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### **An evaluation of species distribution models to estimate tree diversity at genus level in a heterogeneous urban-rural landscape**

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*Published in:*

Landscape and Urban Planning

*DOI:*

[10.1016/j.landurbplan.2020.103770](https://doi.org/10.1016/j.landurbplan.2020.103770)

*Publication date:*

2020

*Document Version*

Peer reviewed version

[Link to publication](#)

*Citation for published version (HARVARD):*

Stas, M, Aerts, R, Hendrickx, M, Dendoncker, N, Dujardin, S, Linard, C, Nawrot, T, Van Nieuwenhuysse, A, Aerts, JM, Van Orshoven, J & Somers, B 2020, 'An evaluation of species distribution models to estimate tree diversity at genus level in a heterogeneous urban-rural landscape', *Landscape and Urban Planning*, vol. 198, 103770. <https://doi.org/10.1016/j.landurbplan.2020.103770>

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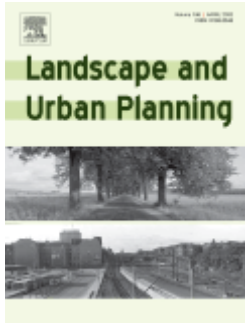
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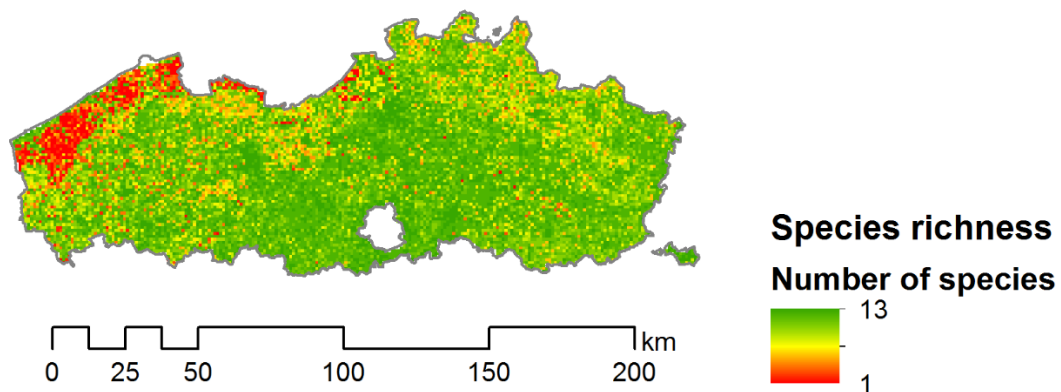
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### Citation:

Stas M., Aerts R., Hendrickx M., Dendoncker N., Dujardin S., Linard C., Nawrot T.S., Van Nieuwenhuysse A., Aerts J.M., Van Orshoven J., Somers B. (2020) An evaluation of species distribution models to estimate tree diversity at genus level in a heterogeneous urban-rural landscape. *Landscape and Urban Planning* 198:103770

<https://doi.org/10.1016/j.landurbplan.2020.103770>

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26 **An evaluation of species distribution models to estimate tree diversity at genus level in a**  
27 **heterogeneous urban-rural landscape**

28

29 **Abstract**

30 Trees provide ecosystem services that improve the environment and human health. The  
31 magnitude of these improvements may be related to tree diversity within green spaces, yet  
32 spatially explicit diversity data necessary to investigate such associations are often missing.  
33 Here, we evaluate two methods to model tree diversity at genus level based on environmental  
34 covariates and presence point data. We want to identify the drivers and suitable methods for  
35 urban and rural tree diversity models in the heterogeneous region of Flanders, Belgium.

36 We stratified our research area into dominantly rural and dominantly urban areas and developed  
37 distribution models for 13 tree genera for both strata as well as for the area as a whole.  
38 Occurrence data were obtained from an open-access presence-only database of validated  
39 observations of vascular plants. These occurrence data are combined with environmental  
40 covariates in MaxEnt models. Tree diversity is modelled by adding up the individual species  
41 distribution models.

42 Models in the dominantly rural areas are driven by soil characteristics (soil texture and drainage  
43 class). Models in the dominantly urban areas are driven by environmental covariates explaining  
44 urban heterogeneity. Nevertheless, the stratification into urban and rural did not contribute to  
45 a higher model quality. Generic tree diversity estimates were better when presences derived  
46 from distribution models were simply added up (binary stacking, True Positive Rate of 0.903).  
47 The application of macro-ecological constraints resulted in an underestimation of generic tree  
48 diversity (probability stacking, True Positive Rate of 0.533). We conclude that summing  
49 presences derived from species distribution models (binary stacking) is a suitable approach to  
50 increase knowledge on regional diversity.

51

52 **HIGHLIGHTS:**

- 53       ➤ Rural species distribution models (SDMs) are driven by soil characteristics.
- 54       ➤ Urban SDMs are driven by environmental covariates explaining urban heterogeneity.
- 55       ➤ Summing presences derived from SDMs is suitable to assess diversity.
- 56       ➤ Summing habitat suitability derived from SDMs underestimates diversity.

57

## 58 **1. Introduction**

59 Trees in urban environments deliver important ecosystem services. Street trees help to cool  
60 cities (Konarska *et al.*, 2016; Scholz *et al.*, 2018), partially mitigating the urban heat island  
61 effect and human heat stress (Lee *et al.*, 2016). Trees can also remove particulate matter from  
62 the atmosphere (Scholz *et al.*, 2018; Selmi *et al.*, 2016). Through climate regulation, air  
63 pollution mitigation and aesthetic and cultural values, trees contribute to better human well-  
64 being (Salmond *et al.*, 2016). Individual trees in the city also contribute to indirect nature  
65 experiences, which benefits human health (Cox *et al.*, 2019). The ecosystem services delivered  
66 by trees are tree species specific (Donovan *et al.*, 2005). Increasing biodiversity is expected to  
67 result in better ecosystem functioning and may yield more stable ecosystem services through  
68 time (Cardinale *et al.*, 2012).

69 Tree diversity has been studied at genus level before (Hoover *et al.*, 2017; Hope *et al.*, 2003)  
70 and might become more popular with increasing availability of observations from citizen  
71 science (Dobbs *et al.*, 2018). Observations of trees through citizen science initiatives have been  
72 found to be more accurate at genus level than at species level (Roman *et al.*, 2017). Plant genus-  
73 level diversity is strongly linked to plant species-level diversity (O'Brien *et al.*, 1998).  
74 Additionally, interactions with host plants often occur at genus level and therefore genus level  
75 diversity is also relevant for insect diversity (Kemp and Ellis, 2017), or ectomycorrhizal fungal  
76 diversity (Gao *et al.*, 2013). Thomsen *et al.* (2016) emphasize that a healthy urban tree  
77 population requires a high generic diversity. Modelling tree diversity at genus level is thus of  
78 high value because it encompasses more reliable observation data and allows for diverse  
79 applications.

