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Testing the applicability of ecosystem services mapping methods for peri-urban contexts: A case study for Paris.

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Abstract

Through their semi-natural and agricultural areas, peri-urban regions are pivotal in providing ecosystem services (ES) to city dwellers. To quantify the ES provided by these areas, it is possible to use ES mapping methods. Many ES mapping methods rely on land cover maps, but most maps are coarse compared to the peri-urban scale. Nevertheless, readily-available land use data and methods are often used to map ES at such scales, without contextualisation. As a result, such methods may not be able to capture the diversity that is present in the peri-urban vegetation, which could have consequences for their accuracy and furthermore for urban planning policies.

To increase our understanding of the applicability of ES mapping methods in peri-urban regions, we assessed to what degree sites with similar plant composition in the green belt of Paris, France, were also projected to have similar ES bundles. We considered two commonly used ES model types: proxy-based models (here: look-up tables) and phenomenological models. We used 252 sites for which botanical survey data were available and applied the ES models to seven ES relevant in the peri-urban context. A cluster analysis was used to group sites, hence facilitating analyse of the spatial congruence between types of vegetation and bundles of ES.

Clustering sites based on plant composition revealed six distinct clusters. Clustering sites based on ES bundles as estimated by phenomenological models and proxy-based models, resulted in four and two clusters, respectively. The proxy-based clustering only highlighted broad-leaved forests as an important ES supply source. The phenomenological model estimates of ES allowed a more nuanced clustering of sites into four different groups. The level of congruence between the different sets of clusters based on plant composition and estimated ES bundles was low. Except for forests, the commonly used ES models tested here were not able to represent the same level of heterogeneity in the peri-urban landscape as was found in the vegetation. Our results demonstrate the need to integrate finer scale approaches and primary data in ES assessments of peri-urban areas.

Highlights

- Peri-urban areas are pivotal in providing ecosystem services (ES) to people.
- It is unclear how accurate commonly-used ES mapping methods are in peri-urban regions
- We compared the congruence of ES bundles with plant composition data for the peri-urban region of Paris
- ES mapping methods poorly represented variation in vegetation composition.
- There is a need for simple though sufficiently fine-scaled methods to map ES in peri-urban contexts.

Keywords: vegetation, botanical survey, cluster analysis, green infrastructure, peri-urban ecosystems.

1. Introduction

Ecosystem services (ES) mapping and assessment is increasingly common, in line with science-policy initiatives globally, like IPBES (http://www.ipbes.net/), and regionally, like MAES for the European Union (http://biodiversity.europa.eu/maes). Land cover data are a common data source for ES assessments (e.g. Andrew et al., 2015; Crossman et al, 2013; Egoh et al, 2012; Malinga et al., 2015; Martínez-Harms and Balvanera, 2012). However, such data are coarse representations of the actual vegetation composition and often do not represent the intensity of land use or land management. Indeed, Eigenbrod et al. (2010; for England) and Schulp et al. (2014a; for Europe) revealed that estimates by ES mapping methods based on land cover data exhibited relatively large uncertainty and variability. Another source of uncertainty and error originates from the coarse resolution of land cover data typically used to map ES. Such data do not represent fine scale heterogeneity in land cover, resulting in potential errors in ES estimates in heterogeneous areas, such as peri-urban regions (Van der Biest et al., 2015).

Other approaches assess ES based on the functional relationship between the traits of plant communities and the provision of ES (Díaz et al., 2007), hence overcoming some of the uncertainties of methods primarily using land cover. However, studies linking plant functional traits and ES are mostly local, given the data-intensity and complexity of such models. This challenges the transposition of trait-based methods to more complex landscapes, and to larger spatial scales (Lavorel et al., 2011). Some studies have been conducted at landscape scales (Crouzat et al., 2015; Homolová et al., 2014) but mostly in areas valuable from a biodiversity conservation perspective, which are often remote, such as alpine grasslands.

