapped wildness and bioscores, notably within artificial vegetated areas such as parks and gardens, while woodland had the highest wildness and bioscore values overall. All bioscore components measuring structural diversity increased with increasing mapped wildness. The bioscore component compositional richness covered site-level species richness for birds, invertebrates and plants, with invertebrate and bird species richness increasing and plant species richness decreasing with increasing wildness. The latter reflects that woodlands had low site-level plant diversity. Overall, woodlands nevertheless harboured many unique plant species, with woodlands and ruderal areas contributing the greatest beta diversity (inter-site variability in species composition). These findings show that urban greenspace management allowing for spontaneous ecological processes (greater wildness) overall also promotes urban biodiversity, pointing to potential synergies between urban design and management goals for reduced management intensity, increased wildness experiences, and higher biodiversity in urban greenspaces.

1. Introduction

Given increasing urbanization worldwide (Chen, Zhang, Liu, & Zhang, 2014), it is important to understand if and how urban greenspaces can be managed for better provisioning of ecosystem services and greater biodiversity benefits. For this reason, urban ecology has gained momentum in recent decades, with the first ecosystem studies carried out in urban areas dating back to the 1970s (Sukopp, 2008). In cities, human activities are the main drivers of ecological processes and patterns (Warren et al., 2010), and urban greenspaces often do not consist of the natural habitat types, but rather of novel ecosystems, systems, that ‘[…] have been potentially irreversibly changed by large modifications to abiotic conditions or biotic composition’ (Hobbs, Higgs, & Harris, 2009). Nevertheless, they can sustain important ecological functions such as nutrient absorption, heat reduction or erosion control and serve as wildlife habitats (Del Tredici, 2014). Urban greenspaces have also been shown to provide important ecosystem services such as the filtration of air and micro-climate regulation that increase the living quality for urban citizens (Bolund & Hunhammar, 1999). Furthermore, exposure to urban biodiversity may have positive health benefits (Cox et al., 2017).

At the same time, there is increasing societal interest in reduced-intensity management regimes for a range of practical and normative reasons (Buck, 2015). At a European level, there is increasing focus on wilderness protection and restoration, as one key approach to avoid and reverse biodiversity losses (European Parliament, 2009). Notably, the maintenance and development of wilderness in nature protection areas is advocated (European Commission, 2013). Additionally, there is strongly increasing interest among both managers and scientists worldwide and in Europe towards rewilding as a strategy for

* Corresponding author at: Research Unit Sustainability and Global Change, University of Hamburg, Grindelberg 5, 20144 Hamburg, Germany
E-mail addresses: anke.mueller@tum.de, anke.mueller@mpimet.mpg.de (A. Müller), peder.bocher@bios.au.dk (P.K. Bøcher), christina.fischer@tum.de (C. Fischer), svenning@bios.au.dk (J.-C. Svenning).

https://doi.org/10.1016/j.landurbplan.2017.09.027
Received 22 February 2017; Received in revised form 17 August 2017; Accepted 25 September 2017
Available online 20 November 2017

biodiversity conservation and natural area management (Corlett, 2016a; Jepson, 2016; Svenning et al., 2016). The concept covers a range of variants, but a common aspect is reduction of human management and restoration of self-managing ecosystems (Navarro & Pereira, 2012). One prominent version in Europe is passive rewilding, which is simply the cessation of human management (Corlett, 2016a). Even though rewilding naturally focuses on rural and natural landscapes, the applicability of the concept to urban settings calls for exploration, especially regarding increasing urbanization. There is emerging evidence that urban wastelands (defined as abandoned sites with spontaneous vegetation) can contribute importantly to urban biodiversity, generally harbouring more species than other urban greenspaces (Bonthoux, Brun, Di Pietro, Greulich, & Bouché-Pillon, 2014). More broadly, there is also increasing interest in exploring possibilities in cities for not just more unmanaged and spontaneous ecological, but also social dynamics in greenspaces to improve the liveability of cities (Jorgensen & Keenan, 2012). The diversity of urban resident groups is reflected in a diversity of recreational needs, and unmanaged urban greenspaces offer unique opportunities for nature experiences, discovery and a range of informal activities (Rupprecht & Byrne, 2014).

A key issue for studying ecological wildness and its services and disservices in an urban setting concerns its definition and measurement, given the pervasive human influence on urban landscapes. Here, the wilderness continuum concept (Carver, Comber, McMorran, & Nutter, 2012) is useful: Instead of a binary definition of 'wild' and 'not wild', it acknowledges a gradient of human modification of landscapes. It allows us to define parts of a landscape as 'wilder' and 'less wild' compared to other parts within a given geographic scope. Relative wildness mapping based on this concept and conducted in Geographic Information Systems (GIS) have been carried out ranging from worldwide assessments to national, regional and even local scales (Carver et al., 2012), enabling the examination of relative wildness in anthropogenic landscapes (Müller, Bøcher & Svenning, 2015). GIS-based relative wildness mapping should therefore also allow us to assess the relative wildness of urban greenspaces.

The overall aim of the present study was to investigate if relative wild urban areas harbour particularly high levels of biodiversity in a European city. This setting is highly relevant for investigating the wildness-biodiversity link, as 70% of the European population live in cities, predicted to further increase by 10% by 2050 (European Union, 2011). Hence, urban areas in this region will continue to grow, forming the setting where an increasing proportion of the overall population will experience ecological wildness and biodiversity on a daily basis. We first assessed the applicability of GIS-based relative wildness mapping to the urban study setting. We then tested for a positive relationship between relative wildness and biodiversity, as assessed in a field survey. Finally, we assessed how relative wildness and biodiversity varied among major urban habitat types.

2. Methods

2.1. Study area

Our study was conducted in Aarhus Municipality, situated in the Central Jutland region in Denmark (Fig. 1) at the coast of the Baltic Sea, with an area of 476.85 km^2 (Statistics Denmark, 2016a) and 331,332 inhabitants (Statistics Denmark, 2016b). It consists of the city of Aarhus and several rural communities. The city of Aarhus is the next-largest city in Denmark and the fastest growing in the whole country (Aarhus Kommune, n.d.). The GIS-based relative wildness mapping was conducted for the whole municipality, whereas the fieldwork to collect biodiversity data was carried out only in the city centre to focus on the most urbanized parts. The city centre was defined as the area within Ringvejen, the outer ring road of Aarhus (Fig. 1, yellow border).

2.2. Wildness mapping

We chose the following four indicators to represent urban wildness in this mapping based on two previous relative wildness mapping studies (Scottish Natural Heritage, 2014; Müller, Bøcher & Svenning, 2015): (1) perceived naturalness of land cover, (2) challenging terrain, (3) remoteness and (4) visibility of built modern artefacts.

For perceived naturalness of land cover, land cover data were mainly derived from Basemap 2012 (Levin, Jepsen, & Blemmer, 2012). Data on agricultural land use were updated from Markkort 2015 (Danish Agrifish Agency, 2015). Two land use classes from Basemap, 'land' and 'unclassified' were reclassified into other land use classes from Basemap by doing a spatial overlap with polygons from KORT10 (Danish Geodata Agency, 2013) and by comparison to orthophotos (COWI, 2014). All the joined land use classes (hereafter referred to as 'the land use dataset') were reclassified into 20 naturalness classes ranging from 'completely sealed areas' with the lowest naturalness value over 'permanent grassland with normal yields' (naturalness class 10) to 'land cover presumably under least human influence' (for a detailed description of the 20 classes, see Table S1, Supplementary data).

To describe challenging terrain, terrain ruggedness and occurrence of wetlands were combined as suggested in previous work (Scottish Natural Heritage, 2014). For ruggedness of terrain, the curvature of a 1.6 m resolution digital terrain model (DTM) (Danish Geodata Agency, 2007a) was calculated. Afterwards the dataset was aggregated and resampled into a 10-m resolution. At this fine scale, a DTM does not only capture the actual terrain, but also anthropogenic structures such as raised roads. When calculating the standard deviation of the curvature, these structures would also tend to show high values. Consequently, the possibly higher ruggedness in such places does not necessarily capture places that are perceived as wild, probably rather the opposite. Therefore, all pixels with construction (roads, railways, buildings) were excluded from the dataset. Afterwards, the standard deviation for each cell in a 250 m neighbourhood was calculated, reflecting the area an individual would consider his or her immediate surrounding. To fully cover challenging terrain in terms of physical properties of the ground, information on the occurrence of wetlands (layer 'vådområde' (wetlands) of KORT10 (Danish Geodata Agency, 2013) and layer 'mose' (swamps) of the protected nature types dataset (Danish Natural Environment Portal, 2007)) was added to the terrain ruggedness dataset: If a pixel cell laid within wetland, 0.3 (mean standard deviation value of terrain ruggedness calculation), was added to the pixel cell value.

