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A spatial framework for targeting urban planning for pollinators and people with local stakeholders: A route to healthy, blossoming communities?

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A R T I C L E I N F O

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A B S T R A C T

Pollinators such as bees and hoverflies are essential components of an urban ecosystem, supporting and contributing to the biodiversity, functioning, resilience and visual amenity of green infrastructure. Their urban habitats also deliver health and well-being benefits to society, by providing important opportunities for accessing nature nearby to the homes of a growing majority of people living in towns and cities. However, many pollinator species are in decline, and the loss, degradation and fragmentation of natural habitats are some of the key drivers of this change. Urban planners and other practitioners need evidence to carefully prioritise where they focus their resources to provide and maintain a high quality, multifunctional green infrastructure network that supports pollinators and people. We provide a modelling framework to inform green infrastructure planning as a nature based solution with social and ecological benefits. We show how habitat suitability models (HSM) incorporating remote sensed vegetation data can provide important information on the influence of urban landcover composition and spatial configuration on species distributions across cities. Using Edinburgh, Scotland, as a case study city, we demonstrate this approach for bumble bees and hoverflies, providing high resolution predictive maps that identify pollinator habitat hotspots and pinch points across the city. By combining this spatial HSM output with health deprivation data, we highlight 'win-win' opportunity areas in most need of improved green infrastructure to support pollinator habitat quality and connectivity, as well as societal health and well-being. In addition, in collaboration with municipal planners, local stakeholders, and partners from a local greenspace learning alliance, we identified opportunities for citizen engagement activities to encourage interest in wildlife gardening as part of a 'pollinator pledge'. We conclude that this quantitative, spatially explicit and transferable approach provides a useful decision-making tool for targeting nature-based solutions to improve biodiversity and increase environmental stewardship, with the aim of providing a more attractive city to live, work and invest in.

1. Introduction

Bees and hoverflies are vital components of urban ecosystems; they support the functioning and resilience of typically ecologically fragile and fragmented areas of urban greenspace by contributing to pollination, biodiversity and pest control (Fontaine et al., 2006; Hall et al., 2016). Their interaction with flowers results in a wide range of direct and indirect benefits to people in cities, most obviously by supporting urban agriculture via pollination, but pollinator insects and their habitats also provide cultural and health-related benefits to society by presenting opportunities to interact with nature (Bates et al., 2011; Maller et al., 2006). Grasslands that are infrequently mowed and developed into urban meadows with a diverse wildflower mix provide important pollinator habitats; they also tend to be regarded as more visually attractive than traditional amenity grassland (Blaauw and Isaacs, 2014; Garbuzov et al., 2015; Hicks et al., 2016; Hülsmann et al., 2015; Southon et al., 2017). Pollinators are also good indicators of urban biodiversity (Blair, 1999; Paoletti, 2012) and people have been found to state a preference for, or self-report more psychological benefits from, areas with higher levels of biodiversity (Fuller et al., 2007;

Abbreviations: CIR, Colour infra-red; DTM, Digital Terrain Model; DSM, Digital Surface Model; ELL, Edinburgh Living Landscape; HSI, Habitat Suitability Index; HSM, Habitat Suitability Model; LERC, Local Environmental Record Centre; MTSS, Maximum Training Sensitivity and Specificity; NDVI, Normalized Difference Vegetation Index; NGO, Non-Governmental Organisation; OBIA, Object Based Image Analysis; SIMD, Scottish Index of Multiple Deprivation; TG, Target Group

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Lindemann-Matthies et al., 2010; Shwartz et al., 2014; Carrus et al., 2015; although see Qiu et al., 2013). Pollinator-friendly, species rich urban areas are therefore likely to provide a 'biodiversity feel good factor' (Dallimer et al., 2012), contributing to the health and well-being of people living and visiting towns and cities (although further evidence and careful analysis is required to 'unpack the people-biodiversity paradox' (Pett et al., 2016)).

Actions to support pollinators and their habitats are increasingly important as evidence from field studies (largely in Europe and North America) has highlighted a decline in pollinator species diversity and the contraction in many species' range (Potts et al., 2010). These are likely to be caused by multiple, inter-related factors, including the loss, degradation and fragmentation of habitats; use of agrochemicals; spread of pathogens; a changing climate (Blaauw and Isaacs, 2014; Garibaldi et al., 2011; Gill et al., 2016; Potts et al., 2010; Ricketts et al., 2008; Vanbergen, 2013). One study reported a 76% reduction in the frequency of bumble bee forage plant presence across the UK between 1978 and 1998, exhibiting a greater magnitude of change than other plant types (Carvell et al., 2006). At landscape scales, the reduction and fragmentation of semi-natural habitats as a result of urbanisation threatens pollinators by decreasing the amount and quality of foraging and nesting resources (Bates et al., 2011; Harrison and Winfree, 2015; Hernandez et al., 2009). However species-, taxon- and context-specific effects of urbanisation have been found and some studies have reported higher bee abundance or activity in urban study sites compared to farmland sites (Baldock et al., 2015a; Kaluza et al., 2016), and the landscape heterogeneity introduced by moderate levels of urbanisation may have a positive influence on some pollinators (Theodorou et al., 2016). Within urban areas, pollinator abundance and diversity is often highly varied as a result of the patchy distribution of these resources, with some greenspace sites rich in nesting and floral resources acting as key islands of habitat (Hülsmann et al., 2015; Kaluza et al., 2016; Theodorou et al., 2016).

As urban areas are expanding in Europe and globally, and pressures on ecosystems associated with increasing human populations and rates of consumption continue to rise (Shaker, 2015), national pollinator strategies (e.g. Defra, 2014) should be implemented that duly acknowledge and act on the importance of urban greenspace for pollinators (Baldock et al., 2015b; Hall et al., 2016). At a city scale, municipal planners should also enhance networks of high quality greenspace in support of pollinators, by targeting their often-limited resources in areas where they are most needed. These strategies can deliver 'win-wins' by also improving delivery of a range of ecosystem services, including opportunities to access nature, to the growing majority of people residing in urban areas (almost three quarters of people were reported to live in urban areas in the European Union (EU) in 2015 (EU, 2016, p.8)). However, trade-offs exist between biodiversity or species conservation and the demand for other land uses, functions and ecosystem services at a range of scales (Eigenbrod et al., 2009; Maes et al., 2012); for example, some greenspace users prefer short, 'neat' amenity grassland for recreation, which tends to be of low biodiversity value. Conservationists, local authorities, spatial planners and developers therefore need to work together to carefully plan, design and manage a biodiverse, multifunctional greenspace network. An evidence base is required to ensure that green infrastructure planning is taking measurable steps to improving urban areas for biodiversity, rather than acting as a 'tick box exercise' or an 'ecological trap' that meets policy obligations on paper, but divests funds away from more effective measures for biodiversity in reality (Garmendia et al., 2016). Action 5 of the 'EU Biodiversity Strategy to 2020' requires member states to 'Map and Assess the state of Ecosystems and their Services' (MAES), including biodiversity (Maes et al., 2016). However, the detailed information needed for this type of mapping exercise is often unavailable, particularly at a city-scale (Sandström et al., 2006). Practitioners would therefore benefit from relevant data and decision making tools that help them to assess the impact of plans on ecosystem services (e.g. De Ridder et al., 2004; Hansen and Pauleit, 2014; Łopucki and Kiersztyn, 2015; Vujić et al., 2016). For example, guidance from the MAES urban pilot project recommends using measures of pollinator abundance and the 'capacity for ecosystems to sustain insect pollination' as indicators for mapping insect pollination services (Maes et al., 2016, p. 79).