The limitations of both types of approaches to map ES are especially important in peri-urban areas. While often less relevant from a biodiversity conservation perspective, peri-urban regions are pivotal in the provision of ES to people, given that over half of global population lives in cities (United Nations, 2015). Peri-urban areas still represent considerable amounts of non-built up land that provide ES (Huang et al., 2011; McGregor and Simon, 2012) although some need to be more closely delivered to beneficiaries (e.g. recreation and air quality regulation), than others (e.g. carbon sequestration) (Casado-Arzuaga et al., 2013; Vejre, Jensen, and Thorsen, 2010; Verhagen et al., 2016a). Here, the relations between vegetation and ES are little studied, although ES have received much attention in an urban context (e.g. Alam, Dupras, and Messier, 2016; Haase et al., 2014; La Rosa, Spyra, and Inostroza, 2016). Studies that map ES in urban areas have mostly used general vegetation covers (Larondelle and Haase, 2013; Tratalos et al., 2007), with some using more detailed vegetation types in ES assessment (Derkzen, van Teeffelen, and Verburg, 2015; Holt et al., 2015; Lehmann et al. 2014). Given that peri-urban landscapes, at least in Europe, are typically heterogeneous in land cover (Couch, Leontidou, and Petschel-Held, 2007; Hoggart, 2016), mapping approaches using coarse resolution data may have relatively large errors in ES estimates (Van der Biest et al., 2015). Indeed, Malinga et al. (2015) conclude that while most ES mapping studies concern the municipal scale and fine grain size (1 hectare), most of these studies used generic data and models for calculating ES generation at these finer grain sizes. Therefore, it is important to improve the understanding of the relations between actual vegetation composition, land cover classification and ES assessments in the peri-urban context, at fine grain size. Even though vegetation composition alone cannot be considered an optimal proxy for ES provision either, it is a relevant indicator of degree to which ES mapping methods based on land cover data capture actual variation in vegetation at the local scale.

This research aims at assessing the congruence between two commonly used ES mapping methods based on land use data, and plant composition for a peri-urban context. We considered (1) proxy-based models (*sensu* Lavorel et al., 2017: "*models that relate ES indicators to land or marine cover, abiotic and possibly biotic variables by way of calibrated empirical relationships or expert knowledge*") – namely the ES assessment matrix by Burkhard et al. (2012) – and (2) phenomenological models (*sensu* Lavorel et al., 2017: "*based on an understanding of biological mechanisms underpinning ES supply* […] *They assume, but do not represent explicitly, a mechanistic relationship between elements of the landscape, considered as ES* [Providers] *units, and the provisioning of ES*") for seven ES: air quality regulation, global climate regulation, flood protection, pollination, wild food provision, erosion regulation, and recreation. We used Paris, France, as a case study. The peri-urban green space of this large metropolis serves a large population and is heterogeneous in landscape character and therefore likely to have spatial variation in ES provision. For 252 sites, we assessed the plant composition from botanical surveys. For these sites ES were quantified using both types of ES modelling approaches. Next, sites were clustered three times, based on either their plant composition or their ES bundle, as estimated by the two model types, and the degree of congruence among the three sets of clusters was determined. By doing so, our study aims at indicating the degree to which land use and vegetation heterogeneity are reflected in the ES bundles estimated by different modelling approaches. Note that this focus on ES bundles is different from earlier studies, which compared individual ES models to each other (Schulp et al., 2014a) and to primary data (Eigenbrod et al., 2010). The ES bundle approach is relevant because policies often aim at protecting the overall level of ecosystem service provision rather than or additional to the provision of individual services (Schulp et al., 2014a). Moreover, Eigenbrod et al. and Schulp et al. focussed on national to continental scales using 1-10km resolution. Here we focus on the smaller-scale peri-urban context, using data as fine-grained as possible (0.1-1km resolution). This is valuable because correct assessment of ES in peri-urban regions is important for designing effective green infrastructure networks to support human well-being – a key objective under the EU Strategy on Green Infrastructure (European commission, 2013), and which can also support Sustainable Development Goal 11: sustainable cities and communities (United Nations 2015b).