The indicator remoteness was depicted by remoteness from mechanized access and noise exposure. Remoteness from mechanized access was measured by calculating the shortest walking distances from mechanized access (major roads) to any pixel on the map following Carver et al. (2012). The land use dataset was reclassified into a cost surface (Table S2, Supplementary data), estimating the seconds it takes a person to pass through each pixel based on assumed travel times for each land use class. The shortest time it would take a hiker to access any pixel in the study area from a point of mechanized access was then calculated by using the pathdistance tool of ArcGIS. Noise exposure from roads, railways and agglomerations was available for most parts of the study area (Danish Environmental Agency, 2012). The weighted means of noise values (L_{den}) were chosen for calculation using the highest measured decibel value. The data for roads, railways and agglomerations (areas of high population density) were then merged, always choosing the highest decibel value if datasets overlapped. All pixels not covered by the existing noise exposure data were considered rather quiet. They were assigned 30 dB, similar to a quiet garden (Cercle Bruit Schweiz, 1998). The dataset was reclassified into inverse values, so high values in the dataset would depict low decibel values and vice versa, ensuring that values for all indicators correlated positively to likely wildness experience. Afterwards, the remoteness of mechanized access dataset and the noise dataset were summed up to

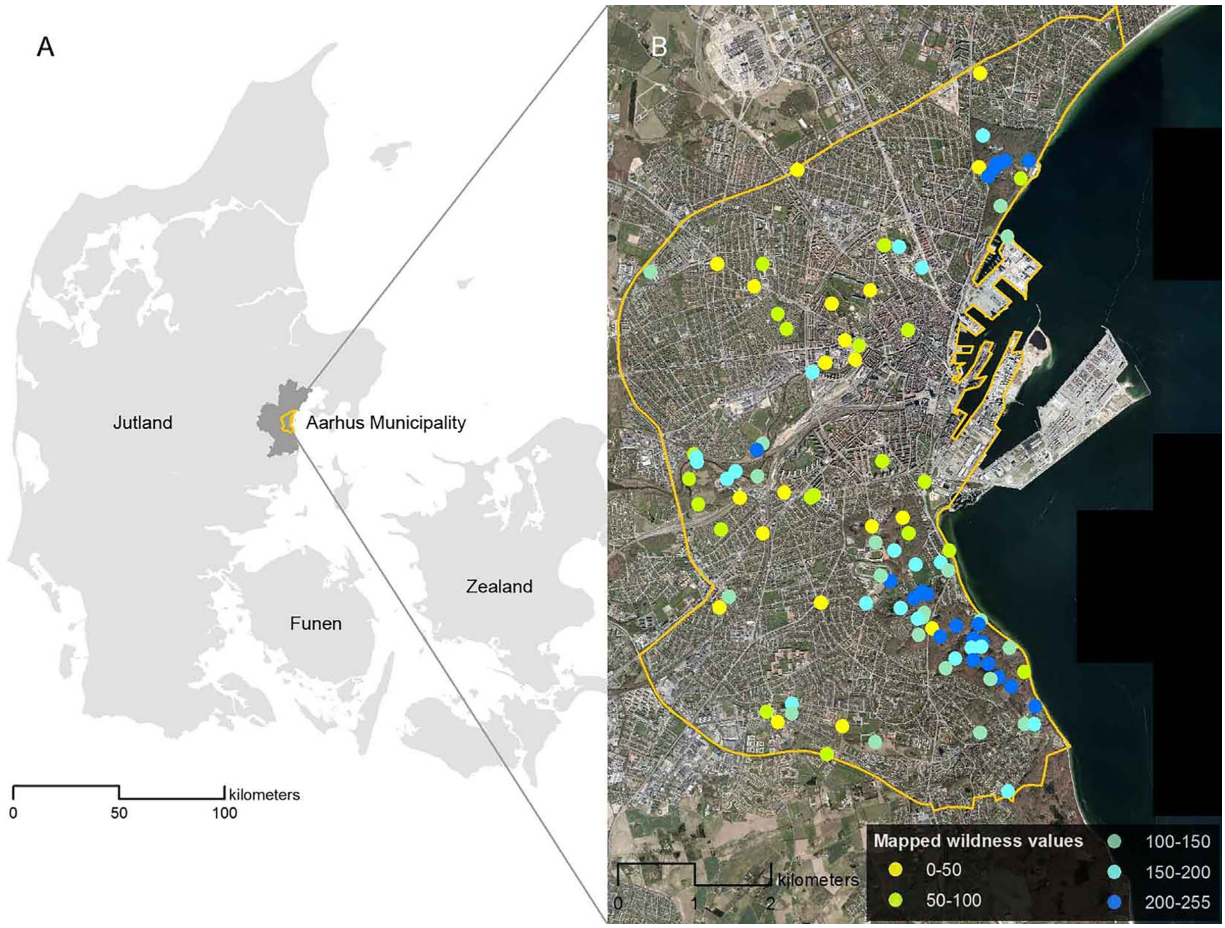

Fig. 1. Location of the study area Aarhus Municipality (A). B depicts the distribution of the sample points, randomly placed in the city centre of Aarhus, stratified across five classes of mapped urban wildness values (20 sample points per class, see different colours of the sample points).

create the final remoteness dataset.

To evaluate the indicator visibility of built artefacts, a viewshed analysis was done, following Scottish Natural Heritage (2014) methodology. In a first step, all layers from KORT10 (Danish Geodata Agency, 2013) containing built artefacts were selected: single buildings, facilities, buildings, railways, borders of all kinds of streets, landing stages, harbours, groynes, chimneys, light posts, telephone poles, windmills, dykes, high-tension power lines and protected ancient monuments. This vector dataset was converted into a raster mask with a 9.6 m resolution and each pixel being assigned a 1 if a built artefact lay within and a 0 if no built artefact lay within the pixel. A digital surface model, DSM (Danish Geodata Agency, 2007b), the standard terrain descriptor utilized for view shed analysis in ArcGIS, was aggregated to a 96 m resolution to create a point observer layer. For this, a point was inserted in the cell centre of each pixel. This created 57,176 observer points. Using an iterator in the model builder of ArcGIS, a view shed was calculated for each observer point. The visibility was set to be infinite. A maximum visibility was not applied as visibility can differ a lot depending on the height of the viewed structures and the elevation of the observer (geometric visibility) and also on the current weather conditions (meteorological visibility). The infinite visibility certainly overestimates visibility and can be understood as a conservative measure. The resulting view shed maps for each observer point were each multiplied with the built artefact mask leaving only pixels containing built artefacts. Then the number of those pixels was stored for each observer point and all the single observer point datasets were merged together. This dataset was converted back into a raster dataset with 96 m resolution, each pixel storing the number of built artefacts that were visible from this pixel. For the wildness perception, the number of visible artefacts per pixel was reclassified into four classes. A change in low numbers of artefacts was expected to have a large influence on wildness experience, whereas it probably does not matter much if an observer can see 10 or more than 10 artefacts. The following reclassification was thus applied: 4, only one artefact visible; 3, 2–5 artefacts visible; 2, 6–10 artefacts visible; and 1, ≥ 10 artefacts visible.

All modelling was done in ArcGIS 10.3 (Environmental Systems Research Institute (ESRI), 2017), using mainly the Spatial Analyst extension. The coordinate system utilized for this modelling was UTM zone 32 N with datum ETRS 1989. Datasets were either available or aggregated/resampled to a 10 m-resolution. All four indicator maps were summarized without the application of weights (equal weighted) to create the final urban wildness map of Aarhus Municipality. All single descriptor maps, the four indicator maps and the final urban wildness map were standardized to a common 0–255 scale using the following equation:

$$s_{ij} = \frac{(x_{ij} - \textit{old minimum value})*(\textit{new maximum value} - \textit{new minimum value})}{\textit{old maximum value} - \textit{old minimum value}} + \textit{new minimum value},$$

where s_{ij} is the standardized value of cell j in map i and x_{ij} is the current value of cell j in map i.