80 Genus-level presence-only data can be used in species distribution models (SDMs). SDMs  
81 correlate species observations and environmental covariates to predict habitat suitability (Elith  
82 *et al.*, 2006). Applications of SDMs are manifold: identifying species distributions, studying

83 impact of climate and land use change scenarios (Dyderski *et al.*, 2018) and identifying areas  
84 of interest for conservation (McCune, 2016). Applications of SDMs in urbanized areas are not  
85 common (Della Rocca *et al.*, 2017), yet they have been successfully applied in urban green  
86 spaces (Milanovich *et al.*, 2012) and human-dominated landscapes (McCune, 2016).

87 Species richness can be modelled by stacking individual SDMs on top of one another to yield  
88 a total richness. Stacking of SDMs is most commonly done after thresholding the continuous  
89 probability output of the individual SDMs, a method known as binary stacking (Calabrese *et al.*  
90 *et al.*, 2014). Nevertheless, discretizing continuous probabilities using fixed thresholds (for  
91 example considering all cases with a modelled probability of  $p > 0.55$  as being present) is  
92 generally discouraged (Merow *et al.*, 2013). Instead, species-specific threshold rules can be  
93 applied (Cao *et al.*, 2013). Still, the literature suggests that binary stacking tends to  
94 overestimate species richness because biotic limitations are not accounted for (Calabrese *et al.*,  
95 2014; Gavish *et al.*, 2017; Guisan and Rahbek, 2011). Nonetheless, combining binary SDMs  
96 is the most straightforward method to create species richness maps (Trotta-Moreu & Lobo,  
97 2010). Combining continuous probability data, which is called probability stacking, is an  
98 alternative stacking approach (Calabrese *et al.*, 2014), although interpretations are less  
99 straightforward. Guisan & Rahbek (2011) have proposed a framework for spatially explicit  
100 species assemblage modelling (SESAM). In the SESAM framework, a macro-ecological model  
101 limits the number of species that can co-occur in one cell. One way of defining the macro-  
102 ecological constraint is by stacking the probabilities of SDMs (probability stacking). D'Amen  
103 *et al.* (2015) reduced overestimation by applying probability stacking successfully in the Alps  
104 of western Switzerland at a fine spatial resolution.

105 In this study, we test a stratified approach, in which we run separate SDMs for rural and urban  
106 areas in a mosaic landscape, to model tree diversity at genus level. We evaluate which  
107 environmental covariates drive the urban and rural models. First, we hypothesize that different

108 environmental covariates drive the urban and rural models. We expect that soil nutrients and  
109 soil moisture determine vegetation in rural areas, because this vegetation resembles the  
110 potential natural vegetation more closely (Walthert and Meier, 2017). For urban areas we  
111 expect an anthropogenic influence on the vegetation composition (Bourne and Conway, 2014).  
112 Second, we hypothesize that the application of a macro-ecological constraint to take biotic  
113 interactions into account would improve models for rural areas, but not for urban areas. We  
114 expect that biotic interactions are more relevant in rural areas (D'Amen et al., 2015). Third, we  
115 hypothesize that binary stacking performs sufficiently well in urban areas. We expect that  
116 biotic interactions are less relevant in urban areas because of human intervention.

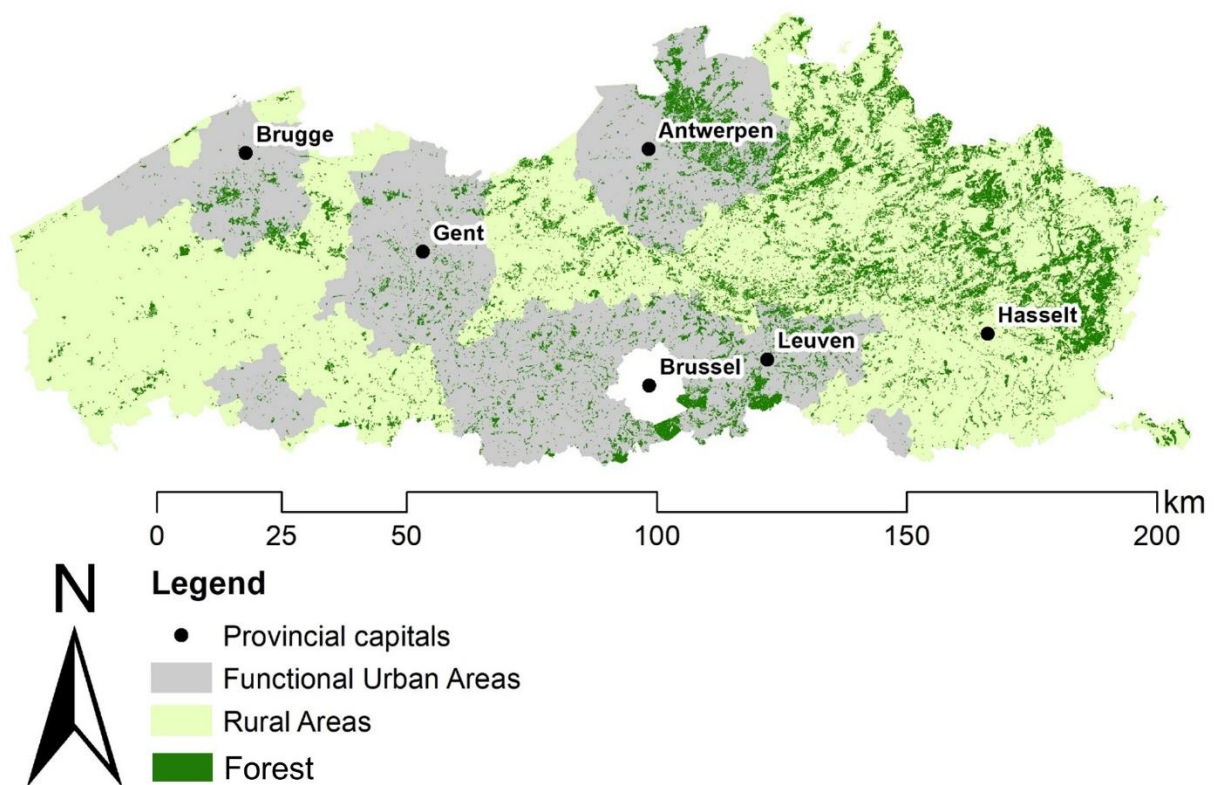
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## 118 **2. Materials and methods**