2. Materials & methods

This section comprises four parts: we first explain the Parisian peri-urban context and the selected study areas (2.1.). Second, we describe the plant data and the survey methods used to collect them (2.2.). Third, we specify the ES mapping models (2.3.), and last, we present how cluster analysis is used to evaluate the congruence between ES bundles and plant data (2.4.).

2.1. Study area

We focus our research on the peri-urban interface of Paris, France, which forms the intermediate between rural and urban land. It is designated as a “green belt” by the Île-de-France Regional Council. This so-called green belt is approximately 20km wide and covers 2662 km^2 around the urban core of Paris (Fig. 1). Contrary to other green belts in Europe such as London or Berlin (Alexandre, 2013; Amati, 2008), the concept of the Île-de-France green belt has never been strictly embedded or enforced in land cover management and spatial planning. As it has been discussed elsewhere (Allen, 2003; Simon, 2008), drawing limits to peri-urban areas is not an easy task as land use is characterised by hybridization and heterogeneity. Here, the area consists of a patchwork of land cover types where woodland, cropland, semi-natural land and urban land coexist and where almost four million people live (Roussel, 2016). For these people, as well as for inhabitants of the city of Paris itself, the region provides important benefits for well-being through ES provision. The pressure on non-built-up areas has recently risen again with the Grand Paris project for which an extended transportation network is planned (Belkind, 2013) as well as new economic and housing developments (Gallez, 2014). In this context, which is applicable to many cities worldwide, it is even more relevant to understand the distribution and composition of peri-urban vegetation and the ES it supplies.

In order to apprehend the spatial complexity of the green belt area, three study areas were selected to represent the landscape diversity of the area (see squares in Fig. 1, detailed in Fig. 2).

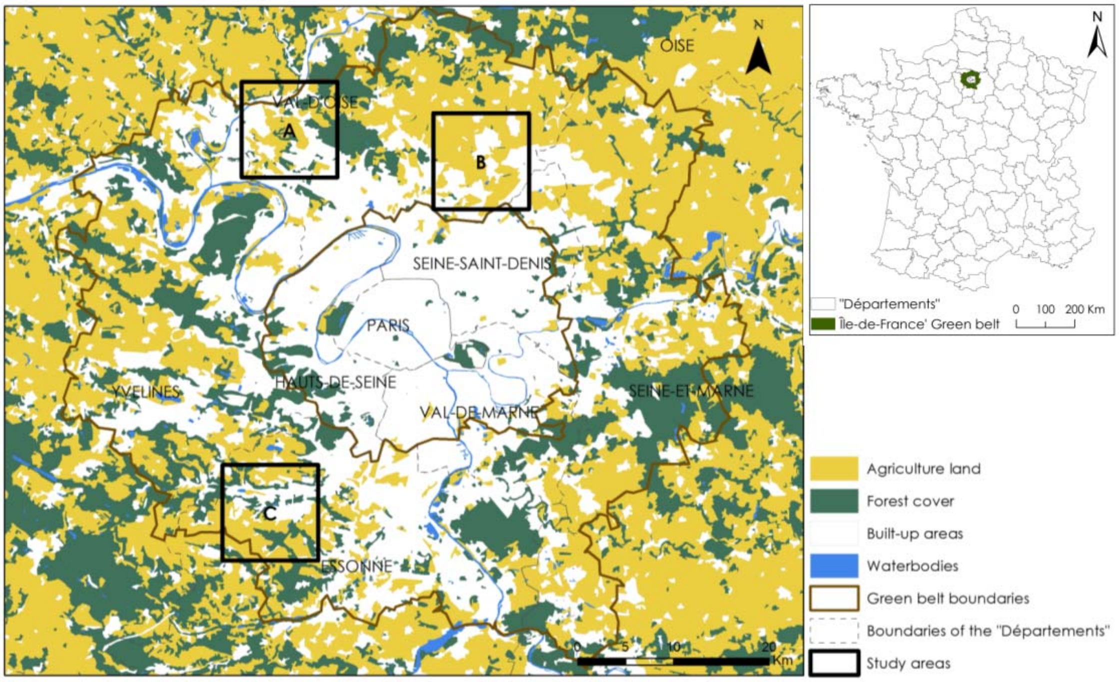

Figure 1: Situation of the green belt in the context of Paris urban area (modified from the French National Geographical Institute map of the region at 1/100 000 – 2011). The black squares indicate the location of the study areas (see Fig. 2). The map at the right depicts the location of the green belt in France.