2.3. Collecting data on biodiversity

The urban wildness map of Aarhus Municipality was clipped to the extent of the smaller central urban study area (Fig. 1) and the

Table 1
Parameters measured in each 10-m diameter sampling plot to calculate the bioscore following Hand et al. (2016). Weather condition and time of day – potential confounding factors for bird and invertebrate species richness – as well as habitat type were also recorded.

Indicator	Parameter	Description	
Structural complexity	Vegetation strata	Number of strata (< 1 m, 1–2 m, 2–5 m, 5–12 m, > 12 m height) by vegetation.	
	Plant growth forms	Number of plant growth forms, possible choices: large tree (woody plant with < 3 main trunks, > 20 cm d.b.h.), small tree (woody plant with < 3 main trunks, < 20 cm d.b.h.), tall shrub (woody plant with > 3 main trunks, > 1 m tall), small shrub (woody plant with > 3 main trunks, < 1 m tall), tall grasses and ferns (grasses over 1 m in height, ferns and flax plants), herbaceous plants (green-stemmed plants excluding grasses), grasses (grass and grass-like species < 1 m in height), lichens and mosses (lichens, mosses and clubmosses), aquatic (freshwater and marine vegetation), climbers (plants growing on other structures).	
Compositional richness	Plant species richness	Number of registered plant species per 79 m^2.	
	Bird species richness	Number of registered bird species per 79 m^2.	
	Invertebrate species richness	Number of recognizable taxonomic units (RTU) for invertebrates per 79 m^2.	
Wildness	Naturalness	Estimated naturalness of the plot, with five possible classes: 0 = completely artificial (e.g. concrete surface), 1 = predominantly artificial habitat with some natural features present, 2 = mix of natural and artificial features, 3 = natural habitat with some alteration and 4 = natural habitat with minimal alteration. Planted non-native plant species were treated as artificial features.	
	Management	Estimated management of the plot, with five possible classes: 0 = completely controlled environment, 1 = mostly human-influenced environment, but some natural features, 2 = half-controlled, half-natural features, 3 = slightly visible human influence, 4 = No visible human influence.	
Greenness	Vegetation cover	Vegetation cover of the ground and of the canopy above the plot (in %). Both were summarized to describe greenness (maximum possible value being 200%).	
Additional	Weather condition	Six categories: sunny; clouded < 50%; clouded > 50%; closed cloud cover; light rain; strong rain.	
	Time	Starting time of bird and invertebrate evaluation.	
	Habitat type	Habitat types after Hand et al. (2016)*, afterwards reclassified into four broader urban habitat types:	
		Artificial vegetated areas (n = 22)	garden rich and garden poor residential areas, recreational green, parks
		Woodland (n = 46)	woodland
		Paved (n = 16)	residential streets, streets, recreational paved, open public area
		Semi-natural and ruderal areas (n = 16)	vacant lots and fringe vegetation, non-woodland area largely maintained in a native state

*Only the habitat types that were recorded in this study are listed here.

remaining values were again standardized to the 0–255 scale. The standardized values were then reclassified into five equally spaced wildness classes (0–50, 51–100, 101–150, 151–200, 201–255). Built-up sites (buildings, big streets, railways) and private areas such as gardens were erased from the dataset as sampling was not possible there. Afterwards, 20 points were randomly placed within each of the five wildness classes. A minimum distance of 50 m between sampling points was enforced to keep the points from clustering in one area. Sample sites were relocated to the centre of the wildness map pixel.

A biodiversity assessment was conducted between 22nd June and 5th August 2016. The sample points were located using a Trimble Juno SB handheld GPS device running ArcPad 10. In each location, a circle with a 10 m diameter was set up. The biodiversity of the sample sites was assessed employing the bioscore method of Hand, Freeman, Seddon, Stein, and van Heezik (2016). It consists of the four components structural complexity, compositional richness, wildness, and greenness, as defined and measured in Table 1. We kept 'wildness' as a bioscore component even if it measures perceived wildness of the sample sites, which we also evaluated with the GIS-based urban wildness mapping. As wildness was only one of four components, measured in a very different manner from the GIS-based wildness mapping (cf. Table 1), we preferred to use the bioscore as originally suggested to enable comparison to other studies. Furthermore, we also assessed the individual relations between the four bioscore components and mapped wildness (see below), allowing us to separate out the relation between the two wildness measures. To assess these four components, the following data were recorded in each sample site: number of vegetation strata occupied and number of plant growth forms. Bird species richness, estimated by standing five minutes in the plot centre and noting how many different bird species could be seen or heard anywhere around the plot. Invertebrate species richness assessed searching the plot optically five minutes for invertebrates and noting down how many different recognizable taxonomic units (RTU) were found. Plant species richness was surveyed by estimating all higher plant species within the circle to species level for each of the following strata: herb layer (< 0.5 m), shrub layer 1 (0.5–2 m), shrub layer 2 (2–5 m) and tree layer (> 5 m), and their Braun-Blanquet cover (Müller-Dombois & Ellenberg, 1974). Strata information was not utilized for calculating the bioscore but for the examination of plant species composition. Furthermore, naturalness, management, vegetation cover, time of day, weather condition and habitat type were registered in each plot (Table 1). Based on this information, the bioscore for each sample site was calculated by the following equation:

$$\text{bioscore} = (s_1 \times \text{structural complexity} + s_2 \times \text{compositional richness} + s_3 \times \text{wildness}) \times \text{greenness},$$

where s_1, s_2 and s_3 are scaling factors, scaling all three summarized components to a maximum value of ten, whereas greenness was standardized to a 1–4 range beforehand.

2.4. Statistical analysis

All statistical analyses were conducted using R version 3.2.3 (R Core Team, 2015). The relationship between mapped urban wildness values and bioscores was examined using standard least-squares linear regression. Additionally, for each of the four components of the bioscores (compositional richness, structural diversity, wildness, and greenness) a separate linear regression was calculated.

As compositional richness was the only component of the bioscore showing no statistically significant relation to the mapped urban wildness values, we looked further into its components. Plant species richness in relation to mapped urban wildness values was analyzed using linear regressions. Bird species richness and invertebrate richness were expected to be confounded by the time of day and the prevailing weather condition. To account for these confounding factors leading perhaps to heteroscedastisity, generalized least squares linear models (gls) implemented in the R-package nlme (Pinheiro, Bates, DebRoy, Sarkar, & R Core Team, 2016) were utilized. Different variances per stratum in the case of categorical explanatory variables (weather conditions) were modelled using the varIdent variance structure. For the numerical explanatory variable 'time of day', a fixed variance structure was applied using varFixed. Additionally, a combination of both variance structures was tested to improve residual homogeneity. The Akaike Information Criterion (AIC) of all three modelling variants was compared and the model with the smallest AIC was chosen. To examine the fit of the gls a pseudo-r^2 was computed using the predictive power measure (Zheng & Agresti, 2000).

Community species composition of the vegetation was visualized with a Non-Metric Multidimensional Scaling (NMDS, R-package vegan (Oksanen et al., 2016)) to assess differences between habitat types. The NMDS was run with the Bray-Curtis-distance metric and three dimensions ($k = 3$), as reasonable stress values could not be obtained with two dimensions. As environmental parameters habitat types, mapped urban wildness values and bioscores were fitted and *p*-values were obtained from random permutations of the data.

To check for a difference in bioscores depending on the four habitat types a one-way ANOVA and for a difference in urban wildness values a Kruskal-Wallis rank sum test, due to non-normal distribution of the response variable, was calculated. In both cases Tukey post-hoc tests were employed to check for differences among factor levels. Furthermore, a linear regression comparing urban wildness values and bioscores was calculated for data subsets for each habitat type. To control for possible spatial autocorrelation of the sample points in each subset, Pearson correlations with geographically effective degrees of freedom (Dutilleul's method) were calculated in SAM – Spatial Analysis in Macroecology (Rangel, Diniz-Filho, & Bini, 2010). There was no spatial autocorrelation detected for any subset.

3. Results

3.1. Relative wildness mapping

From a first visual assessment, the relative wildness mapping succeeded in distinguishing between different management intensities of certain parts of the city landscape. While mainly streets and built-up areas were mapped as least wild, forests and wetlands scored highest at the whole municipality level (Fig. 2). Naturally, the city centre was overall mapped less wild than the urban fringes or rural parts of Aarhus municipality but the mapping still returned very high values for forest and river greenspaces located there.