### 119 *2.1. Study area and stratification*

120 Flanders is the northernmost of the three administrative regions of Belgium with an area of  
121 13,522 km<sup>2</sup> and a population density of 482 inhabitants per km<sup>2</sup>. The area has a north-south  
122 soil gradient of decreasing fraction of sand and increasing fraction of silt. The climate  
123 according to Köppen is a maritime temperate climate (Cfb) (Peel *et al.*, 2007). The  
124 Organization for Economic Co-operation and Development (OECD) considers Flanders  
125 entirely as urbanized (Vervoort, 2016). Nevertheless, based on the Urban Audit of 2018  
126 published by Eurostat, core cities and functional urban areas (FUAs) are delineated for  
127 Flanders. The delineation is used to distinguish dominantly urban areas from more rural areas  
128 in Flanders (Fig. 1). FUAs define a metropolitan area outside the geographical city boundaries,  
129 taking into account demographic, economic and environmental factors (Khalili *et al.*, 2018). In  
130 Flanders, the FUAs are located around the capitals of each province, except for Hasselt (Fig.  
131 1). Hasselt is a provincial capital located in the east of the region where forest cover is higher.  
132 As only 10.6% of the study area consists of forest (De Keersmaeker *et al.*, 2015), other (urban)

133 green spaces are of high importance for biodiversity (Aronson et al., 2017; Lepczyk et al.,  
134 2017). Green space is most commonly defined as a vegetated area (Taylor and Hochuli, 2017).  
135 We will focus on vegetated areas containing woody vegetation. The Belgian and Luxemburg  
136 Institute for Floristics (IFBL) developed a regular grid of 4×4 km squares further divided into  
137 1×1 km squares covering Belgium and Luxemburg. This grid is used as a reference for mapping  
138 species distributions (Van Rompaey, 1943). In this study the 1×1 km IFBL grid is used to mask  
139 observation data and environmental covariates.



140

141 **Fig. 1: Stratification of the Region of Flanders into functional urban areas and rural areas**  
142 **according to the Urban Audit of Eurostat (2018). The provincial capital cities are labeled and the**  
143 **forest cover in the region is visualized.**

144 *2.2. Species distribution modelling*

145 Species distribution models (SDMs) estimate the relationship between recorded occurrences at  
146 sites (samples) and the environmental and/or spatial characteristics of those sites  
147 (environmental covariates) (Elith *et al.*, 2011). This relationship is then used to predict  
148 occurrences elsewhere. In the stratified approach, separate SDMs are run for the urban and  
149 rural strata. Subsequently, the urban and rural data are combined in an integrated approach to

150 form a model that covers the whole region of Flanders. The models are validated with  
151 independent field data to evaluate their utility in the urban and rural strata. The workflow for  
152 the integrated approach, which is parallel for the stratified approach, is visualized in Fig. 2 and  
153 explained in the following sections.

### 2.2.1. Occurrence data

154 The occurrence data of thirteen tree genera were included in the study: *Aesculus* (horse  
155 chestnut), *Alnus* (alder), *Betula* (birch), *Carpinus* (hornbeam), *Corylus* (hazel), *Fagus* (beech),  
156 *Fraxinus* (ash), *Juglans* (walnut), *Platanus* (plane), *Populus* (poplar), *Quercus* (oak), *Salix*  
157 (willow) and *Tilia* (linden). Presence-records of these genera were extracted from Florabank1  
158 (Van Landuyt & Brosens, 2017) available on GBIF.org. The Florabank is an open-access  
159 presence-only database of validated observations of vascular plants, from checklists, literature  
160 and herbarium specimen information. The observations are georeferenced and attributed to the  
161 centers of 14317 1km×1km IFBL grid cells (Van Landuyt *et al.*, 2012).

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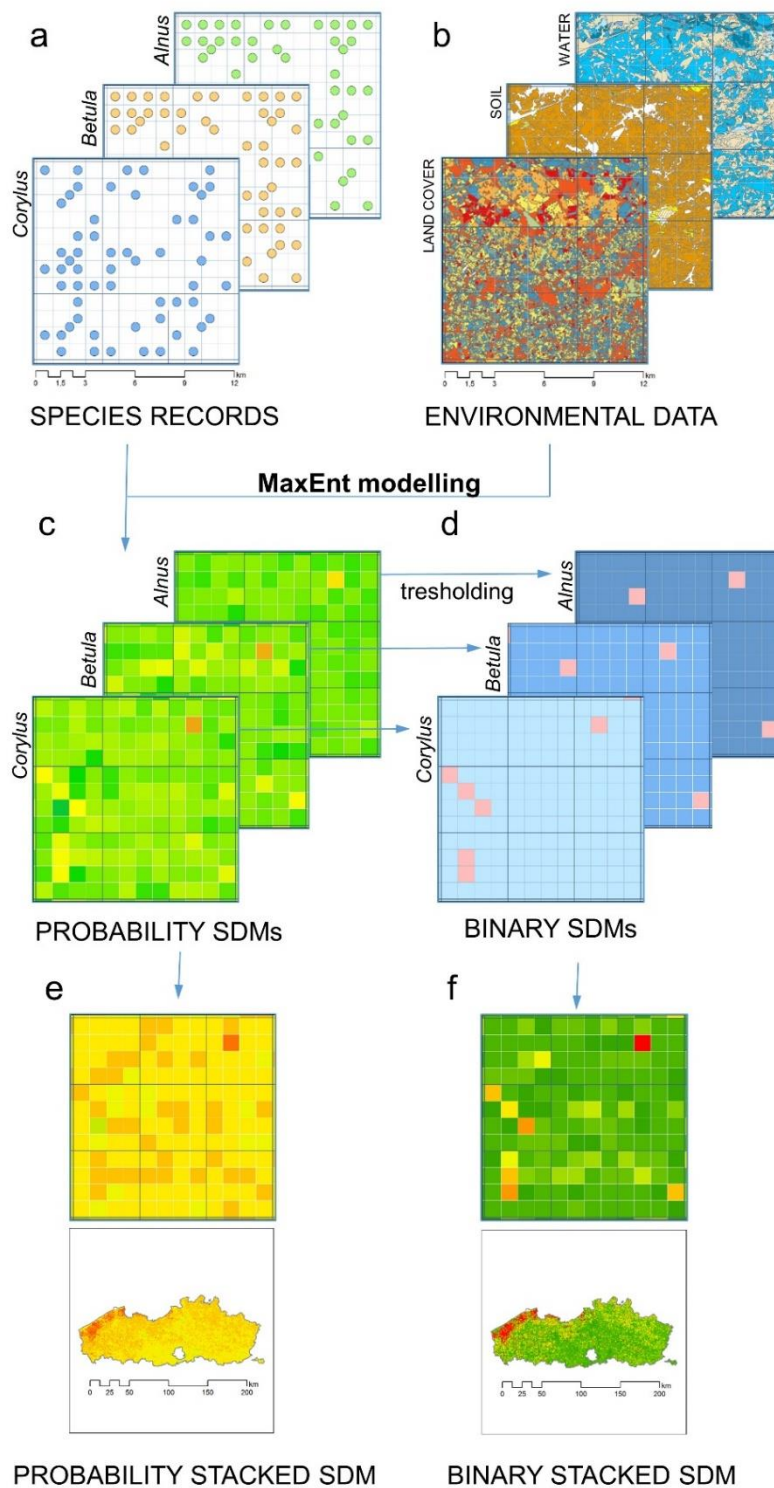


Fig. 2: Mapping tree diversity at the genus level from presence-only and environmental data. Presence-only data from Florabank (a) and environmental data from various sources (b) are combined in a Species Distribution Model (SDM) using MaxEnt. The continuous output of MaxEnt, probability models (c), can be converted to presence-absence models (binary SDMs) by applying genus specific thresholds (d). Stacking the probability models and applying the probability ranking rule results in a probability stacked SDM (e). Aggregating the thresholded SDMs results in a binary stacked SDM (f).