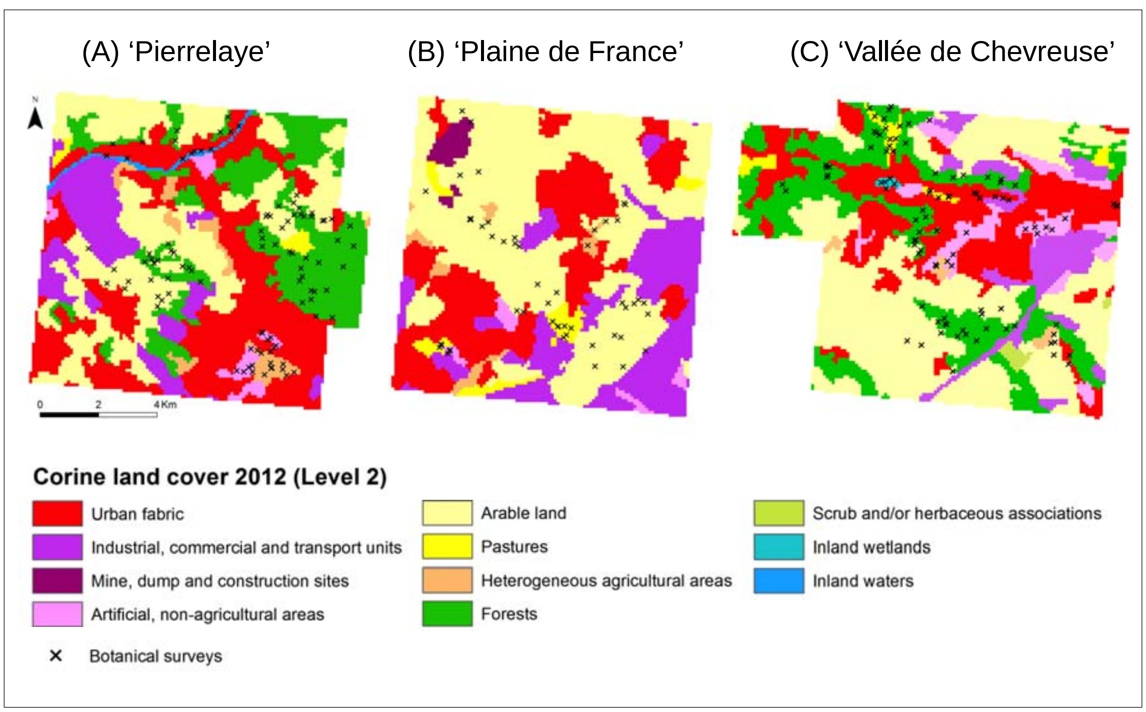

Figure 2: CORINE land cover maps (year 2012) of the three study areas (A-B-C, Fig 1).. Study area A was extended to be able to represent the forest context; Study area C was extended to be able to represent the nature reserve context. NB: for more clarity, we used here level 2 of CORINE Land Cover classification data, instead of level 3 which is commonly used in ES models, proxy-based and phenomenological.

The first area (Fig. 2A Pierrelaye) is situated in the north-west of Paris urban area and includes the isolated agricultural land around the small city of Pierrelaye, the state forest of Montmorency and the valley of the Oise river. The area presents a combination of the different land cover/land use types in a peri-urban context. Vegetation patches vary in size, form and are scattered throughout the area. The eastern extension intends to catch forests, which are a recurrent land cover in the context of Paris (see Fig. 1).

The second area (Fig. 2B Plaine de France) lays in the north-east part of the Paris urban area around the city of Goussainville, on an agricultural plateau called "Plaine de France", in proximity of the International Airport Charles de Gaulle. The area represents intensive agricultural lands under urban pressure. Most of the plateau is used for crop production, with semi-natural vegetation being scarce. The Paris metropolis has been extending here for the last forty years. Some grassland and woodland remain on the narrow and gentle slopes of a few small brook valleys.