As wildness was also a component of the bioscore, estimated on-ground for each sample site, a comparison to the mapped urban wildness value for each site by linear regression was utilized as a ground-truthing. The significant positive relationship (linear regression, slope = 0.02, $t_{96} = 6.43$, $p < 0.001$, $r^2 = 0.30$) revealed that the mapping was capable of capturing differences in urban wildness that would also be perceived on-ground, with high mapped urban wildness values corresponding to rather natural settings and sites with low urban wildness values being under obviously strong human influence (Fig. 3).

We investigated the robustness of our mapping performing a sensitivity analysis (Appendix A). The resulting maximum standard deviation of 5.9 (2% variability relative to the full 0–255 value range) and mean standard deviation of 2.8 (1% variability relative to the full 0–255 value range) show that our mapping produced reasonably robust results, with major roads and certain parts of large greenspaces being least sensitive to a change in indicators (Fig. S4, Supplementary material). Furthermore, remoteness (from noise and mechanized access) seems to be the indicator influencing the final wildness mapping results the most. Correlation coefficients of the original wildness map and the four wildness maps only taking three indicators into account, respectively, all showed values > 0.7. Apparently, the general trend in pixel values stays the same, even when one indicator input is left out of the calculation.

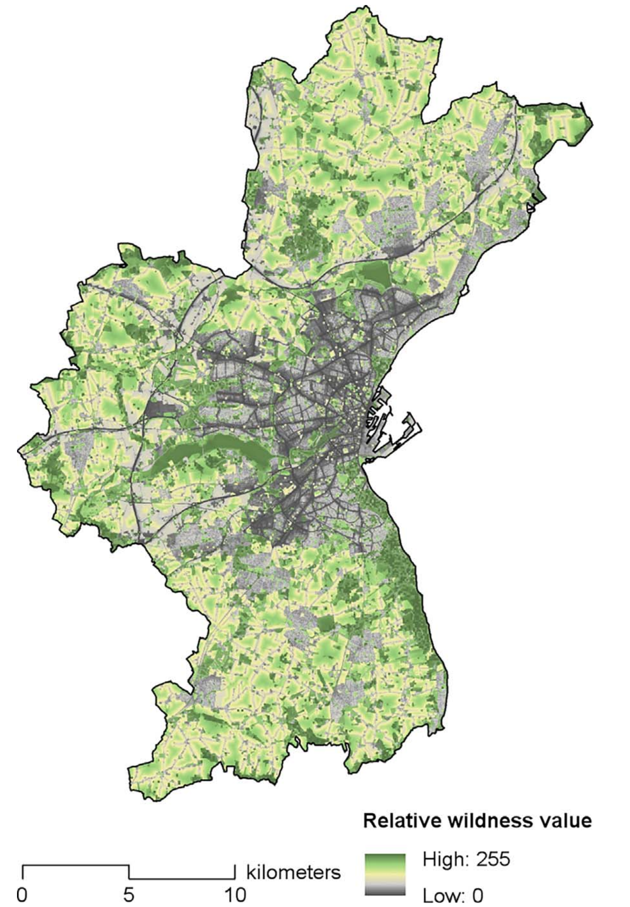

Fig. 2. Urban wildness map: Landscape-scale relative wildness mapping for Aarhus Municipality on a 10 m resolution.

3.2. Relation between urban wildness and bioscore and comparison across habitat types

There was a positive relationship between mapped urban wildness and bioscore (slope = 0.12, $t_{96} = 4.38$, $p < 0.001$, $r^2 = 0.17$, Fig. 4). Furthermore, linear regressions revealed positive relationships between the mapped urban wildness values and the bioscore components; structural complexity (slope = 0.01, $t_{96} = 2.93$, $p < 0.01$, $r^2 = 0.08$), wildness (slope = 0.02, $t_{96} = 6.43$, $p < 0.001$, $r^2 = 0.30$), and greenness (slope = 0.01, $t_{96} = 3.52$, $p < 0.001$, $r^2 = 0.11$). The linear regression for mapped urban wildness and compositional richness showed no significant relationship (slope = −0.01, $t_{96} = -0.41$, $p = 0.69$, $r^2 = 0.01$). The only significant relationship between mapped urban wildness and bioscore for the data subsets for each habitat type was found for artificial vegetated areas (Table 2).

Mean urban wildness values (mean ± se) according to habitat types were: artificial vegetated area 105.1 ± 12, woodland 159 ± 8.9, paved 44.8 ± 1.4 and semi-natural/ruderal areas 141.4 ± 17.1. There was a significant difference (Kruskal-Wallis rank sum test: $p < 0.05$) for mean urban wildness values across the four habitat types (Fig. 5A). The lowest bioscore (0.07) was found in a parking lot (habitat type: paved), while the highest bioscore (82.45) was recorded for a riparian forest (habitat type: woodland). There was a significant difference (ANOVA: $p < 0.05$) found for mean bioscores across the four habitat types (Fig. 5B).

Fig. 3. Sample sites with the highest (upper row) and lowest (lower row) mapped urban wildness values (depicted on the photos). The standardized value range of the mapping was 0–255, as sample sites were placed randomly, they do not display the full value range of the mapping.

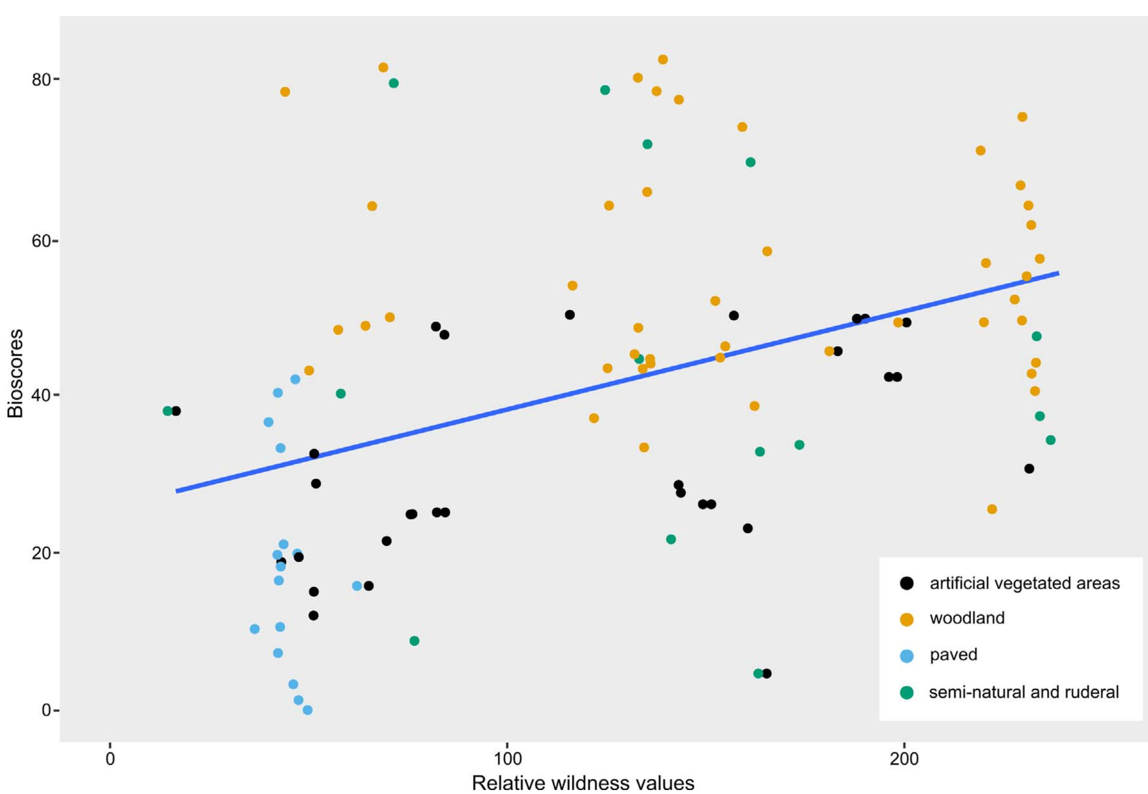

Fig. 4. The linear regression ($p < 0.001$) revealed a positive relationship between urban wildness values and bioscores. The sample points are coloured according to habitat type.