164 2.2.2. *Environmental covariates*

165 Soil texture class, soil drainage class, mean lowest and highest groundwater table depth, land  
166 cover type and habitat type were the environmental covariates used. In Belgium, natural plant  
167 communities are primarily determined by variation in soil nutrient content and soil moisture  
168 (Cornelis *et al.*, 2009). Thus, soil texture and drainage class were extracted as categorical soil  
169 variables from the Belgian soil map (Dondeyne *et al.*, 2014). This vector geodataset was first  
170 resampled to a raster, using the IFBL grid as the mask layer and the cell assignment type  
171 ‘maximum combined area’. The ‘maximum combined area’ rule selects the attribute value from  
172 the polygon with the largest total area overlapping with the grid cell (ESRI, 2017). Mean  
173 highest and lowest groundwater tables data were obtained from a soil hydrology raster  
174 (ECOPLAN, 2014) and resampled to the IFBL grid.

175 Land cover data were obtained from one of the base layers in the ECOPLAN ecosystem  
176 services information system (ECOPLAN, 2014). The geodataset contains a basic land cover  
177 classification (the list of classes is available in Appendix 1). The grid with a spatial resolution  
178 of 5m was resampled to the IFBL grid, retaining the land cover with the largest area in the grid  
179 cell.

180 Habitat data were obtained from the Biological Valuation Map (BVM), a geodataset of habitat  
181 types with attribute information on the ecological context and value of the delineated areas  
182 (Vriens *et al.*, 2011). The BVM contains information about heterogeneity of urban areas, such  
183 as the density and context of built-up areas, industrial areas and recreational areas. The classes  
184 of the BVM are listed in Appendix 1. The BVM is a vector geodataset and was resampled to  
185 the IFBL grid using ‘maximum combined area’ as the cell assignment type. Resampling and  
186 masking of the environmental geodatasets were performed in ArcGIS 10.5.1-software (ESRI,  
187 Redlands, CA, 2017).

188 2.2.3. *Probability models*

189 Probability models of the spatial distribution of each of the 13 genera were developed using  
190 MaxEnt version 3.3.3k. MaxEnt is a machine-learning algorithm highly suitable to develop  
191 models from presence-only data (Elith *et al.*, 2006; Phillips *et al.*, 2006). The algorithm is based  
192 on the principles of maximum entropy and finds an optimal probability distribution using a  
193 combination of occurrence data and environmental data (Elith *et al.*, 2011). MaxEnt is known  
194 to perform well even when environmental covariates are linearly correlated (De Marco and  
195 Nóbrega, 2018). The logistic output of MaxEnt is an attempt at expressing the raw output as a  
196 probability of presence (Elith *et al.*, 2011). A 10-fold cross validation was applied. Model  
197 performance was assessed with the area under the receiver operating characteristic curve  
198 (AUC) statistic, ranging between 0 and 1. When AUC values are higher than 0.5, the model  
199 performs better than a random distribution. For every genus, three models were developed: one  
200 using the entire dataset (integrated approach), then one for the rural and one for the urban areas  
201 (stratified approach). To evaluate the driving factors in these models, we determined the  
202 environmental covariate with the highest percentage of contribution to the model.

203

#### 204 *2.2.4. Binary stacking*

205 We applied the ‘10 percentile training presence’ rule on the MaxEnt-output (Ficetola *et al.*,  
206 2009; Pearson *et al.*, 2006; Skowronek *et al.*, 2017), for every genus and model approach  
207 separately, resulting in a threshold value above which 90% of the training samples are correctly  
208 classified. Thus, a unique threshold value is used for every genus to create a binary output (0  
209 = absence, 1 = presence). Binary stacking is the process of adding up the individual binary  
210 models, resulting in a generic tree diversity varying from 0 to 13 genera.

211

#### 212 *2.2.5. Probability stacking*

213 As a cell-specific macro-ecological constraint we summed the MaxEnt-probabilities per grid  
214 cell (D'Amen *et al.*, 2015), resulting in a possible generic tree diversity range between 1.96  
215 and 8.76. To determine which genera occur in the constrained cells we used the 'probability  
216 ranking' rule. The genera are assigned to the cell according to decreasing order of probability  
217 of presence determined from the SDMs (2.2.3), until the cell-specific macro-ecological  
218 constraint is reached. Probability ranking as described in the SESAM framework is  
219 incorporated in the package 'ecospat' available for R (Di Cola *et al.*, 2017) and was executed  
220 with R software 3.4.3 (R Core Team, 2017).

221

### 222 2.3. Validation

223 The probability models (2.2.3) were cross-validated before they were stacked. In addition, the  
224 stacked models were validated with independent field data. The independent field data  
225 consisted of recordings of the genus's occurrence around 208 randomly selected point  
226 locations, with a search effort per point of ten minutes with two observers. The sampling  
227 protocol is derived from the timed-meander sampling protocol (Goff *et al.*, 1982), which is  
228 applied in various fields of ecology (Threlfall *et al.*, 2017) and is favored because of its cost-  
229 effectiveness (Hamm, 2013). The 208 point locations are distributed over 130 IFBL cells.  
230 There are 87 rural cells and 43 urban cells. A genus is present in a cell if it is observed in at  
231 least one of the random point locations within the cell. The field data are assumed to provide  
232 the true condition that is compared to the predicted condition provided by the SDMs at genus  
233 level. True condition data and predicted condition data were compared in a confusion matrix,  
234 describing true positive (TP), true negative (TN), false positive (FP) and false negative (FN)  
235 outcomes. Based on the values in the confusion matrix, we evaluated the model performance  
236 by calculating the true positive rate (TPR). TPR is the number of true positives divided by the  
237 total of positive cases, the sum of true and false positives. The TPR informs simultaneously

238 about the presences that are correctly predicted and about those that were incorrectly identified  
239 as positives. A TPR of 80% would indicate that 80% of the presences are true positives while  
240 20% are false positives. However, in the present study, the false positives are not necessarily  
241 false as the species could have been missed during the validation field work. Therefore we  
242 focus on the true positives when interpreting the TPR. Additionally, the percentage of false  
243 negatives is included in the evaluation, because this percentage provides information on the  
244 underestimation of the stacking method. The higher the percentage of false negatives, the more  
245 the tree diversity at genus level is underestimated.