This study provides a modelling framework for informing strategic urban green infrastructure planning as a 'nature-based solution'. Nature-based solutions are 'actions which are inspired by, supported by or copied from nature …[that] aim to help societies address a variety of environmental, social and economic challenges in sustainable ways' (European Commission, 2015, p. 5). To meet these goals, our approach determines *what* actions should be taken to provide or improve a city's green infrastructure, and *where* to implement these actions across a city to maximise the benefits they provide to both pollinators and people. This adaptable framework can be applied to any taxonomic group or urban area of interest for which adequate data are available.

We use habitat suitability modelling (HSM; also commonly referred to as species distribution models (Elith and Leathwick, 2009; Franklin, 2009; Guisan and Zimmermann, 2000) to provide this evidence for two groups of wild insect pollinators (bumble bees and hoverflies). HSM typically involves identifying relationships between a species' known distribution, often obtained from organised surveys or existing records, and environmental factors over space. These models deliver spatially explicit, quantitative predictions of habitat suitability at a landscape scale, from which species distributions can be inferred. Understanding what urban features influence the distribution of particular taxa is key to informing greenspace strategies aimed at improving and protecting their habitats (Cox et al., 2016); multiscale HSM highlight important environmental correlates of a species' presence at a range of spatial scales, offering valuable insights into the species' habitat requirements and ecology (e.g. Bellamy et al., 2013). They can also be used to model the impact of potential changes, providing a useful tool for quantitatively appraising the biodiversity impact of alternative greenspace or housing development scenarios (Mortberg et al., 2007).

To identify areas where local people would benefit from improved green infrastructure, we focus on areas of high health deprivation using the health index of Scotland's Index of Multiple Deprivation data (The Scottish Government, 2012). It is now largely accepted that access to high quality greenspace and opportunities to interact with nature close to people's homes has a positive impact on health and well-being (Cox et al., 2017a, 2017b; Gascon et al., 2015; Hartig et al., 2014; Keniger et al., 2013; Maller et al., 2006). However, we know from the literature that residents of socioeconomically deprived areas tend to experience lower quality environmental conditions, including poor access to greenspace (Pearce et al., 2010). This is reflected in self-reports of greenspace satisfaction, which are most negative in the 15% most deprived areas in Scotland (TNS, 2014). Moreover, this report also showed that those people in the most deprived group are least likely to visit their local greenspace. This is problematic as greater greenspace exposure buffers the negative effect of income inequality on health outcomes (Mitchell and Popham, 2008). Research has also shown that environmental interventions in deprived areas improve frequency of use and environmental perceptions by local people (Ward Thompson et al., 2013). Therefore, these areas would ideally be prioritised for actions to provide pollinator-friendly, visually attractive green infrastructure as these could act as nature-based solutions to health deprivation. By overlaying these health target zones with priority pollinator improvement areas identified by the HSM, we highlight win-win opportunities where these enhancements would have both social and ecological benefits.

2. Materials and methods

2.1. Edinburgh case study

This work was done as part of GREEN SURGE, a collaborative EU research project providing practitioners with knowledge and tools relevant to urban green infrastructure (Konijnendijk van den Bosch, 2013). We used Scotland's capital city, Edinburgh, as a case study city to trial our methods. In common with many European cities, Edinburgh is under pressure to meet the rise in demand for affordable housing whilst supporting the health and well-being of Edinburgh residents by providing high quality greenspace within and around the city. Scotland's Proposed Third National Planning Framework pledges that 'quality of life and resilience in city regions will be supported by green infrastructure', (The Scottish Government, 2014). The City of Edinburgh Council plans to build 19,500 houses by 2022, focussing on vacant and derelict land and some greenbelt in four 'Strategic Development Zones' (City of Edinburgh Council, 2015a). This puts pressure on the publically owned urban greenspaces within and at the periphery of the city, which were estimated to account for around 27% of the 11,000 ha of 'urban' Edinburgh in a 2005 Open Space Strategy and audit (City of Edinburgh Council, 2015b).

Given the need to involve stakeholders operating at different spatial and hierarchical scales to successfully deliver nature-based solutions such as urban green infrastructure, especially if also including private green spaces (Gaston et al., 2005; Goddard et al., 2010; Smith et al., 2006), there has been a continuous exchange of science-driven and local knowledge using GREEN SURGE 'learning alliances' (Smith et al., 2015). Engaging with local actors enables collaborative learning and the development of 'hybrid knowledge' as a result of integrating local (formal and informal) and scientific knowledge (Raymond et al., 2010). This hybrid knowledge has the potential to inform innovative solutions to socio-environmental challenges (Olsson et al., 2004) and the social gains from the engagement process itself can increase the perceived likelihood of the success of biodiversity outcomes (Young et al., 2013).

In Edinburgh, the local greenspace learning alliance is the Edinburgh Living Landscape (ELL) project. The aim of this collaborative group of researchers, practitioners and planners is to further the 'creation and restoration of robust, resilient and connected green (and blue) infrastructure' (Scottish Wildlife Trust, 2014, p. 2) in and around the city. We used an ELL-led workshop to engage stakeholders and to get feedback on the potential usefulness and limitations of our approach for providing an evidence base for targeting green infrastructure resources in Edinburgh. We focussed discussions on the development and implementation of a 'pollinator pledge', an ongoing ELL-led initiative that aims to encourage strategic improvements to Edinburgh's green infrastructure for pollinators, by increasing environmental stewardship and citizen interest in urban pollinators.

2.2. The methodological framework

To identify *what* actions should be taken to provide or improve green infrastructure, and *where* to implement these actions across a city to maximise the benefits they provide to both people and pollinators, we carried out the following steps (each of which are explained in further detail in the methods sections below):

1) Mapping urban landscape features: we used landcover and remote sensed data to provide detailed, citywide spatial information on Edinburgh's greenspaces and built infrastructure features that may influence pollinator habitat suitability
2) Modelling pollinator-urban feature interactions: using a multiscale, presence-only HSM, we tested the strength and direction of the influence of these features on bumble bee and hoverfly habitat suitability at different spatial scales (100 m – 2 km) to identify actions that can be taken to improve greenspace quality
3) Mapping target areas for improving pollinator habitat suitability and connectivity: we developed and validated an optimal HSM. The resulting habitat suitability indices (HSI) were projected across Edinburgh at a 100 m resolution to highlight important hotspots and target areas where actions should be focussed to improve greenspace quality and connectivity for pollinators
4) Mapping target areas for improving people's health and identifying 'win-wins': we identified areas of relatively poor human health and overlaid these onto the pollinator target areas to locate parts of the city where improvements to green infrastructure should be targeted as a nature-based solution with ecological and societal benefits
5) Engaging stakeholders: we presented our outputs to local stakeholders and identified opportunities for citizen engagement to encourage interest in wildlife gardening and increase environmental stewardship within and around mapped target zones.

2.3. Mapping urban landscape features

We obtained the best available spatial landcover and land use datasets for Edinburgh's administrative area boundary and a surrounding 2 km buffer (Supplementary Material, Table SM 1). Although pollinator distributions are likely to be influenced by very fine scale habitat features, such the availability of wildflowers (Hicks et al., 2016), information on vegetation structures and floral resources mapped at a citywide scale are a common data gap. We therefore took a step towards filling this gap, using an Object Based Image Analysis (OBIA) rule set in Definiens eCognition (v8, http://www.ecognition.com/) and remote sensed data to map the areas most likely to be covered by vegetation, and extracted areas of expected tree canopy from this vegetation layer (see Supplementary Material Methods for further details). The datasets collected were manipulated and combined in ArcGIS (v. 10.2; www.esri.com) to extract maps of green (e.g. tree canopies) and 'grey' (e.g. roads) infrastructure features that we identified as potential predictors of pollinator habitat suitability based on expert knowledge and the literature (Table 1). Some of these datasets were also combined to develop a 'best available' urban land use-land cover map for Edinburgh, which was mapped to our most up to date, highest resolution map

Table 1
Details of the candidate environmental predictor variables used in this study.