Table 2
Linear regressions comparing mapped urban wildness values and bioscores for subsets of the data per habitat type.

Habitat type	Regression coefficient	Df	t	p	Pearson's r	SAC-corrected p
Artificial vegetated areas	0.10	20	2.4	0.03	0.47	0.01
Woodland	−0.03	43	−0.7	0.49	−0.11	0.47
Paved	−0.46	14	−0.73	0.48	−0.19	0.50
Semi-natural and ruderal areas	−0.03	13	−0.34	0.74	−0.10	0.25

3.3. Bird, invertebrate and plant species richness in relation to urban wildness

There was a (marginal) positive relationship between urban wildness values and bird species richness (slope = 0.01, t_{98} = 1.84, p = 0.07, pseudo-r^2 = 0.03), as well as invertebrate species richness (slope = 0.02, t_{98} = 3.77, $p < 0.001$, pseudo-r^2 = 0.12). For plant species richness and urban wildness values, a negative relationship was found (slope = −0.03, t_{96} = −2.76, p = 0.01, r^2 = 0.07). The NMDS revealed that vegetation community composition can be explained by habitat type (envfit: r^2 = 0.52, p = 0.001), urban wildness value (envfit: r^2 = 0.23, p = 0.001) and bioscore (envfit: r^2 = 0.44, p = 0.001) (Fig. 6). Whereas semi-natural/ruderal areas, artificial vegetated areas and paved areas largely overlap, woodland did not overlap with any other habitat type. Plant species composition of woodland differed from the other habitat types, with many species being found only in this habitat type (list of registered species in Table S3, Supplementary data). Furthermore, the ordiellipses of woodland and semi-natural and ruderal areas are larger than the ones for paved and artificial vegetated areas (Fig. 6), indicating greater beta diversity among sample sites of these habitat types.

4. Discussion

4.1. Relative wildness mapping in urban areas

Landscape-scale relative wildness mapping has so far mostly focused on large natural areas (Carver et al., 2012; Carver, Tricker, & Landres, 2013). We applied GIS-based wildness mapping to inventory urban greenspaces according to their perceived relative wildness. One needs to be careful when communicating such mapping results as they may be sensitive to errors and uncertainties related to data inputs and algorithms used (Feizizadeh & Blaschke, 2014). Promisingly though, here we found our GIS-based mapping was robust to changes in indicator inputs. Furthermore, our ground-truthing results showed that the GIS-based mapped urban wildness values and field-estimated wildness (bioscore component) for each sample site corresponded well, even though there is substantial unexplained variation.

4.2. Urban wildness areas and biodiversity

Bioscores showed a clear increase with increasing mapped relative urban wildness values. The bioscore components that represent habitat diversity, namely greenness, wildness and structural complexity, also increased with increasing urban wildness values. As these three measures of habitat diversity have been shown to increase biodiversity in previous studies (e.g. Coops, Fontana, Harvey, Nelson, & Wulder, 2014; Jorgensen & Tylecote, 2007; Threlfall, Williams, Hahs, & Livesley, 2016), these findings point to a positive relationship between urban wildness and urban biodiversity. Nevertheless, the low r^2 values indicate that a substantial part of variation could not be explained by the mapped relative wildness values. We note that some sites in the dataset with high mapped relative wildness values, but low bioscores or vice versa reflect conditions where wildness simply does not correlate with biodiversity for the measured groups. These include sites in beech forest with highly shaded conditions, a thick layer of fallen leaves, and little understorey vegetation (a typical effect of beech (Mölder, Bernhardt-Römermann, & Schmidt, 2008)), accounting for high wildness perception, but low bioscore measures. Other sites located within parks or forest, but very close to major roads tended to show high bioscores, but low mapped wildness values, as proximity to roads does not have strong negative or may even sometimes have positive effects on the measured organism groups (Skov & Svenning, 2003). Linked to this, we find that when considering the four wildness mapping indicators separately all are positively correlated to the bioscores for the whole dataset, but relationships within habitat types are generally not significant (Fig. S5, Supplementary data). This suggests that much of the bioscore-wildness relation is mediated by habitat type, which itself exhibits much variation beyond that linked to wildness, helping to explain the noisy relation between bioscore and perceived wildness. However, there is one significant and four marginally significant ($p < 0.1$) positive relations within habitats between bioscore and either perceived naturalness of land cover, remoteness or visibility of built modern artefacts (Fig. S5, Supplementary data), suggesting that these indicators capture aspects of the environment of relevancy for biodiversity beyond those captured by habitat type. Future research should further investigate the link between wildness and biodiversity within urban habitat types, e.g. for more organism groups.

Compositional richness, measuring biodiversity in a classical assessment of species richness showed no significant relationship to

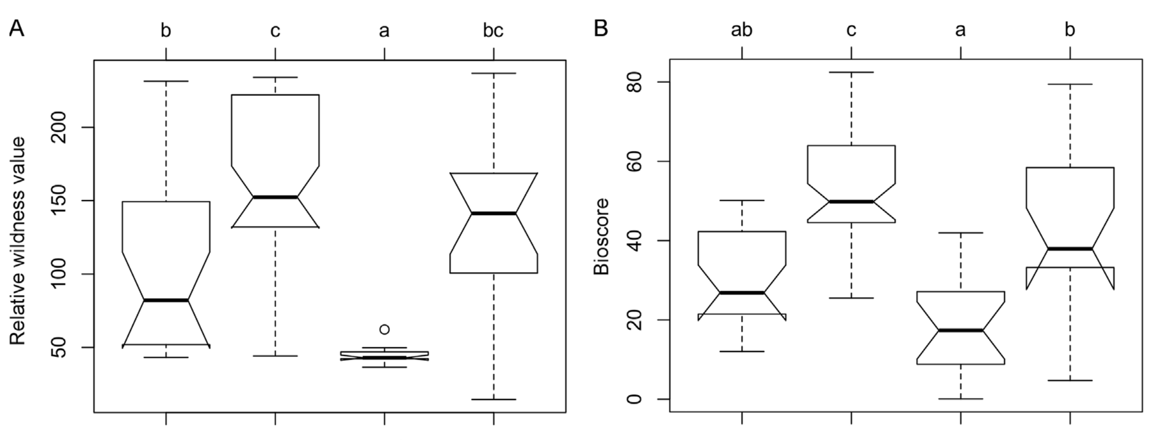

Fig. 5. Box plots of urban wildness and bioscore values per habitat type depicting median values. Different letters indicate significant differences among habitat types.