The last area (Fig 2C Vallée de Chevreuse) is centred on the valley of the river Yvette, in the south-west of Paris urban area, also known as "Vallée de Chevreuse". The topography and related land cover and land use is typical of the Île-de-France valleys: slopes can be steep and are mostly covered by trees, while the valley bottom is occupied by built-up area with some riparian habitat along the river. The valley is notably known for nature protection measures, such as a nature reserve on the western part of the area, hence, the extension in this direction.

2.2. PLANT DATA

We took advantage of existing plant data from a parallel research project, in the form of botanical surveys that identify plant species composition in each of the three areas. The site selection also intends to catch the landscape diversity representative of the areas. It is composed of 252 botanical field surveys 51 for the Plaine de France, 100 for the Vallée de Chevreuse, 101 for the Pierrelaye surroundings (Fig. 2). The smaller number of surveys for Plaine de France is a consequence of the limited semi-natural vegetation cover in the area, which is dominated by intensive cropland. . The selection process corresponds to a stratified sampling method (Godron, 2012; Kent, 2012): botanical surveys are representatives of landscape elements inside local landscapes (1x1km) composing the four major types of landscapes (10x10km). For each of the three study areas, local landscapes were defined in reference to topography and main spatial occupation (urban, rural, forest etc.) given by IGN topographical maps at 1/25000. Landscape elements were based on vegetation cover types on aerial photographs (France 'Institut Géographique National', 2012, 0.5m resolution) inside each local landscape type: woodland, shrub, herbaceous. The number of botanical surveys is proportional to the vegetation cover types observed in a landscape (e.g. if 10% of the area is woodland and 20% is herbaceous, then we selected twice as many sites in grassland than in woodland). However, the complexity of landscapes and vegetation on the ground made it necessary to choose the sites in detail along the way. No minimum distance between the sites was required as long as land cover was different (it is also true for forested areas as they are interspersed with trails or small paths that interrupt forest cover,)

Botanical surveys were conducted according to the minimum area approach (Kent, 2012): starting from the centre of a plot/landscape element, we identified all the species until no new species were found, or until we reached the plot/landscape element limit. Surveys included all plant types: herbaceous, wooden (trees and shrubs), annual and perennial. Each site was surveyed once and all surveys were conducted between May and July, in the years 2014 and 2015, in order to keep the highest seasonal homogeneity in the flora.

In total, 520 plant species were identified among the 252 sites. For statistical analysis we considered only species that were encountered in at least five sites (i.e. in more than 2% of the sites), yielding 288 species. Species and sites were put together in a contingency table indicating the presence or the absence of species per site.

2.3. ES MAPPING

To map ES for the study area, we employed two types of methods: first, as a baseline, ES were mapped with a proxy-based approach, using an ES assessment matrix (Burkard et al., 2012). Second, phenomenological ES models were applied to the study areas to see whether they would refine the estimate of ES supply as compared to the matrix method. Seven services were considered: air quality regulation, global climate regulation, flood protection, pollination, wild food provision, erosion regulation and recreation. Selection of services was based on their relevance in a peri-urban context, and on availability of phenomenological models to map them.

2.3.1. Proxy-based model

In this approach, described by Burkhard et al. (2012), the mapping of ES is based on the level 3 of CORINE Land cover types at a 100m resolution. Each land cover type is given a score per ES on a scale between 0, no relevance, and 5, very high relevance. The method is widely applied (e.g. Baral et al., 2013; Burkhard et al., 2015; Sohel, Ahmed Mukul, and Burkhard, 2015), is easy to use, but due to its simplicity it has its shortcomings when there is an interest to map ES more accurately for specific study areas (Bagstad et al., 2014; Vallés-Planells et al., 2014; Van der Biest et al., 2015). Here, we consider the use of the ES matrix as a first step in ES assessment in a peri-urban context.