Category	Metric type	Measurement radii (m)
Green infrastructure	Distance to green corridors (m)	N/A
	Distance to inland water course (m)	N/A
	Diversity of green infrastructure type	100, 500, 1000, 2000
	Percentage cover of allotments	100, 500, 1000, 2000
	Percentage cover of heathland	100, 500, 1000, 2000
	Percentage cover of improved grass	100, 500, 1000, 2000
	Percentage cover of inland water	100, 500, 1000, 2000
	Percentage cover of other grassland	100, 500, 1000, 2000
	Percentage cover of parks	100, 500, 1000, 2000
	Percentage cover of productive greenspace	100, 500, 1000, 2000
	Percentage cover of tree canopy cover	100, 500, 1000, 2000
	Percentage cover of vegetated domestic gardens	100, 500, 1000, 2000
	Percentage cover of vegetation (including trees)	100, 500, 1000, 2000
	Percentage cover of domestic gardens	100, 500, 1000, 2000
	Percentage cover of vegetated domestic gardens	100, 500, 1000, 2000
	Urban land use/land cover type	N/A
Grey infra-structure	Distance to paths (m)	N/A
	Distance to roads (m)	N/A
	Percentage cover of buildings	100, 500, 1000, 2000
	Road density of (m/m3)	100, 500, 1000, 2000
Terrain	Aspect	N/A
	Mean altitude (m)	100, 500, 1000, 2000
	Mean slope (°)	100, 500, 1000, 2000

(Ordnance Survey MasterMap Topography Layer). This work was carried out as part of an Innovate-UK co-funded project led by eCountability Ltd to develop a decision support tool ('SPADES™'; SPAtial Decisions on Ecosystem Services'). For example, the OBIA-derived vegetation layer was used to subdivide domestic gardens into vegetated and non-vegetated areas. Green corridors were identified as all water courses and areas of vegetation (including trees) running alongside paths or roads. All of the urban landscape features extracted as potential predictors of pollinator habitat suitability were mapped at a 100 m resolution and many were measured at multiple extents using a moving window analysis (see Section 2.4.3 for more details).

2.4. Modelling pollinator-urban feature interactions

Presence-only habitat suitability models were created using the MaxEnt approach (Phillips et al., 2006). Details of the input data and an outline of the steps used to create and test the models are provided below.

2.4.1. Input data: species

We developed separate models for the two wild pollinator groups for which we could obtain sufficient high quality data, bumble bees (of the genus Bombus) and hoverflies (of the family Syrphidae). Records of insect presence were obtained from Edinburgh's Local Environmental Record Centre (LERC), The Wildlife Information Centre (TWIC; http://www.wildlifeinformation.co.uk/; Fig. 1). Only those presence records made within the City of Edinburgh Council's administrative areas during the summer months (April - September inclusive) between 2005 and 2015 were included in an effort to remove information on potentially suboptimal overwintering habitats, which resulted in 208 bumble bee and 83 hoverfly records. We then filtered records to ensure that they were made at a resolution of ≤ 100 m (six or more figure grid references). To remove duplicates and to reduce spatial dependence between records, which can cause overfitting and inflate measures of model performance (Boria et al., 2014; Veloz, 2009), records were spatially filtered so that they were at least 100 m apart. Our final sample size consisted of 107 bumble bee and 41 hoverfly records. Although these are not large sample sizes, MaxEnt has been shown to perform well with small numbers of presence records (e.g. ≤25, Pearson et al., 2007).

2.4.2. Input data: sampling bias

LERC data commonly suffers from sampling bias because of the *ad hoc* manner in which the records tend to be made; recorders tend to favour more easily accessible areas (Boakes et al., 2010; Graham et al., 2004). To account for this sampling bias, we followed the Target Group (TG) sampling approach that involves providing information on survey effort across the study area for the taxonomic group to which the modelled species belongs (Phillips et al., 2009; Phillips and Dudik, 2008). This approach has also been found to improve the performance of presence-only models built with these type of species data (Merow et al., 2013; Syfert et al., 2013). Records of all insect species (of the class Insecta) were used as the TG because they were presumed to have been collected using similar survey strategies and are therefore are likely to suffer from similar sampling bias. To reduce the impact of species richness influencing an area's estimated sampling effort, which can be an issue with the TG approach (Warton et al., 2013), the records were first filtered to remove multiple records at the same location on the same day so that the number of days that a location was sampled was used to measure sampling effort, rather than the total number of records at that location. This provided 1,946 insect records. The pollinator insect records were commonly recorded along Edinburgh's pathways (public rights of way, bridleways, cycle paths, and City of Edinburgh Council's local and core path network), with 86% of the records falling within 200 m of a path. We therefore supplemented our TG background with 8,000 records located at random within 200 m of a path (Supplementary Material Fig. SM1), to reach a number close to the 10,000 background locations recommended by Phillips and Dudik (2008) to maximise MaxEnt performance.

2.4.3. Input data: candidate predictor variables

We developed candidate predictor variables providing information on the composition, spatial configuration and structure of the built environment and green infrastructure features of the urban landscape within various sized (100 m – 2 km) radii around each 100 m cell (Table 1). We used the 'Multiscale Maxent Toolbox' to create these multiscale variables in ArcGIS (Bellamy et al., 2013; Bellamy and Altringham, 2015). The scales were chosen to reflect the typical movement capabilities of bumble bee and hoverfly species and previously reported environmental correlates of their presence or activity (Bates et al., 2011; Carvell et al., 2006; Hopfenmuller et al., 2014;

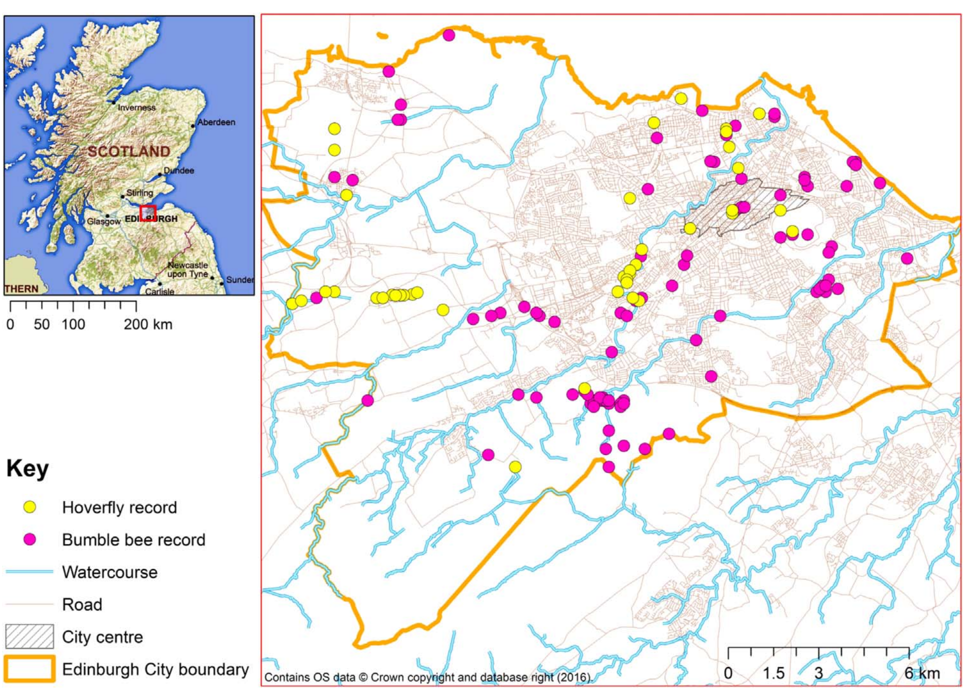

Fig. 1. *Distribution of the filtered species records used to create the models for the City of Edinburgh, Scotland.* Inset map shows the location of Edinburgh in Scotland. Species data were obtained from The Wildlife Information Centre, Edinburgh and are displayed at the minimum resolution allowed according to a data licence agreement.