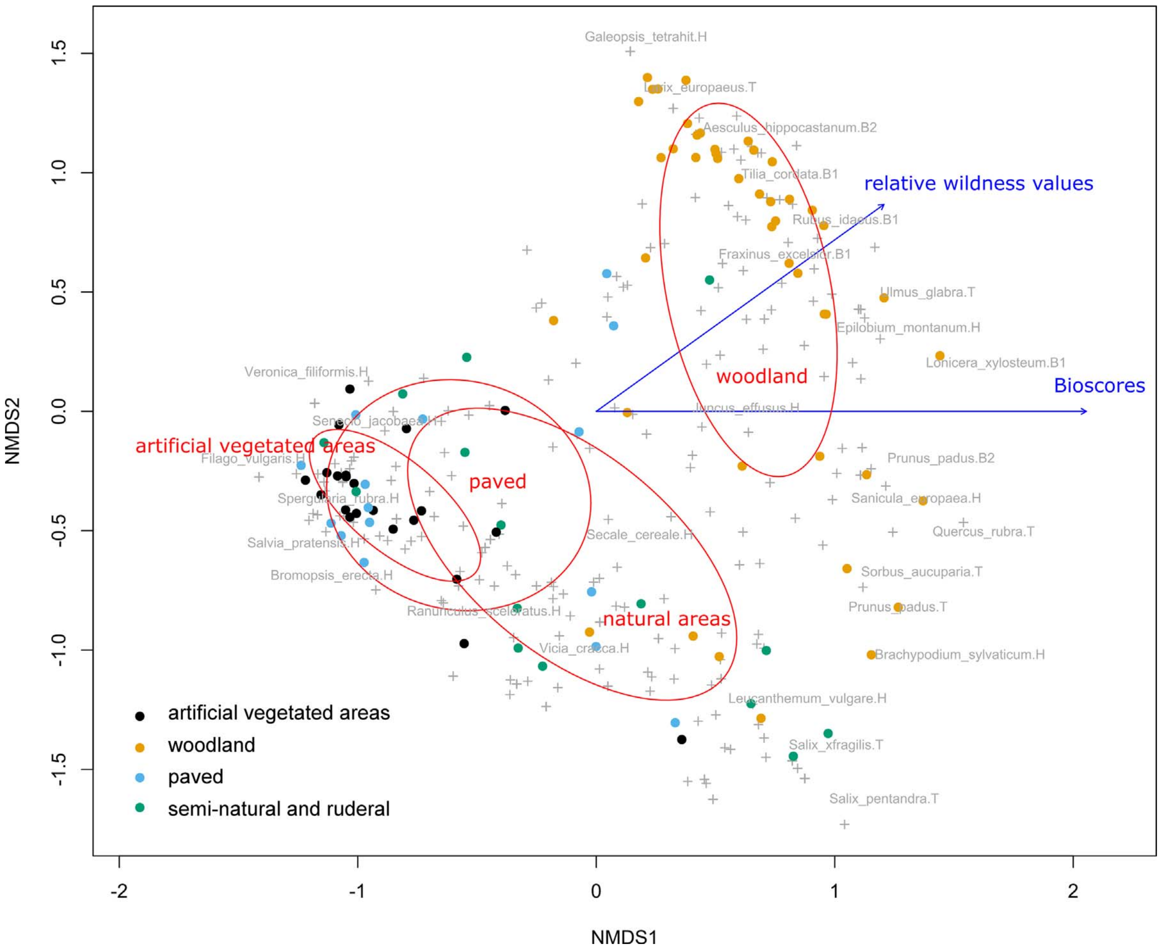

Fig. 6. NMDS for vegetation community composition (stress value: 16%). Ordiellipses in red show the community composition of the single habitat types. Dots depict the sample sites with colours according to habitat type, while species are represented by grey crosses (only certain species names are plotted to prevent cluttering). Species with a 'H' at the end of their name were found in the herb layer, species with a 'B1' or 'B2' in the shrub layer and species with a 'T' in the tree layer. The environmental parameters relative wildness values and bioscores are illustrated by blue arrows. The NMDS result was rotated so that NMDS1 is parallel to the environmental parameter bioscores. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

mapped urban wildness values. Still, bird species richness showed a marginally positive tendency in relation to increasing urban wildness, while invertebrate diversity increased significantly with increasing urban wildness. Studies of bird and invertebrate species richness in urban habitats show they are closely related to habitat diversity (Dallimer et al., 2012; Paker, Yom-Tov, Alon-Mozes, & Barnea, 2014), presence of old trees and high-quality remnants of natural vegetation (Angold et al., 2006). All these mentioned characteristics are more likely provided by areas with higher urban wildness values, as these are areas with less sealed surface, have higher habitat diversity and are less fragmented compared to other parts of the urban landscape.

Plant species richness was the only biodiversity dimension measured that decreased with increasing urban wildness. This might be attributed to the fact that estimated wildness tended to be particularly high in woodland (mostly forests close to the city centre). Woodland in general provides shady understory conditions leading to a smaller number of plant individuals thriving there and therefore a smaller species number in the relatively small sample sites compared to other habitat types. Additionally, many forest understory herbs are poor dispersers (Hermy, Honnay, Firbank, Grashof-Bokdam, & Lawesson, 1999) and therefore are rarely able to colonize recently established urban woodlands (Honnay et al., 2002), leading to rather species-poor herbaceous understory vegetation. On the other hand, the NMDS revealed plant species composition of woodland habitats varying considerably from those of the other habitat types, and the larger ordiellipse suggests high internal beta diversity. Therefore, plant species richness will be enhanced by woodland patches at the larger urban landscape-scale.

All these findings together suggest that there is a positive relationship between urban wildness and biodiversity. However, the bioscore was originally designed to measure perceived biodiversity by human inhabitants and is likely not able to measure true species richness of all relevant biodiversity dimensions accurately (Hand et al., 2016). Perceived and real biodiversity are likely to correlate though, and, furthermore, perceived biodiversity may be even more important from a human perspective, as recent studies suggest that perceived biodiversity increases urban residents' well-being (Hedblom, Knez, & Gunnarsson, 2017; Shwartz, Turbé, Simon, & Julliard, 2014).

The habitat types woodland and semi-natural/ruderal areas are clearly associated with high bioscores as well as high urban wildness values. An increase of these habitat types within the city landscape is therefore likely to also increase wildness and biodiversity experiences for urban inhabitants. Since other studies have shown the importance of these habitat types in providing ecosystem services such as improving climatic and air hygienic conditions in urban areas (Burkhardt et al., 2008), it is likely that expanding these habitat types will lead to synergies between increased wildness and biodiversity experiences and other factors influencing human well-being in cities. Furthermore, recent results show that there might be a direct link between urban wildness and better ecosystem functioning (Corlett, 2016b; Palta, Grimm, & Groffman, 2017).

Interestingly, artificial vegetated areas such as parks or gardens are

Table A1
Comparison of the differences in wildness values between the original wildness map and four alternative wildness maps where one indicator was excluded from the calculation respectively.

Indicator excluded	Mean difference in wildness values	Max. difference in wildness values	Correlation coefficient R
Visibility of built artefacts	13	88	0.89
Challenging terrain	1	83	0.99
Naturalness of land cover	15	88	0.74
Remoteness	34	68	0.96

the only habitat type were bioscores significantly increased with increasing urban wildness values. This has important management implications, as it shows that reduced management of gardens and parkland offers one among several possibilities for enhancing urban biodiversity as has been shown e.g. for native Mediterranean bird species (Shwartz, Shirley, & Kark, 2008) or carabid beetles (Venn & Kotze, 2014). Importantly, this is the one of the four urban habitat types that is most utilized by people, and hence offers strong scope for enhancing urban inhabitants' wildness and biodiversity experiences. On the other hand, perception studies often conclude that well-managed greenspaces are regularly preferred by urban inhabitants. Urban wildness areas therefore would also need to be carefully designed, e.g. with so called 'cues to care' that show the perceived 'mess' as actually being intended by the designers and not a by-product of neglect (Botzat, Fischer, & Kowarik, 2016; Weber, Kowarik, & Säumel, 2014).

5. Conclusion

We show that GIS-based relative wildness mapping is useful for pinpointing areas that are also relatively wild areas on the ground in an urban context. Urban wildness was furthermore linked to higher bioscores and hence to perceived biodiversity, indicating that inhabitants would benefit not only from the wildness experience, but also an enhanced experience of biodiversity. It is also likely that at least some actual biodiversity dimensions will benefit from enhanced urban wildness directly. Urban wildness areas could therefore provide a valuable component of city greenspaces that can be managed and maintained at low costs. Producing urban wildness maps is one possibility to visualize existing urban wildness and thus maybe facilitate an awareness and higher valuation for this kind of nature, and its better integration into urban planning.

Acknowledgments

A. Müller acknowledges the ERASMUS Program and the German National Academic Foundation for funding her stay at Aarhus University. This work is a contribution to the Danish National Research Foundation Niels Bohr professorship project Aarhus University Research on the Anthropocene (AURA) and the VILLUM Investigator project "Biodiversity Dynamics in a Changing World" (BIOCHANGE, grant 16549) funded by VILLUM FONDEN. We thank Meredith Root-Bernstein for proofreading the manuscript.