2.3.2.Phenomenological models

All the ES values for the 252 sites were extracted using GIS software. All ES models were developed at EU level referring to EU level data. We directly applied it to our study areas unless it was possible to modify the land cover data inside the GIS model. This was the case for the carbon sequestration model, which has been adapted to our study area, that is the Île-de-France Region, in order to obtain more precise values. The initial models were built on 1km resolution land cover types. We were able to run it with CORINE land cover resolution (100x100m). The erosion regulation model we used was already based on CORINE land cover classes. Even though data have different resolutions, the purpose is to create the most precise bundle of ES.

Air quality regulation was expressed through nitrogen dioxide (NO2) removal by urban vegetation in ton/ha/year. NO2 is one of the main pollutants emitted by transports, industries and households. The indicator is calculated as the product of dry deposition velocity and pollutant concentration at a 100m resolution (Lavalle et al., 2015).

Global climate regulation is quantified through carbon sequestration (Schulp et al., 2008). The results are expressed in tonnes per hectares and have been adapted for a 100m resolution, using the 2012 CORINE Land Cover map.

Flood protection. We used the map of Stürck et al. (2014), which defines a flood regulation supply index between 0 and 1, at one kilometre resolution for the European union. The index integrates environmental variables such as river catchments types and zones, precipitation types, land use information, water-holding capacity of the soil (WHC). It also takes into account the hydrological spatial modelling of rainfall runoff optimized for the analysis of the hydrological impact of land use and climate changes in large river basins.

Pollination. We used the probability that a location is visited by pollinators, measured in percentage, which takes into account the proximity of bee habitats as defined by Schulp et al. (2014b).

Wild food provision. We took into account vascular plants related to human consumption as described by Schulp et al. (2014c). They measured the number of edible plants and berries at 1km resolution. 81 edible plants and berries have been listed as used in more than four European countries while 592 plants have actually been identified as edible in total. Similar to the other models large scale available data sources of vascular plant occurrence were used. Our own plant inventories were not used for this assessment to ensure an independent comparison is possible afterwards.

Erosion regulation. We considered the mean cover management factor "C" per land-cover type, as defined by Panagos et al. (2015). It is the weighted average of the soil loss ratio per land cover types, as defined by CORINE Land Cover. It is expressed between 0 and 1, 1 being considered as a reference condition: down hill tillage with no vegetation.

Recreation: Assessment of this ES is described in Verhagen et al. (2016a), which considers urban leisure opportunities. Maps are based on land cover data, distance to coasts, forest location characteristic and agricultural landscape structure. It has to be noticed that the proxy-based ES matrix we refer to considers "Recreation and aesthetic values" together (table 2).

2.4. STATISTICAL ANALYSIS

2.4.1. Cluster analysis

To assess the level of congruence between plant composition and ES models, we applied a hierarchical cluster analysis to our data (Kent, 2012, Milligan and Cooper, 1987). All 252 sites were clustered three times: once based on the plant composition data, and twice based on (standardised) ES values - once for the proxy-based ES values and once for the phenomenological modelling ES values. This approach allows assessing whether sites with similar plant compositions are also estimated to have similar ES bundles. As the ES are expressed in different units, all ES values were standardised using a Z score approach (standardized value Z = (value - mean) / standard deviation) before clustering. The resulting values range between -1.5 and +1.5, with zero indicating the mean value for that ES over all sites, according to the model in question (proxy or phenomenological). Cluster analysis was conducted using Ward's method and square Euclidean distance (SPSS® Statistics 22). Because of its binary dimension, we selected a specific binary data option in SPSS for the plant table. Dendrograms were used to decide on a meaningful number of clusters per cluster analysis focusing on the distance between two clustering nodes (Norusis and SPSS, 2011). In case of ties (i.e. the absence of clear breakpoints) we chose the clustering that could be explained best based on the biotic properties of the sites (land cover, vegetation type).