Matteson and Langellotto, 2010). To prevent edge effects skewing variable measurements near the city boundary, data within a 2 km buffer of Edinburgh's boundary were used in the creation of these variables.

2.4.4. Creating, testing and fine-tuning the models

We developed a five-step habitat suitability modelling approach in ArcGIS and the open source analysis software, R (v. 3.3.3; https://www.r-project.org/):

Step 1. Spatially aggregation of species data

SDMToolbox (v 1.1c; http://sdmtoolbox.org/) was used to separate the bumble bee and hoverfly records into three spatially aggregated groups based on Voronoi polygons and the clustering of a species group's records across the study area (Brown, 2014). These spatial data folds were used for spatially constrained three-fold cross validation, retaining one data group at a time for model testing. Spatial aggregation of the data folds reduces the potential for false inflation of model performance measures caused by spatial autocorrelation between training and test records (Bellamy et al., 2013; Merckx et al., 2011; Veloz, 2009).

Step 2. Initial variable selection

A species group's model was first run incorporating all the variables, default settings and the spatially aggregated three-fold cross validation. If a variable was measured at multiple scales then these were all included in this initial model run. All variables with a mean jackknife test gain of < 0.1 or test AUC (area under the receiver operating characteristic curve) of ≤ 0.6 were identified as poor predictors and removed. The mean values from spatially constrained three-fold cross validation were used for this selection process. The remaining variables were entered into the subsequent variable selection procedure.

Step 3. Model optimisation

The R package 'MaxentVariableSelection' (Jueterbock, 2015; Jueterbock et al., 2016) automates model setting optimisation and variable selection, allowing combinations of user-defined regularisation parameters and features to be compared, a procedure that is laborious to do manually. This is an important, but often neglected step, which can help prevent model under- or over-fitting, and maximise performance (Anderson and Gonzalez, 2011; Merow et al., 2013; Shcheglovitova and Anderson, 2013). We tested combinations of feature types (linear (L), quadratic (Q), product (P), threshold (T), and hinge (H)): L, H, LQ, LQH, LQHPT (following Brown, 2014); and varied the regularisation multiplier by a step of 1, between the values of 1–5. The MaxentVariableSelection procedure uses an iterative process to exclude variables that have a low variable contribution or that correlate with the best performing variable ($r \geq 0.7$). Once a set of uncorrelated variables with an individual variable contribution over the user defined threshold (we used 2%) was identified, the optimal regularisation parameter and feature types were selected using sample-size-adjusted Akaike information criterion (AICc), which favours parsimony (Burnham and Anderson, 2002). AICc values were estimated by the R package from single models built with all presence data (Warren et al., 2010).

Step 4. Model validation

A species group's model was re-run using the selected variables, optimal model settings, and the spatially aggregated data folds to obtain the final measures of model performance, using the R package 'dismo' (Hijmans et al., 2016). We measured mean AUC across the three test folds and applied the Maximum Training Sensitivity and Specificity (MTSS) occupancy rule to partition data into suitable and non-suitable areas (as recommended by Liu et al., 2013). We then measured extrinsic omission rates (the proportion of test points that fall outside this suitable area) and their statistical significance using a binomial test.

Step 5. Model projection

All species data and optimal model settings were used to train the model and project habitat suitability predictions at a 100 m resolution across Edinburgh for both groups of insects.

2.5. Mapping target areas for improving pollinator habitat suitability and connectivity

The direction of the relationships between the predictor variables remaining in the optimised model and the likelihood of species presence (as illustrated by the response curves) were assessed alongside measures of variable importance. This information was used to recommend practical 'on the ground' actions that could be taken to improve urban areas for bumble bees and hoverflies in Edinburgh at the relevant spatial scales (100 m – 2 km).

We also further interrogated the mapped HSI values provided by the models to identify areas to target actions aimed at protecting important pollinator habitats or improving habitat quality and connectivity. This was carried out for bumble bees and hoverflies individually so that actions tailored to each pollinator group can be targeted in the appropriate areas:

1) *Core pollinator habitat to protect*
   We extracted suitable habitat (see Section 2.4.4, step 4) for each pollinator species group (referred to as 'core habitat areas' from this point), where actions to protect pollinator habitats should be targeted.
2) *Poor pollinator areas to improve suitability*
   An additional threshold was applied to the continuous HSI values to select those cells falling within the lowest quartile of HSI values. These areas show the least suitable habitat for each pollinator species group, where we recommend actions to improve pollinator habitats.
3) *Poor pollinator areas to improve connectivity*
   Further analysis was carried out using ArcGIS to identify areas of poor habitat suitability (see 2, above) where habitat enhancements should result in the biggest improvements to pollinator habitat connectivity across the city (referred to as 'corridor target areas' from this point), or more locally (referred to as 'fragmentation improvement areas' from this point). 1 km was selected to represent typical movement distances of these pollinator groups.

i. *Corridor target areas* were located within 1 km of three or more of the largest contiguous areas of core pollinator habitat areas (the core habitats falling within the top two quartiles of polygon areas, which were 1.9 km^2 (bumble bee) or 3.7 km^2 (hoverfly)). These were chosen to represent areas where habitat improvements would help to link up large areas of core habitats, forming longer corridors of high quality pollinator habitat across the city.
ii. *Fragmentation improvement areas* were located within 1 km of a high number of spatially disparate core habitat patches (≥ 15 for bumble bee or ≥ 10 for hoverfly). These were chosen to represent areas where habitat improvements would help to link up clusters of fragmented core habitat patches, resulting in increased local pollinator habitat connectivity.

2.6. Mapping target areas for improving people's health and identifying 'win-wins'

We identified areas of relatively low human health across Edinburgh. To do this, we used the Scottish Government's SIMD (Scottish Index of Multiple Deprivation), which ranks data zones, each containing an average of 800 residents, from most deprived (rank number 1) to least deprived (rank number 6,505) across the country based on a number of indicators (Scottish Government, 2012). We

Table 2
Optimal model settings and mean model performance ± S.D.

Species group	Feature types	Beta multiplier	Train (n)	Test (n)	Test AUC	Core habitat area size (%)	Test omission rates
Hoverfly	LQ	3	27.3 ± 6	13.7 ± 6	0.70 ± 0.1	4.5	0.39***
Bumble bee	H	2	71.3 ± 17	35.7 ± 17	0.69 ± 0.1	21.3	0.30***

Feature types: L = linear, Q = quadratic, H = hinge; Train (n) = average number of training data; Test (n) = average number of test data; Core habitat area size is provided as the percentage of the Edinburgh study area; Test omission rates = average proportion of test data which fell outside of the core habitat area using the MTSS occupancy threshold rule.
*** $P < 0.001$.

identified and mapped the SIMD data zones that fell within Edinburgh's lowest ten quantiles of ranks of the 2012 health indicator (ranks falling between 50 and 1,074). This component of the SIMD incorporates information from several measures of physical and mental health, including mortality rates, hospital stays, illness factors and subscriptions for anxiety, depression or psychosis (Scottish Government, 2012).