246

#### 247 *2.4 Compare model outcomes*

248 To compare model outcomes we calculated average modelled tree diversity at genus level and  
249 95% confidence intervals for binary stacked vs. probability stacked models and for integrated  
250 vs. stratified approaches and for urban vs. rural areas. We used the paired sample t-test (with a  
251 statistical cutoff value of 0.05) to test whether overall average modelled tree diversity at genus  
252 level differed between binary stacked and probability stacked models. We then used the paired  
253 sample t-test to test whether modelled generic tree diversity differed between integrated and  
254 stratified approaches within the binary stacked models, both for the entire dataset and for a  
255 dataset stratified in urban vs. rural areas.

256

257 **3. Results**

258 *3.1 Probability models*

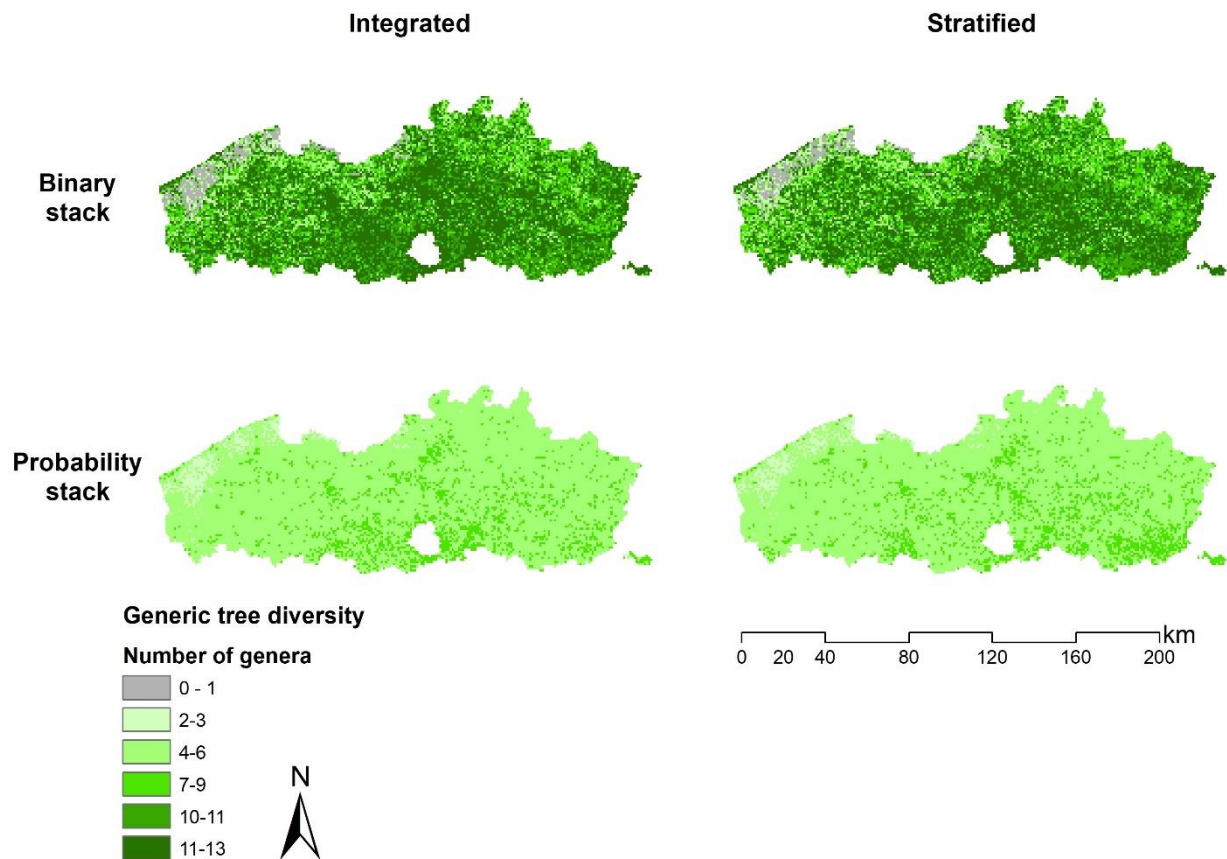
259 The species distribution models outperformed the random spatial distribution (all AUC > 0.5;  
260 Table 1), for the stratified approach as well as the integrated approach. On average the AUC is  
261 0.60 with a standard deviation of 0.007. The strength of the strongest predictor ranges from  
262 32.7-85.7 percent of contribution (Table 1). For 11 out of 13 urban models, the strongest  
263 predictor is the Biological Valuation Map (BVM), containing information on urban  
264 heterogeneity. For the rural model as well as the integrated model, we found that for some  
265 genera the strongest predictors were the soil variables texture class and drainage class.

**Table 1: Summary of the species distribution models for each genus. Reporting the number of grid cells occupied by an observation (presences), the average area under the curve (AUC) as a measure to evaluate the models, the standard deviation of the AUC (SD), the strongest predictor (SP) and the percent of contribution (PC) of this strongest predictor to the MaxEnt model. (BVM = Biological Valuation Map)**

Genus	presences	Rural stratum				Urban stratum				Integrated model			
		AUC	SD	SP	PC	AUC	SD	SP	PC	AUC	SD	SP	PC
<i>Aesculus</i>	1162	0.63	0.03	BVM	54.3	0.62	0.04	BVM	61.2	0.64	0.03	BVM	55.1
<i>Alnus</i>	6918	0.55	0.01	BVM	38.0	0.54	0.02	BVM	40.7	0.54	0.01	BVM	39.9
<i>Betula</i>	8888	0.60	0.01	BVM	40.9	0.58	0.02	BVM	45.6	0.58	0.01	BVM	47.7
<i>Carpinus</i>	3029	0.64	0.02	texture class	42	0.63	0.02	BVM	29.2	0.63	0.02	texture class	41.3
<i>Corylus</i>	5490	0.60	0.02	texture class	45.8	0.59	0.02	BVM	34.5	0.59	0.01	texture class	47.1
<i>Fagus</i>	2710	0.64	0.02	drainage class	38.1	0.64	0.02	BVM	32.7	0.64	0.02	drainage class	37.2
<i>Fraxinus</i>	6635	0.58	0.01	texture class	67.6	0.55	0.02	texture class	42.6	0.57	0.01	texture class	71.2
<i>Juglans</i>	1833	0.61	0.03	texture class	44.9	0.59	0.03	BVM	40.3	0.61	0.02	texture class	46.5
<i>Platanus</i>	50	0.63	0.34	BVM	47.5	0.90	0.06	BVM	81.7	0.81	0.10	BVM	85.9
<i>Populus</i>	5380	0.57	0.02	drainage class	55.9	0.56	0.02	BVM	41.8	0.56	0.01	drainage class	47.1
<i>Quercus</i>	8182	0.57	0.01	BVM	36.2	0.55	0.02	texture class	38.4	0.55	0.01	BVM	38.7
<i>Salix</i>	8349	0.53	0.01	drainage class	65.0	0.52	0.02	BVM	39.9	0.52	0.01	drainage class	46.4
<i>Tilia</i>	1171	0.61	0.04	BVM	42.6	0.63	0.04	BVM	54.3	0.64	0.03	BVM	43.8

### 3.2 Stacked species distribution models

For the integrated as well as the stratified approach binary stacking resulted in a generic tree diversity varying between 0 and 13. Probability stacking resulted in a lower generic tree diversity between 2 and 9 (Fig. 3). Spatial differences in generic tree diversity between the integrated and the stratified approach are not strongly pronounced.