Appendix A. Analysis of mapping robustness

To examine the robustness of the GIS-based relative wildness mapping we assessed the sensitivity of the final wildness map regarding the error and uncertainty related to the four input indicator datasets. For the city center of Aarhus, we applied bootstrapping to add random "noise" to the input dataset as was previously suggested to measure model input errors for GIS-based wildness mapping (Carver et al., 2013). For this, each indicator map was multiplied with a random value between 0.9 and 1.1 to induce small changes in the original pixel values before standardizing and summarizing the indicator maps again to create an alternative wildness map. This process was repeated one hundred times and the standard deviation of the resulting one hundred wildness maps was calculated to depict overall sensitivity to input indicator values. The resulting maximum standard deviation of 5.9 (2% variability regarding the 0–255 value ranges) and the mean standard deviation of 2.8 (1% variability with regard to the 0–255 value range) show that our mapping produced reasonably robust results within the city center of Aarhus, with major roads and certain parts of large greenspaces being least sensitive to a change in indicators (Fig. S4, Supplementary material). We furthermore investigated the effect that excluding one input indicator map from the wildness mapping would have on the final results for the whole municipality of Aarhus. For this we calculated four additional wildness maps where we excluded one of the indicator maps, respectively. We calculated the correlation coefficients of the four resulting wildness maps when compared to the original wildness map by linear regression (Table A1). Compared to the originally derived wildness map, the map without remoteness shows the highest mean difference in wildness values and thus remoteness (from noise and mechanized access) seems to be the indicator influencing the final wildness mapping results the most. Visibility of built artefacts and naturalness of land cover do show similar, rather modest mean differences in wildness values compared to the original wildness map and challenging terrain obviously has the lowest influence on the wildness mapping results. All correlation coefficients showed values > 0.7 indicating a strong uphill relationship. The general trend in pixel values obviously stays the same, even when one indicator input is left out of the calculation.

Appendix B. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.landurbplan.2017.09.027.

References

Aarhus Kommune (2017). *Welcome to Aarhus.* (n.d.) Retrieved from https://www.aarhus.dk/da/omkommunen/English.aspx.
Angold, P. G., Sadler, J. P., Hill, M. O., Pullin, A., Rushton, S., Austin, K., ... Thompson, K. (2006). Biodiversity in urban habitat patches. *Science of The Total Environment, 360*(1–3), 196–204. http://dx.doi.org/10.1016/j.scitotenv.2005.08.035.
Bolund, P., & Hunhammar, S. (1999). Ecosystem services in urban areas. *Ecological Economics, 29*(2), 293–301. http://dx.doi.org/10.1016/S0921-8009(99)00013-0.
Bonthoux, S., Brun, M., Di Pietro, F., Greulich, S., & Bouché-Pillon, S. (2014). How can wastelands promote biodiversity in cities? A review. *Landscape and Urban Planning, 132*, 79–88. http://dx.doi.org/10.1016/j.landurbplan.2014.08.010.
Botzat, A., Fischer, L. K., & Kowarik, I. (2016). Unexploited opportunities in