These health deprivation zones were then overlaid with the pollinator target zones described above to highlight areas of spatial congruity. These represent 'win-wins' - areas where the provision or improvement of pollinator-friendly, attractive green infrastructure as a nature-based solution is likely to provide benefits to both people's health outcomes and pollinator habitats. Summary statistics were performed in ArcGIS to provide a 'ward' (administrative zone) level overview of the data derived from this study, enabling comparison of the proportional cover of vegetation, tree canopy, the most deprived health zones, pollinator core habitats and poor pollinator areas between wards.

2.7. Engaging stakeholders

We presented our methods and outputs to twenty local experts and decision makers as part of an ELL-led participatory workshop. These included municipal planners and local experts from research organisations, NGOs and charities who had been invited because of their expert knowledge on Edinburgh's greenspaces, pollinators, urban green infrastructure planning or public engagement. Attendees were able to ask questions about the modelling methods and how the data were produced. They were encouraged to discuss and evaluate the usefulness of the data for shaping the pollinator pledge and targeting nature-based solutions in Edinburgh in general. To facilitate discussion, we provided A0 size maps of Edinburgh, overlaid with transparent prints of the pollinator and health target zones. Based on these data maps and their own local knowledge, they were also asked to identify opportunities and provide ideas on what the pledge could entail, where activities should be focussed, and mechanisms for engaging the community (Edinburgh Living Landscape, 2016). Fifteen attendees answered two survey questions at the end of the workshop designed to gauge how accurate and useful they thought our approach was: 'How well do the pollinator model outputs fit with your expectations of habitat suitability across Edinburgh?' (The answer options were 'generally good', 'generally poor' or 'not sure'). And, 'should we rely on outputs for targeting the pledge? (The answer options were 'yes', 'no', or 'don't know').

3. Results

3.1. Mapping urban landscape features

The OBIA analysis revealed that 41.5% (113 km^2) of the surface of Edinburgh's 273 km^2 administrative area is likely to be vegetated or covered by a tree canopy. The estimated tree canopy cover was 12.4% (39 km^2), under which the surface type (vegetation or non-vegetation) is unknown. The mean predicted areal coverage of vegetated surfaces and tree canopies for Edinburgh's 17 administrative wards was 34.4% and 13.9% respectively, compared to 12.8% (vegetation) and 20.6% (trees) across the city centre ward (Supplementary Material, Table SM2). Following segregation of Edinburgh's domestic gardens into predicted tree canopy, vegetation and non-vegetated surfaces, vegetated surfaces and tree canopy cover were estimated to account for 47.8% (14.6 km^2) and 16.8% (5.1 km^2) of total garden area (30.5 km^2) respectively.

3.2. Habitat suitability model performance and pollinator-urban feature interactions

The three-fold cross-validation revealed that the optimal models had reasonable predictive power, with mean AUC values of ≥0.69 and statistically fewer presence records falling outside of the core habitat area than predicted by chance alone (Table 2). The model optimisation procedure reduced the candidate variables (Table 1) down to six variables for both species groups, but the optimal parameter settings identified were different for each.

The hoverfly and bumble bee optimal models incorporated a comparable set of variables (Fig. 2 and Supplementary Material Fig. SM2). These indicated that both species groups had a high affinity with inland water at a small spatial scale, with positive relationships between the likelihood of the presence of each species group and the percentage cover of inland water within 100 m having the highest percentage contribution (38% (bumble bee) and 49% (hoverfly) of total variable contribution) to each model (Fig. 2). Foot-, cycle- and bridle- paths were also predicted to be associated with bumble bee presence, although habitat suitability gradually increased with distance to paths from around 200 m, so that areas further than 1 km from these routes were almost as suitable as the paths themselves (Supplementary Material Fig. SM2). Hoverflies were positively associated with areas close to roads. The presence and amount of allotments within 500 m also had a positive impact on the presence of both pollinator groups, although this relationship was stronger for hoverflies. Trees appear to be important pollinator resources, with the likelihood of bumble bee presence increasing with canopy cover within 1 km, and hoverfly habitat suitability increasing with proximity to trees. Contrastingly, the amount of improved grassland within 1 km negatively influenced bumble bee presence. Against expectations, hoverflies showed a negative association with vegetation cover within 1 km and garden cover within 100 m, although these relationships had a relatively low contribution to the optimal model (9% and 3% respectively).

3.3. Target areas for improving pollinator habitat suitability and connectivity

Bumble bee habitat suitability indices (HSI) tended to be higher across the study area, with 21% of the study area falling within the bumble bee's core habitat area compared to 5% for the hoverfly (Figs. 3 and 4). However, both species showed a similar distribution of their most highly suitable (core) habitats, alongside Edinburgh's rivers and canals, a pattern that was driven by similar strong, positive relationships with inland water at a small spatial scale (Fig. 2). Good bumble bee habitat was also predicted directly alongside Edinburgh's path network or over 1 km from it, but was interrupted by large, homogeneous areas of improved grassland, including golf courses. Large patches of poor hoverfly habitat were predicted to the north and south