**Fig. 3:** Tree diversity at genus level determined by binary (upper) and probability (lower) stacking of the MaxEnt models developed in an integrated (left) and stratified (right) modelling approach. The cell size is 1km×1km.

### 3.3 Validation

The binary stacking method, with an overall TPR of 0.90, performed better than the probability stacking method, with a considerably lower overall TPR of 0.52-0.53 (Table 2). Overall, the integrated and the stratified approach performed equally well. The binary stacking method had a lower percentage of false negatives (6.0 - 6.1 %) than the probability stacking method (29.3 – 29.9 %) (Table 3).

**Table 2: Validation results: true positive rate (TPR) derived from the confusion matrix.**

Model	Approach	TPR		
		Overall	Urban	Rural
Binary stacked	Integrated	0.90	0.94	0.89
	Stratified	0.90	0.95	0.88
Probability stacked	Integrated	0.53	0.54	0.53
	Stratified	0.52	0.53	0.53

**Table 3: Validation results: percentage of false negatives (%) derived from the confusion matrix.**

Model	Approach	FN (%)		
		Overall	Urban	Rural
Binary stacked	Integrated	6.1	4.3	7.0
	Stratified	6.0	3.3	7.3
Probability stacked	Integrated	29.3	30.8	28.6
	Stratified	29.9	31.5	29.2

### 3.4 Comparison of model outcomes

#### 3.4.1 Binary stacking vs. probability stacking

The overall average tree diversity at genus level was higher for binary stacked models (9.8-9.9) than for probability stacked models (5.6) (Table 4). The diversity based on the validation data was 8.1, in line with results from the binary stacking approach. There was a significant mean difference of 4.3 (95% CI 4.2-4.3) between binary stacked and probability stacked models for the integrated approach (paired t-test  $t = 200.1$ ,  $df = 13458$ ,  $p < 0.001$ ). There was a significant mean difference of 4.2 (95% CI 4.1-4.3) between binary and probability stacked models for the stratified approach (paired t-test  $t = 153.7$ ,  $df = 13458$ ,  $p < 0.001$ ).

*Table 4: Average modelled tree diversity at genus level based on binary and probability stacked models, for integrated and stratified approaches and for urban and rural areas.*

Model	Approach	Tree diversity (95% CI)		
		Overall	Urban	Rural
Binary stacked	Integrated	9.9 (9.8-9.9)	10.1 (10.0-10.1)	9.8 (9.7-9.8)
	Stratified	9.8 (9.7-9.8)	9.5 (9.4-9.6)	10.0 (9.9-10.0)
Probability stacked	Integrated	5.6 (5.6-5.6)	5.7 (5.7-5.8)	5.6 (5.5-5.6)
	Stratified	5.6 (5.6-5.6)	5.5 (5.5-5.6)	5.6 (5.6-5.6)
Validation data		8.1 (7.6-8.7)	8.6 (7.8-9.5)	7.9 (7.3-8.6)

#### 3.4.2 Binary stacking: rural vs. urban areas

For binary stacked models, the integrated approach yielded a statistically significant higher overall estimated tree diversity at genus level [mean difference integrated vs. stratified 0.09 (95% CI 0.06-0.11),  $t = 6.72$ ,  $df = 13458$ ,  $p < 0.001$ ]. However, subtracting the stratified result from the integrated result revealed a spatial differentiation (Fig. 4). The stratified approach resulted in a significantly higher diversity in the rural areas, thus a negative mean difference of  $-0.18$  (95% CI  $-0.21 - -0.15$ ) ( $t = -12.5$ ,  $df = 8359$ ,  $p < 0.001$ ). The integrated approach, however, resulted in a significantly higher diversity in the urban areas, thus a positive mean difference of  $0.52$  (95% CI  $0.48-0.57$ ) ( $t = 22.3$ ,  $df = 5098$ ,  $p < 0.001$ ).

Green pixels (Fig. 4) represent a higher diversity obtained with the integrated approach. These green pixels are often clustered within the FUAs (black outline, Fig. 4). Conversely, red pixels represent a higher diversity obtained with the stratified approach. These red pixels are observed in clusters outside the FUAs, especially in extremely rural areas such as: ‘De Westhoek’ in the west and ‘Haspengouw’ in the south-east of Flanders (Fig. 4).