understanding liveable and biodiverse cities. A review on urban biodiversity perception and valuation. *Global Environmental Change, 39*, 220–233. http://dx.doi.org/10.1016/j.gloenvcha.2016.04.008.
Buck, H. J. (2015). On the possibilities of a charming Anthropocene. *Annals of the Association of American Geographers, 105*(2), 369–377. http://dx.doi.org/10.1080/00045608.2014.973005.
Burkhardt, I., Dietrich, R., Hoffmann, H., Lechner, J., Lohmann, K., Schoder, F., & Schultz, A. (2008). Urbane Wälder. Abschlussbericht zur Voruntersuchung für das E + E—Vorhaben, Ökologische Stadterneuerung durch Anlage urbaner Waldflächen auf innerstädtischen Flächen im Nutzungswandel—ein Beitrag zur Stadtentwicklung. *Naturschutz Und Biologische Vielfalt,* 63.
COWI (2014). *Danmarks digitale ortofoto 2014*. Retrieved from: http://www.cowi.dk/menu/service/geografiskinformationogit/kortoggeodataprodukter/ortofotos/ddo2014/.
Carver, S., Comber, A., McMorran, R., & Nutter, S. (2012). A GIS model for mapping spatial patterns and distribution of wild land in Scotland. *Landscape and Urban Planning, 104*(3–4), 395–409. http://dx.doi.org/10.1016/j.landurbplan.2011.11.016.
Carver, S., Tricker, J., & Landres, P. (2013). Keeping it wild: Mapping wilderness character in the United States. *Journal of Environmental Management, 131*, 239–255. http://dx.doi.org/10.1016/j.jenvman.2013.08.046.
Cercle Bruit Schweiz (1998). *Lärm (Noise)*. Retrieved from http://www.laerm.ch/dokumente/publikationen/laerm.pdf.
Chen, M., Zhang, H., Liu, W., & Zhang, W. (2014). The global pattern of urbanization and economic growth: Evidence from the last three decades. *Public Library Of Science, 9*(8), e103799. http://dx.doi.org/10.1371/journal.pone.0103799.
Coops, N. C., Fontana, F. M. A., Harvey, G. K. A., Nelson, T. A., & Wulder, M. A. (2014). Monitoring of a national-scale indirect indicator of biodiversity using a long time-series of remotely sensed imagery. *Canadian Journal of Remote Sensing, 40*(3), 179–191. http://dx.doi.org/10.1080/07038992.2014.945826.
Corlett, R. T. (2016a). Restoration, reintroduction, and rewilding in a changing world. *Trends in Ecology & Evolution, 31*(6), 453–462. http://dx.doi.org/10.1016/j.tree.2016.02.017.
Corlett, R. T. (2016b). The role of rewilding in landscape design for conservation. *Current Landscape Ecology Reports, 1*(3), 127–133. http://dx.doi.org/10.1007/s40823-016-0014-9.
Cox, D. T. C., Shanahan, D. F., Hudson, H. L., Plummer, K. E., Siriwardena, G. M., Fuller, R. A., ... Gaston, K. J. (2017). Doses of neighborhood nature: The benefits for mental health of living with nature. *Bioscience, 67*(2), 147–155. http://dx.doi.org/10.1093/biosci/biw173.
Dallimer, M., Rouquette, J. R., Skinner, A. M. J., Armsworth, P. R., Maltby, L. M., Warren, P. H., & Gaston, K. J. (2012). Contrasting patterns in species richness of birds, butterflies and plants along riparian corridors in an urban landscape. *Diversity and Distributions, 18*(8), 742–753. http://dx.doi.org/10.1111/j.1472-4642.2012.00891.x.
Danish Agrifish Agency (2015). *Marker 2015, data on Danish agricultural land use.* Retrieved from: https://kortdata.fvm.dk/download/Markblokke_Marker?page=MarkerHistoriske.
Danish Environmental Agency (2012). *Strategic noise maps from Denmark for major roads, railways, airports and agglomerations.* Retrieved from: http://cdr.eionet.europa.eu/dk/eu/noise/df8/envuklbfw/.
Danish Geodata Agency (2007a). *The Danish national height model, Terrain, DK-DTM.* Retrieved from: http://download.kortforsyningen.dk/content/dhm-2007terr%C3%A6n-16-m-grid.
Danish Geodata Agency (2007b). *The Danish national height model, surface, DK-DSM.* Retrieved from: http://download.kortforsyningen.dk/content/dhm-2007overflade-16-m-grid.
Danish Geodata Agency (2013). *Kort 10.* Retrieved from: http://download.kortforsyningen.dk/content/kort10.
Danish Natural Environment Portal (2007). *Beskyttede naturtyper (protected nature types).* Retrieved from: http://arealinformation.miljoeportal.dk/distribution.
Del Tredici, P. (2014). The flora of the future. *Places Journal*.
Environmental Systems Research Institute (ESRI) (2017). *ArcGIS for desktop 10.3*.
European Commission (2013). *Guidelines on wilderness in Natura 2000-management of terrestrial wilderness and wild areas within the Natura 2000 network.* Retrieved from http://ec.europa.eu/environment/nature/natura2000/wilderness/pdf/WildernessGuidelines.pdf.
European Parliament (2009). *European parliament resolution of 3 february 2009 on wilderness in Europe.* (P6_TA(2009)0034). Strasbourg Retrieved from http://www.europarl.europa.eu/sides/getDoc.do?pubRef=-//EP//TEXT+TA+P6-TA-2009-0034+0+DOC+XML+V0//EN.
European Union (2011). *Cities of tomorrow—Challenges, visions, ways forward.* Retrieved from http://ec.europa.eu/regional_policy/sources/docgener/studies/pdf/citiesoftomorrow/citiesoftomorrow_final.pdf.
Feizizadeh, B., & Blaschke, T. (2014). An uncertainty and sensitivity analysis approach for GIS-based multicriteria landslide susceptibility mapping. *International Journal of Geographical Information Science, 28*(3), 610–638. http://dx.doi.org/10.1080/13658816.2013.869821.
Hand, K. L., Freeman, C., Seddon, P. J., Stein, A., & van Heezik, Y. (2016). A novel method for fine-scale biodiversity assessment and prediction across diverse urban landscapes reveals social deprivation-related inequalities in private, not public spaces. *Landscape and Urban Planning, 151*, 33–44. http://dx.doi.org/10.1016/j.landurbplan.2016.03.002.
Hedblom, M., Knez, I., & Gunnarsson, B. (2017). Bird diversity improves the well-being of city residents. In E. Murgui, & M. Hedblom (Eds.). *Ecology and conservation of birds in urban environments* (pp. 287–306). Cham: Springer International Publishing.
Hermy, M., Honnay, O., Firbank, L., Grashof-Bokdam, C., & Lawesson, J. E. (1999). An ecological comparison between ancient and other forest plant species of Europe, and the implications for forest conservation. *Biological Conservation, 91*(1), 9–22. http://dx.doi.org/10.1016/S0006-3207(99)00045-2.
Hobbs, R. J., Higgs, E., & Harris, J. A. (2009). Novel ecosystems: Implications for conservation and restoration. *Trends in Ecology & Evolution, 24*(11), 599–605. http://dx.doi.org/10.1016/j.tree.2009.05.012.
Honnay, O., Bossuyt, B., Verheyen, K., Butaye, J., Jacquemyn, H., & Hermy, M. (2002). Ecological perspectives for the restoration of plant communities in European temperate forests. *Biodiversity & Conservation, 11*(2), 213–242. http://dx.doi.org/10.1023/a:1014531011060.
Jepson, P. (2016). A rewilding agenda for Europe: Creating a network of experimental reserves. *Ecography, 39*(2), http://dx.doi.org/10.1111/ecog.01602 n/a-n/a.
Jorgensen, A., & Keenan, R. (2012). *Urban wildscapes.* Taylor & Francis.
Jorgensen, A., & Tylecote, M. (2007). Ambivalent landscapes—Wilderness in the urban interstices. *Landscape Research, 32*(4), 443–462. http://dx.doi.org/10.1080/01426390701449802.
Levin, G., Jepsen, M. R., & Blemmer, M. (2012). *Basemap, technical documentation of a model for elaboration of a land-use and land-cover map for Denmark*. (47 pp. Technical Report from DCE ?Danish Centre for Environment and Energy No. 11). Retrieved from www2. dmu.dk/Pub/TR11.pdf.
Mölder, A., Bernhardt-Römermann, M., & Schmidt, W. (2008). Herb-layer diversity in deciduous forests: Raised by tree richness or beaten by beech? *Forest Ecology and Management, 256*(3), 272–281. http://dx.doi.org/10.1016/j.foreco.2008.04.012.
Müller-Dombois, D., & Ellenberg, H. (1974). *Aims and methods of vegetation ecology.* New York: John Wiley and Sons.
Müller, A., Bøcher, P. K., & Svenning, J.-C. (2015). Where are the wilder parts of anthropogenic landscapes? A mapping case study for Denmark. *Landscape and Urban Planning, 144*, 90–102.
Navarro, L. M., & Pereira, H. M. (2012). Rewilding abandoned landscapes in Europe. *Ecosystems, 15*(6), 900–912. http://dx.doi.org/10.1007/s10021-012-9558-7.
Oksanen, J., Blanchet, F. G., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., ... Wagner, H. (2016). *Vegan: Community ecology package (Version 2.4-1).* Retrieved from https://CRAN.R-project.org/package=vegan.
Paker, Y., Yom-Tov, Y., Alon-Mozes, T., & Barnea, A. (2014). The effect of plant richness and urban garden structure on bird species richness, diversity and community structure. *Landscape and Urban Planning, 122*, 186–195. http://dx.doi.org/10.1016/j.landurbplan.2013.10.005.
Palta, M. M., Grimm, N. B., & Groffman, P. M. (2017). “Accidental” urban wetlands: Ecosystem functions in unexpected places. *Frontiers in Ecology and the Environment, 15*(5), 248–256. http://dx.doi.org/10.1002/fee.1494.
Pinheiro, J., Bates, D., DebRoy, S., Sarkar, D., & R Core Team (2016). *nlme: Linear and nonlinear mixed effects models (Version 3.1-128).* Retrieved from http://CRAN.R-project.org/package=nlme.
R Core Team (2015). *R: A language and environment for statistical computing: R foundation for statistical computing.* Retrieved from https://www.R-project.org/.
Rangel, T. F., Diniz-Filho, J. A. F., & Bini, L. M. (2010). SAM: A comprehensive application for spatial analysis in macroecology. *Ecography, 33*(1), 46–50. http://dx.doi.org/10.1111/j.1600-0587.2009.06299.x.
Rupprecht, C. D. D., & Byrne, J. A. (2014). Informal urban greenspace: A typology and trilingual systematic review of its role for urban residents and trends in the literature. *Urban Forestry & Urban Greening, 13*(4), 597–611. http://dx.doi.org/10.1016/j.ufug.2014.09.002.
Scottish Natural Heritage (2014). *Mapping of Scotland’s wildness and wild land: Non-technical description of the methodology.* Retrieved from http://www.snh.gov.uk/docs/A1342460.pdf.
Shwartz, A., Shirley, S., & Kark, S. (2008). How do habitat variability and management regime shape the spatial heterogeneity of birds within a large Mediterranean urban park? *Landscape and Urban Planning, 84*(3–4), 219–229. http://dx.doi.org/10.1016/j.landurbplan.2007.08.003.
Shwartz, A., Turbé, A., Simon, L., & Julliard, R. (2014). Enhancing urban biodiversity and its influence on city-dwellers: An experiment. *Biological Conservation, 171*, 82–90. http://dx.doi.org/10.1016/j.biocon.2014.01.009.
Skov, F., & Svenning, J.-C. (2003). Predicting plant species richness in a managed forest. *Forest Ecology and Management, 180*(1), 583–593. http://dx.doi.org/10.1016/S0378-1127(02)00646-1.
Statistics Denmark (2016a). *Arealdække (area cover)*. Retrieved 11.08.2016 http://www.statistikbanken.dk/statbank5a/selectvarval/saveselections.asp.
Statistics Denmark (2016b). *Befolkningen 2016 (population 2016)*. Retrieved 11.08.2016 http://www.statistikbanken.dk/BEF44.
Sukopp, H. (2008). On the early history of urban ecology in Europe. In J. M. Marzluff, E. Shulenberger, W. Endlicher, M. Alberti, G. Bradley, C. Ryan, U. Simon, & C. ZumBrunnen (Eds.). *Urban ecology: An international perspective on the interaction between humans and nature* (pp. 79–97). Boston, MA: Springer US.
Svenning, J.-C., Pedersen, P. B. M., Donlan, C. J., Ejrnæs, R., Faurby, S., Galetti, M., ... Vera, F. W. M. (2016). Science for a wilder Anthropocene: Synthesis and future directions for trophic rewilding research. *Proceedings of the National Academy of Sciences of the United States of America, 113*(4), 898–906. http://dx.doi.org/10.1073/pnas.1502556112.
Threlfall, C. G., Williams, N. S. G., Hahs, A. K., & Livesley, S. J. (2016). Approaches to urban vegetation management and the impacts on urban bird and bat assemblages. *Landscape and Urban Planning, 153*, 28–39. http://dx.doi.org/10.1016/j.landurbplan.2016.04.011.
Venn, S., & Kotze, D. (2014). Benign neglect enhances urban habitat heterogeneity: Responses of vegetation and carabid beetles (Coleoptera: Carabidae) to the cessation of mowing of park lawns. *European Journal of Endocrinology, 111*(5), 703–714.
Warren, P. S., Harlan, S. L., Boone, C., Lerman, S., Shochat, E., & Kinzig, A. (2010). *Urban ecology and human social organization urban ecology.* Cambridge: Cambridge University Press172–201.
Weber, F., Kowarik, I., & Säumel, I. (2014). A walk on the wild side: Perceptions of roadside vegetation beyond trees. *Urban Forestry & Urban Greening, 13*(2), 205–212. http://dx.doi.org/10.1016/j.ufug.2013.10.010.
Zheng, B., & Agresti, A. (2000). Summarizing the predictive power of a generalized linear model. *Statistics in Medicine, 19*(13), 1771–1781. http://dx.doi.org/10.1002/1097-0258(20000715)19:13<1771:AID-SIM485>3.0.CO;2-P.