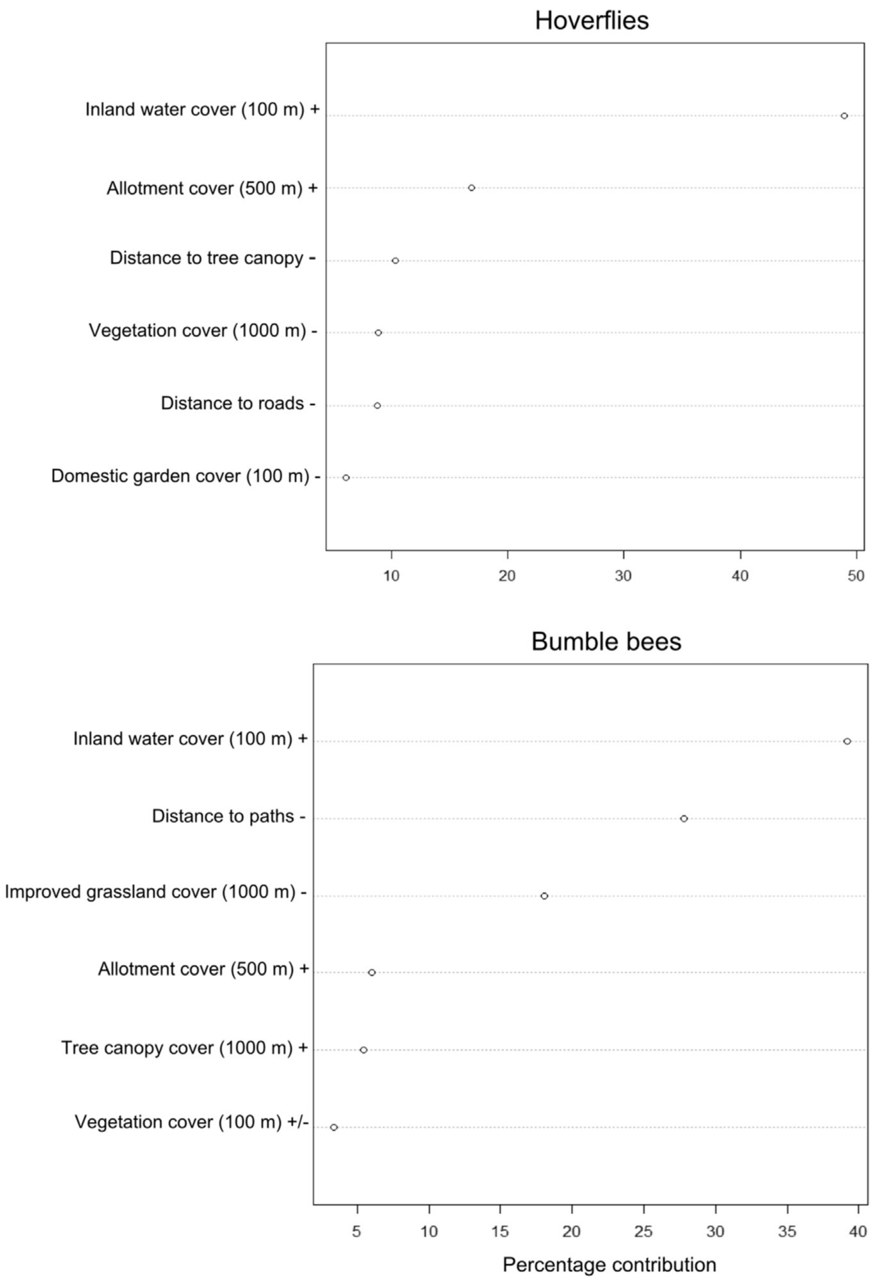

Fig. 2. *Optimal model variable importance as measured by percentage contribution.* The scale at which a variable is measured is provided in brackets (where applicable) and general direction of association is indicated using + (positive), - (negative), +/- (mixed).

of the city, within the rural fringes and away from inland water features and trees.

Because bumble bee and hoverfly core habitats largely followed the river and canal network, they were relatively well connected east to west at a landscape scale. This meant that there were few pockets of poor habitat identified in corridor target areas, where action to improve the green infrastructure for pollinators would help to link up the largest patches of high quality pollinator habitat; those that were found were mainly in the north and east of the city (Figs. 3 and 4). More fragmentation improvement areas were identified, mostly around the periphery of the city centre. This was particularly true for bumble bees because, although this pollinator group's core habitats covered a larger area, these were more fragmented away from the inland water network.

3.4. Target areas for improving people's health

The 55 SIMD data zones that fell within the lowest ten quantiles of Edinburgh's 2012 health deprivation indicator ranks in were clumped in 19 regions around the periphery of the city centre (Fig. 4). The wards with the highest proportional coverage of these health target zones were Portobello/Craigmillar, Forth, Sighthill/Gorgie and Liberton/Gilmerton, and Leith Walk (Supplementary Material, Table SM2). These wards all had lower than average tree cover (< 14%), but only the vegetation cover across Sighthill/Gorgie and Leith Walk wards was lower than the average of 34% (see Supplementary Material Results for further details of the OBIA derived data). Although these data did not appear to show any association between health deprivation and pollinator habitat quality, when the health and pollinator target zones were overlaid, several spatially congruous, ‘win-win’, opportunity areas were identified (Fig. 4). These were mainly in the Liberton/Gilmerton, Portobello, Forth and Craigentinny wards.

3.5. Stakeholder engagement

In general, the HSM models were well received by stakeholders at the ELL-led workshop; eight of fifteen respondents to a post-workshop

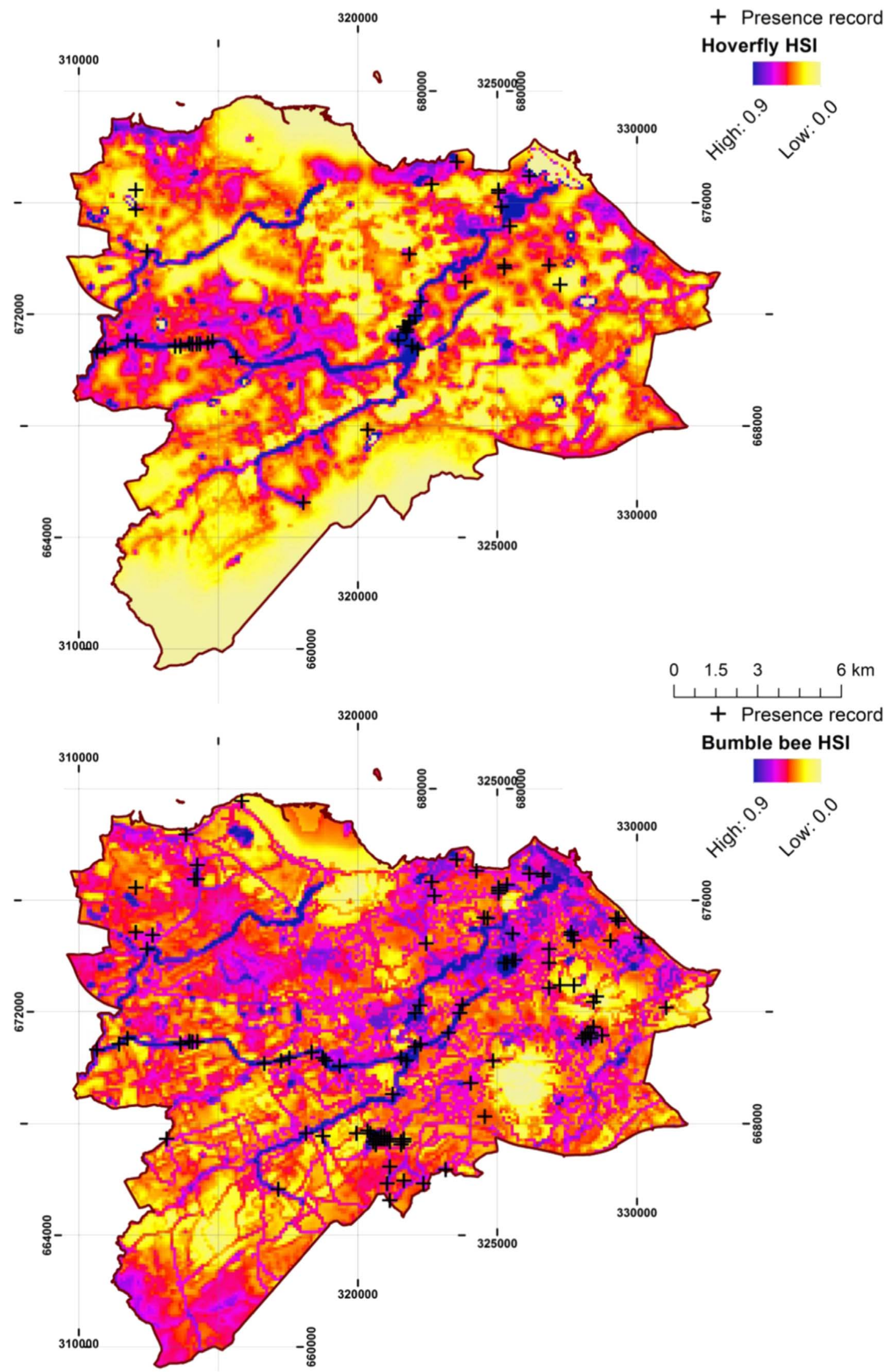

Fig. 3. *Output Habitat Suitability Indices (HSI) for hoverflies and bumble bees.* The higher the HSI value (darker areas), the more suitable the 100 m cell is predicted to be for this species group. The species group presence records used to create the model are overlaid.