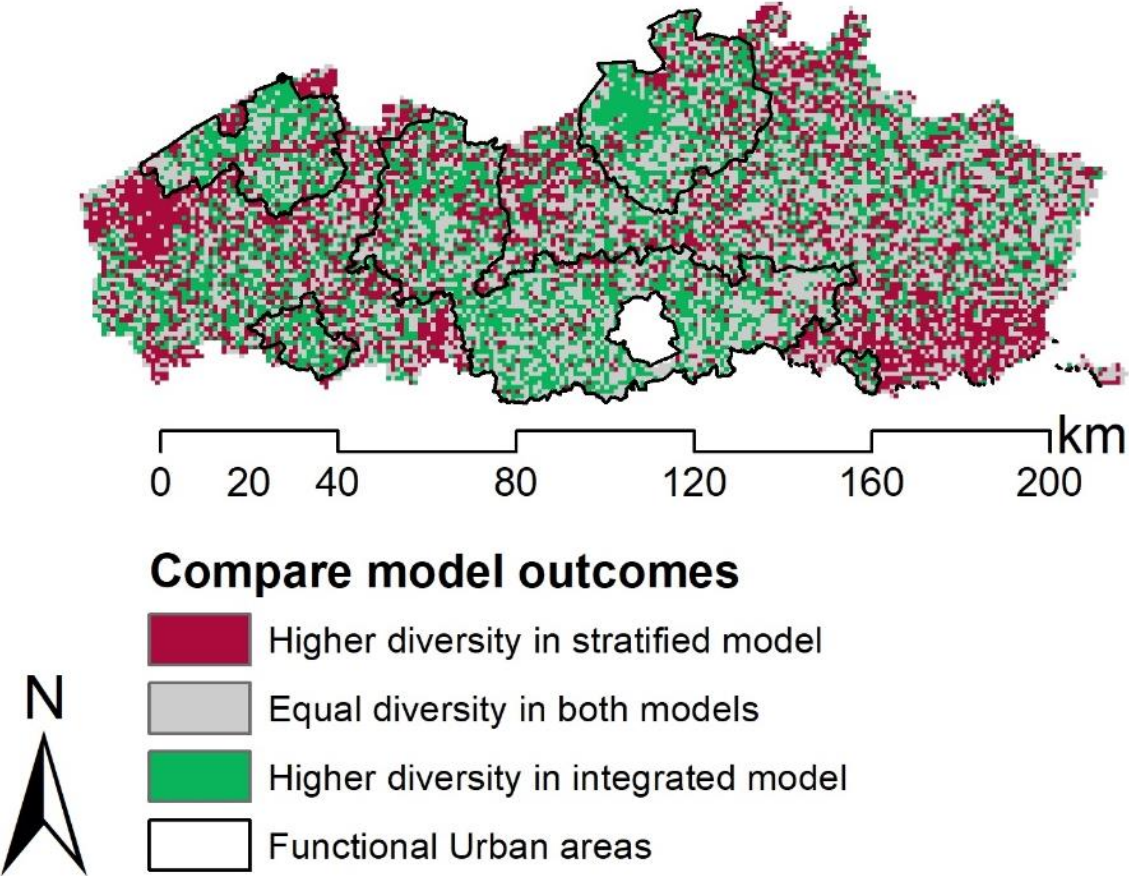


Fig. 4: The difference in the binary stacking results from the integrated and stratified approach. Cell size is 1km×1km.

## 4. Discussion and conclusions

### 4.1 Environmental covariates

Rural and urban models were driven by different environmental covariates, which confirms the first hypothesis. As expected, soil variables, such as texture class and drainage class, were of high importance to explain the distribution of native trees in rural areas. Texture class was an important covariate for *Corylus*, for example, because this genus requires richer loamy soils (Özenç, 2001). *Salix* and *Populus* can tolerate wet soils (Zalesny and Bauer, 2007) and as a result drainage class was an important environmental covariate in their SDMs. Earlier studies demonstrated that including soil factors in plant SDMs results in improved predictions (Buri *et al.*, 2017). For urban areas we expected that the vegetation would be determined by anthropogenic influences. The urban heterogeneity, which is better described in the Biological Valuation Map (BVM), was the most important covariate in the urban SDMs (Table 1). In the BVM seven urban/built up types are included (ua, ud, un, ur, uv, uc and ui), while the other land cover map (ECOPLAN) contains only three urban land cover types (9101, 9201, 9202) (see Appendix 1). It has been emphasized in the past that including environmental covariates that account for the diverse functions of urban areas is important to understand urban plant species patterns (Godefroid and Koedam, 2007). Future developments of species distribution models in urban areas need to include covariates that address the variety of anthropogenic influence.

### 4.2 Stacking methods

At a fine spatial resolution, as in the work of D'Amen *et al.* (2015), binary stacking overestimates species diversity in a natural environment, because dispersal limitations and biotic interactions are not taken into account. In this study, however, we worked at a spatial resolution of 1km×1km and biotic interactions are less important at this relatively coarse resolution (Thuiller *et al.*, 2015). The probability stacking method should prevent from overestimating tree diversity at genus level. Nevertheless, the low true positive rate (Table 2)

and the high percentage of false negatives (Table 3) showed that the probability stacking method underestimated tree diversity in Flanders. Additionally, in urbanized areas, human decision-making and management most likely override natural species selection and biotic interactions are therefore less likely to drive species composition. Therefore, binary stacking is the preferred method for biodiversity modelling at 1km×1km resolution in both urban areas and rural areas. Nevertheless, the scale-dependent applicability of a macro-ecological constraint needs further research as there is, to our knowledge, no literature on this topic.

#### *4.3 Comparison of model outcomes*

Binary stacking resulted in a significantly higher diversity at genus level compared to probability stacking (Table 4). This difference is not an overestimation of binary stacking, but due to an underestimation of probability stacking. Using the binary stacking method and the stratified approach, higher diversity was clustered in the rural areas (Table 4 and Fig. 4). Rural areas are thus more prone to overestimation and would probably benefit more from applying macro-ecological constraints, compared to urban areas. Nevertheless, applications of macro-ecological constraints seem to be of higher relevance in more natural areas (D'Amen *et al.*, 2015), far less urbanized than the rural areas in Flanders.

To conclude, we find that binary stacking is most suitable for both urban and rural areas in Flanders. Stratification of the study area did not improve model quality considerably, but confirmed that different environmental covariates contributed to the models of urban and rural areas. Probability stacking is to be considered in natural areas, but does not perform well in urbanized areas, especially at the moderate spatial resolution of 1km×1km.

#### *4.4 Limitations*

All Species Distribution Models (SDMs) had relatively low Area Under the Curve values (average AUC:  $0.60 \pm 0.01$ ), but all performed better than random distributions (Table 1). By

stacking SDMs, errors in individual species models accumulate and degrade predictions of species diversity (D'Amen *et al.*, 2015; Pottier *et al.*, 2013). The importance of the BVM as an environmental covariate emphasizes the relevance of including urban heterogeneity in SDMs. Unfortunately, at a moderate resolution of 1×1km relevant intra-urban variation of the tree canopy (Weinberger *et al.*, 2016) cannot be observed.

#### *4.5 Applications*

The model resulting from this study can be expanded by stacking more binary SDMs, by producing species-level models or by producing models of other plant groups. Spatially-explicit biodiversity data are vital for emerging environmental health studies (McInnes *et al.*, 2017), for example to study relationships between residential and dynamic exposure and human health outcomes (Cox *et al.*, 2017; Shanahan *et al.*, 2016). Hjort *et al.* (2016) present a concept to calculate individual long-term or life time exposure to pollen with geographic information systems. Landscape and urban planners could also use tree diversity maps to identify areas with low diversity and optimize the delivery of ecosystem services or decrease potential social inequalities in access to biodiverse green space by increasing biodiversity in focus areas (Wolch *et al.*, 2014). Finally, when subsets of models for allergenic species are used, diversity maps could be interpreted as allergy risk maps and inform pollen allergy patients about pollen allergy risks (McInnes *et al.*, 2017).

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**Appendix 1:**

Table 1: Ecoplan land cover classification (in Dutch). Retrieved from: <https://www.uantwerpen.be/nl/onderzoeksgroep/ecoplan/ecoplan-tools/ecoplan-geoloket/>

<b>Complex</b>	<b>Code</b>	<b>Dutch name</b>
water	10101	Stilstaand water
	10201	Getijde mesohaline
	10202	Getijde Oligohalien
	10203	Getijde zoet
	10204	Zoet
	10301	Zee
Bush land	1101	Ruigten en pioniersvegetatie
	1201	Struweel
	1301	Struiken boomgaard
	1302	Laagstam
	1303	Hoogstam
	1401	Ander hooggroen
	1402	Ander laaggroen
Forest complexes	2101	Berk
	2102	Beuk
	2103	Beuk – naaldhout
	2104	Eik
	2105	Eik – naaldhout
	2106	Populier
	2107	Populier – naaldhout
	2108	Ander loofhout
	2109	Ander loofhout – naaldhout
	2201	lork
	2202	Lork – loofhout
	2203	Fijnspar
	2204	Fijnspar – loofhout
	2205	Zwarte den
	2206	Zwarte den – loofhout
	2207	Grove den
	2208	Grove den – loofhout
	2209	Ander naaldhout
	2210	Ander naaldhout – loofhout
	Grasslands	3101
3102		Voedselrijk grassland
3201		Voedselrijk grasland
Heath lands	4101	Droge heide
	4201	Vochtige heide
Bare soils	5101	Kale bodem
	5201	Duinen
	5301	Strand
	5401	Niet verharde wegen
Agricultural land	6101	Aardappel
	6102	Mais
	6103	Graan
	6104	Zaden
	6105	Peulvruchten
	6106	Suikerbiet

	6107	Groenten
	6108	Kruiden
	6109	Fruit
	6110	Vlas en hennep
	6111	Voedergewassen
	6112	Andere akker
	6201	Eénjarige sierplanten
	6202	Meerjarige sierplanten
	6203	Kerstbomen
Wet vegetation types	7101	Moeras
	7201	Rietland
	7301	Alluvial bos
	8101	Slikken
	8201	Schorren
Built up area	9101	Gebouwen
	9201	Verhard
	9202	Verhard andere

Table 2: Biological valuation map classification of habitat types (in Dutch). Retrieved from: <https://www.geopunt.be/catalogus/datasetfolder/bf31d5c7-e97d-4f71-a453-5584371e7559>

Complex	Code(s)	Dutch name
Stagnant water	ad	Bezinkingsbekken
	ae, aer, aev	Eutroof
	ap, apo, app	Diep of zeer diep water
	ao, aoo, aom	Oligotroof tot mesotroof water
	ah	Brak of zilt water
Swamps	ms	Zuur laagveen
	mm	Galigaanvegetatie
	mk	Alkalisch laagveen
	mc	Grote zeggenvegetatie
	mz	Brak tot zilt moeras
	mr	Rietland en andere Phragmition vegetaties
	md	Drijfzoom en/of drijftil
Grasslands	ha	Struisgrasvegetatie
	hc	Dotterbloemgrasland
	hk	Kalkgrasland
	hm, hmm, hme	Vochtig schraalgrasland
	hmo	Vochtig heischraalgrasland
	hn	Droog heischraalgrasland
	hu	Mesofiel hooiland
	hj	Vochtig grasland gedomineerd door russen
	hp×, hpr×	Soortenrijk premanent cultuurgrasland
	hpr(×)+da, hp(×)+da, h+da	Soortenrijk premanent cultuurgrasland met zilte elementen
	hp	Soortenarm permanent cultuurgrasland
	hx	Zeer soortenarm, vaak tijdelijk grasland
	hf, hfc, hft	Moerasspirearugte
	hr	verruigd grasland
	hz	grasland op zware metalen vergiftiged bodems
hpr	weidelandcomplex met veel sloten of microreliëf	

High fenn	t	hoogveen
Heath lands	cg	Droge struikheivegetatie
	cv	Droge heide met bosbes
	ce, ces	Vochtige tot natte dopheivegetatie
	cm	Gedegradeerde heide met dominantie van pijpenstrootje
	cp	Gedegradeerde heide met dominantie van adelaarsvaren
	cd	Gedegradeerde heide met dominantie van bochtige smele
Dunes and tidal flats	ds	Slikken
	da	Schorre
	dd	Stuifduinen aan de kust
	dl	Strand
	dz	Zandbank
Bush land	sd(b)	Duinstruweel
	sp	Doornstruweel
	sk	Struweel op kalkrijke bodem
	sg, sgu, sgb	Brem- en gspeldoornstruweel
	sz	Opslag van allerlei aard
	sf	Vochtig wilgenstruweel op voedselrijke bodem
	so	Vochtig wilgenstruweel op venige of zure grond
	sm	Gagelstruweel
	se	Kapvlakte
Beech forests	fe	Beukenbos met wilde hyacint
	fa	Beukenbos met voorjaarsflora, zonder wilde hyacint
	fm	Beukenbos met parelgras en lievevrouwebedstro
	fk	Beukenbos op mergel
	fl	Beukenbos met witte veldbies
	fs	Zuur beukenbos
Oak forests	qe	Eiken-haagbeukenbos met wilde hyacint
	qa	Eiken-haagbeukenbos
	qk	Eiken-haagbeukenbos op mergel
	ql	Eikenbos met witte veldbies
	qs	Zuur eikenbos
	qb	Eikenberkenbos
Wet forests	vc	Bronbos
	va	Alluviaal elzen-essenbos
	vf	Elzen-eikenbos
	vn	Nitrofiel alluviaal elzenbos
	vm	Elzenbroek
	vo	Oligotroof elzenbroek met veenmossen
	vt	Berkenbroek
Ruderal forests	ru, rud	ruderaal olmenbos
Coniferous forests	pi, ppi, pa, ppa	Naaldhoudsbestand zonder ondergroei
	pmh, pms, pmb, ppmh, ppms, ppmb	Naaldhoutbestand met ondergroei
Poplar forests	lhi, lhb, lsi, lsb, lsh	Populiersbestand

Other deciduous forests	n	Loofhout aanplant (exclusief populier)
Agricultural fields	bk, bl, bs, bu	Akker
Urban and built up areas	ua, ud, un, ur	Bebouwing
	uv, uc	Recreatiegebied
	ui	Industrie
Small landscape elements	kj	Hoogstamboomgaard
	kb	Bomenrij
	kh	Houtkant
	khw	Houtwal
	k	lijnvormige begroeiing van perceelsranden, sloten en bermen
	kk	Doline, ingang ondergrondse mergelgroeve
	km	Muurvegetatie
	kn	Veedrinkpoel
	kt	Talud
	kw	Holle weg
Other mapped elements	ko	Stort
	kr	Groeve
	kf	Voormalig militair fort
	kg	Terril
	kz	Opgehoogd terrein
	ki	Vliegveld
	kg	Kwekerij of Serre
	ka	Eendenkooi
	kr	Rots
	kd	Dijk
	ks	Verlaten spoorweg
	kl	Laagstamboomgaard
	kp	Park
	kpa	Arboretum
	kpk	Kasteelpark