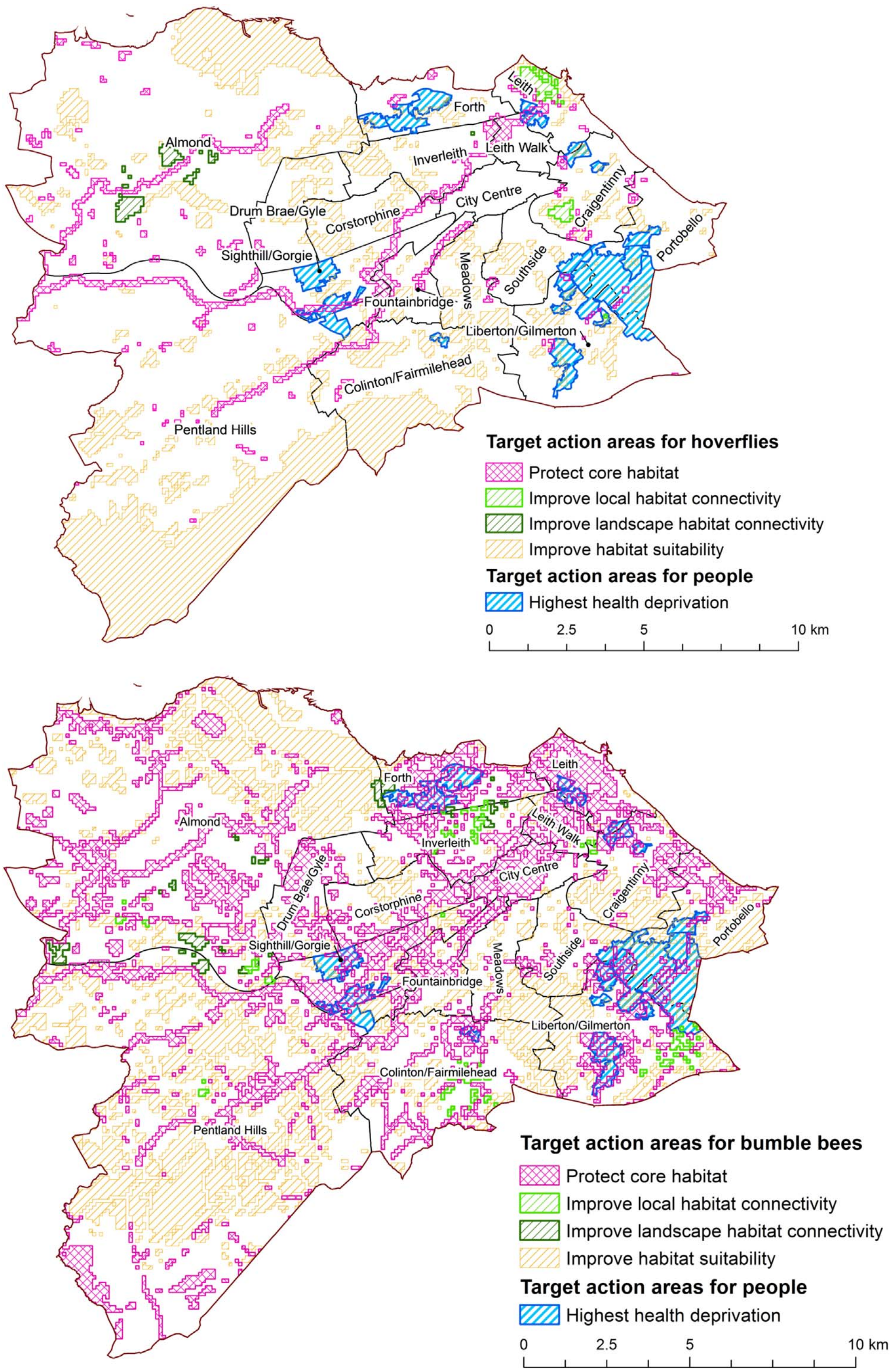

Fig. 4. *Pollinator and people target zones.* Contains OS and National Statistics data © Crown copyright and database right (2016).

survey agreed that the mapped outputs provided a 'generally good' fit of their expectations of pollinator habitat suitability across Edinburgh (seven responded that they were not sure, no one indicated that they thought they were of poor fit). In response to the second question, nine participants indicated that they felt the data should be used for spatially targeting activities and resources for the pollinator pledge (three disagreed with this; the remainders indicated that they were not sure). Facilitated discussion groups during the workshop revealed that participants felt that the pollinator and health data provided a 'good starting point' for shaping the pollinator pledge, but that the HSM should be iteratively validated and improved with new survey data.

Workshop participants also proposed several opportunities for engaging the community, organisations and business actors in a planned ELL pollinator pledge outreach programme. For example, they identified vacant and derelict land, golf courses, school and business grounds, and domestic gardens as potential areas within the target improvement zones for focussing activities. Allotments and watercourses were also highlighted as opportunity areas for activities aimed at protecting and enhancing these important pollinator habitat patches and corridors. Several actions were suggested that the public could pledge in an effort to enhance Edinburgh habitat quality for pollinators, such as reducing mowing intensity, discontinuing the use of pesticides, and improving the structural and floral diversity of greenspaces. It was also agreed that the model outputs should be accessible to the public using 'OurEcosystem', an online spatially explicit tool developed by GREEN SURGE project partner Ecometrica, which is being used to encourage public access and exploration of GREEN SURGE outputs for each case study city (van der Jagt and Viergever, 2015). Following the event, an interactive webpage was developed to summarise and document the discussion for a wider audience (Edinburgh Living Landscape, 2016).

4. Discussion

4.1. The utility of a modelling framework for targeting nature-based solutions

We have demonstrated a structured approach for providing city-wide, quantitative and spatially explicit evidence on *what* green infrastructure improvements should be made, and *where* they should be focused, to increase the impact of nature-based solutions to urban social and ecological issues. Although the small sample size of pollinator records available to us limited the predictive power and reliability the models we were able to produce, we were able to show how these methods can be used as a framework to inform the strategic, targeted use of resources, allowing practitioners and urban planners to get the most 'bang for their buck' from efforts to improve green infrastructure networks. This framework is applicable to any other city and focal species of interest for which data are available, thus providing a transferrable framework that can be adapted to account for local contexts, issues and pressures.

HSM tends to be used in isolation to consider species habitat requirements for biodiversity and conservation planning. Our incorporation of anthropocentric health data is a step towards addressing the need for a 'dramatically improved harmony between the biophysical, socio-economic, and political components of landscapes' (Sandström et al., 2006), and more interdisciplinary methods to inform urban biodiversity planning (Angelstam et al., 2003). HSM is suggested as a useful tool for bridging ecological data gaps in urban green infrastructure planning, but local authorities tend to lack the skills and resources to develop these models in-house (Guisan et al., 2013; Mörtberg et al., 2007; Sandström et al., 2006). Other barriers commonly attributed to applying HSM in decision-making are a lack of input data; mistrust in the models; preference for simpler expert opinion; low regional biodiversity prioritisation; and poor mechanisms for dialogue between planners and modellers (Guisan et al., 2013; Sandström et al., 2006; Tulloch et al., 2016). Angelstam et al. (2003), identified a lack of instruments to facilitate communication between policy and practice as a major institutional obstacle of implementing evidence-based policy on ecological networks. These factors may explain why, despite providing a low-cost, evidence-based approach for supporting strategic conservation decision making, relatively few examples exist of these tools being adopted by practitioners in the peer-reviewed and grey literature (Elith and Leathwick, 2006; Guisan et al., 2013; Tulloch et al., 2016; Villero et al., 2016).

We attempted to address and mitigate many of these major barriers to model uptake and implementation in policy and practice by actively engaging stakeholders in our research. The workshop provided a venue for co-learning and developing ideas for using and improving our approach in partnership. Participants were able to gain a better understanding of the methods involved in developing the data outputs and thus became more aware of opportunities for their application and potential limitations of their use. These types of engagement activities have been shown to provide social benefits, such as increased stakeholder trust in the scientific output, which can result in an increase in the perceived likelihood of meeting biodiversity goals (Young et al., 2013). We were also able to co-develop ideas for implementing the pollinator pledge, combining our scientific outputs with local knowledge to develop innovative ideas for encouraging the public to take action to improve the city for pollinators as part of the pollinator pledge. It is hoped that by improving citizen interest in pollinators and wildlife gardening, any resulting increase in environmental stewardship and voluntarism will improve the likelihood of actions successfully enhancing urban biodiversity and wider ecosystem service provision (Andersson et al., 2014; Dennis and James, 2016), and will increase people's ability to accurately judge biodiversity levels (Shwartz et al., 2014).

The stakeholder feedback helped to inform future research directions and improvements to our methods in an effort to deliver more trustworthy, locally relevant, publically accessible and user-friendly outputs. Although MaxEnt has been shown to cope well with small sample sizes (e.g. ≤ 25, Pearson et al., 2007) and our model validation indicated a reasonable level of accuracy, the feedback from stakeholders indicated that future work should focus on validating the HSM with independent survey data, and on providing iterative updates to the models as new data becomes available. Despite the fact there were only small areas of overlap between our target health and pollinator improvement zones, our stakeholders were able to use these areas to identify areas to prioritise for green infrastructure improvements. As well as collecting this additional pollinator data, participatory GIS (e.g. Brown and Kyttä, 2014; Huck et al., 2014) could be employed to help determine how well our predictions fit with local opinion on the value of particular sites for providing access to nature and other benefits, and for identifying areas where people desire green infrastructure improvements. Furthermore, the fairly widespread and strong appreciation of nature can be used as a 'gateway' for engaging urban stakeholders about wider ecological issues that are not typically considered because they are more inconspicuous, indirect or difficult to understand (Andersson et al., 2015). Future research should also incorporate spatial indicators of the varied cultural and regulatory ecosystem services pertinent to urban areas, enabling an 'ecosystems approach' to urban planning by providing a strategic and tactical framework for managing the environment in a holistic and integrated way (Fish, 2011; Maes et al., 2016; Waylen et al., 2014).

4.2. Recommendations for improving urban areas for pollinators and people

We acknowledge that to be able to confidently infer recommendations to inform pollinator strategies from our HSMs we need to further validate and update them with independent field data. However, many of the urban habitat associations our models identify do make ecological sense and fit with previous studies, and so we discuss them here.

The HSMs indicated that hoverflies and bumble bees show many similar habitat associations, suggesting that a range of green infrastructure solutions can be used which will similarly benefit both species. We found that Edinburgh's rivers and canal are predicted to be important habitat corridors for both species groups, connecting the city at a landscape scale from east to west. This association was particularly marked for hoverflies, which may be explained by the fact that some hoverfly larval stages are aquatic (Stubbs and Falk, 2002). In addition, compared to many other urban greenspace types, the relatively high abundance and richness of the largely unmanaged vegetation along many stretches of these watercourses is likely to provide a substantial, well-connected foraging resource, supporting a high number and diversity of pollinator species. Positive associations were also found with these other linear features, roadsides (hoverflies) and pathways (bumble bees), where parallel vegetation tends to be mown less intensively. Other studies have reported that linear features provide important bumble bee nesting resources in more rural contexts (Kells and Goulson, 2003; Osborne et al., 2008). These habitat corridors are also popular recreation and travel routes for people and therefore provide the health and cultural benefits associated with good opportunities to interact with nature (Bratman et al., 2015; Keniger et al., 2013; Maller et al., 2006). This would suggest that efforts to maintain and enhance these highly valuable green infrastructure networks should be prioritised, particularly along watercourses. This can be achieved at relatively low cost, using low intensity management regimes to maintain tall ruderal and to encourage a diverse wildflower mix with a high proportion of native British weeds and perennials (Carvell, 2002; Hicks et al., 2016). Initiatives to provide and better link pollinator habitat corridors at larger landscape scales, such as BugLife's 'B-Lines' project (BugLife, 2016), should also be supported.

Our models suggest that more space for allotments in towns and cities should be provided, as these greenspaces appeared to have a positive influence on bumble bee and hoverfly habitat suitability at the 500 m scale. This is in agreement with previous studies that have reported the importance of allotments to pollinators in urban settings (Ahrné et al., 2009; Andersson et al., 2007; Tommasi et al., 2005), although the provision of a substantial floral component, rather than a focus on food cultivation, is likely to be important (Foster et al., 2016). These varied and productive urban ecosystems tend to be relatively high in biodiversity in general, and provide a range of cultural, provisioning and regulating services to society, including supporting urban climate adaptation, community cohesion and food production (Barthel et al., 2010; Speak et al., 2015). Planting urban trees and woodlands is another urban greening mechanism that provides pollinator foraging and nesting resources whilst delivering a basket of ecosystem services (Hicks et al., 2016; Monteiro et al., 2016).

In contrast to some previous studies (Ahrné et al., 2009; Bates et al., 2011; Hernandez et al., 2009; Matteson and Langellotto, 2010; McFrederick and LeBuhn, 2006), we did not find any evidence of urban land use intensity reducing bumble bee or hoverfly habitat suitability at any scale, as measures of building cover and road density did not feature in the optimal models. However, increases in improved grassland cover within a 1 km radius of an area appeared to have a strong, negative impact on bumble bee presence, suggesting that large, homogeneous areas of these intensively managed urban habitats do not provide the range of foraging and nesting resources this species group requires. Many studies incorporating urban site characteristic information have reported local nesting and foraging resource availability as a key driver of pollinator abundance or richness (Ahrné et al., 2009; Matteson and Langellotto, 2010; Roy et al., 2016). We were not able to account for this fine-grained resource availability in our models, but greenspace managers and private garden owners should adopt pollinator friendly gardening to improve the urban landscape for these species. For example, planting a diverse wildflower seed mix and less intensive mowing regimes are recommended (Baldock et al., 2015b; Hicks et al., 2016). However, continual improvements to remote sensing technology will enhance our ability to map finer habitat features and characteristics over large areas, improving the accuracy, detail and resolution of habitat suitability models for species with fine-grained habitat associations (Mathieu et al., 2007; Schulp and Alkemade, 2011).

Because of the low sample size of species records available, we were unable to develop models for individual species. It is likely that our models largely represent the habitat requirements of the more generalist bumble bee and hoverfly species, which tend to dominate survey and record centre datasets because of their prevalence (although skilled naturalists tend to record rarer species (van der Wal et al., 2015)). Ideally, we would develop individual species models for a wider range of pollinator insect groups, focusing on particular life cycle stages to provide a more detailed and holistic insight into the value of urban infrastructure for providing foraging, nesting, overwintering and mating resources (Carvell, 2002). Larger, more comprehensive datasets would also allow us to investigate the impact of urban land uses composition and spatial configuration on pollinator abundance and diversity, which is likely to be more marked and informative compared to species presence. Where possible, we suggest investing systematic, tailored field surveys and specialist surveyors to collect larger samples of reliable and precise species data from a stratified selection of field sites across a city to develop an HSM. However, this is an expensive approach to deploy over large areas. Presence-only habitat suitability models built with the most accurate and precise LERC and environmental data available, which have been carefully validated and parameterised to account for sampling bias and to limit model complexity, can provide useful insights and are a much cheaper, more rapid option when resources are limited (Bellamy and Altringham, 2015; Pearson et al., 2007).

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at http://dx.doi.org/10.1016/j.envres.2017.06.023.

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