2.4.2 Cluster comparison

In order to compare how ES evaluation methods and plant composition correspond or differ, we used two approaches. First we determined the degree of similarity among clusters in terms of number of sites shared, using the Jaccard similarity index (Kaufman and Rousseeuw, 2009). This index gives each pair of clusters a value between 0 and 1, 1 indicating 100% similarity (i.e. both clusters consist of the same sites; see table 4). Second, we compared the bundle of ES values supplied by each of the clusters, according to the two ES methods: for each cluster and each ES, we calculated the average ES supply, once based on the phenomenological modelling values, and once based on proxy values. The resulting two bundles of average ES supply per cluster were visualised in spider diagrams, to allow an assessment of the congruence of the methods within clusters, and to compare variation among clusters.

3. Results

This section is divided in three parts: we first present the clustering per dataset (3.1), followed by assessing cluster congruence (3.2), for all three datasets (3.2.1) and pair-wise (3.2.2-3.2.4). Third, we present and compare the ES bundles as predicted by the two ES models (3.3), for the three study areas (3.3.1) and for the clusters (3.3.2).

3.1 CLUSTERING RESULTS

3.1.1. Plant clusters

According to the dendrogram for plants (see suppl. material), there was little difference between 4, 6 or 9 plant clusters. From an ecological perspective, the 6 clusters matched well with typical vegetation types like mature timber forests, wet grasslands, fallows etc., for which 6 clusters were used in subsequent analyses. Cluster descriptions (Table 1) were based on plant composition in relation with field photographs of the sites in each cluster, allowing us to define types of vegetation.

Table 1: Types of vegetation according to the cluster analysis of plant species composition per sites.

Cluster	Species presence above 67%	Species unique to the group	Number of sites
1 – chestnut forest vegetation	*Castanea sativa* (Chestnut) *Rubus cæsius* (Dewberry) *Lonicera periclymenum* (Common Honeysuckle)	*Betula pubescens* (Downy Birch)	57
2 – low/high shrub vegetation	*Urtica dioica* (Stinging Nettle) *Geum urbanum* (Wood Avens)	None	39
3 – grassland vegetation	*Dactylis glomerata* (Cocksfoot) *Plantago lanceolata* (Ribwort Plantain) *Urtica dioica* (Stinging Nettle) *Arrhenatherum elatius* (Tall Oat Grass) *Cirsium arvense* (Creeping Thistle)	None	39
4 – woodland vegetation	*Hedera helix* (Common Ivy) *Fraxinus excelsior* (Common Ash) *Geum urbanum* (Wood Avens) *Cratægus monogyna* (Common Hawthorn) *Acer pseudoplatanus* (Sycomore) *Corylus avellana* (Hazel) *Rubus cæsius* (Dewberry) *Urtica dioica* (Stinging Nettle)	*Potentilla indica* (Indian Strawberry)	54
5 – fallow or herbaceous paths vegetation in	*Bromus sterilis* (Sterile Brome) *Artemisia vulgaris* (Mugwort) *Cirsium arvense* (Creeping Thistle)	*Apera spica-venti* (Loose Silky-bent) *Reseda luteola* (Weld) *Brassica nigra* (Black Mustard)	50

cropland context		*Arenaria serpyllifolia* (Thyme-leaved Sandwort) *Lactuca virosa* (Blue lettuce) *Matricaria discoidea* (Pineappleweed)	
6 – wet grass or shrub vegetation	*Symphytum officinale* (Common Comfrey) *Lythrum salicaria* (Purple Loosestrife) *Urtica dioica* (Stinging Nettle) *Galium aparine* (Goose-grass) *Arrhenatherum elatius* (Tall Oat Grass) *Holcus lanatus* (Yorkshire Fog) *Epilobium hirsutum* (Great Hairy Willow-herb) *Filipendula ulmaria* (Meadow Sweet) *Scrophularia auriculata* (Water Figwort)	*Galium uliginosum* (Swamp Bedstraw)	13

The first cluster (57 sites) can be designated as Chestnut forests vegetation with few sites (4) being clearings in the forest. The second cluster (39 sites) refers to various types of low or high shrub formations in disturbed vegetation contexts dominated by *Urtica dioica*. The third cluster (39 sites) puts together grasslands where species from the Poacea family prevail (*Arrhenatherum elatius*, *Dactylis glomerata*). Some sites (6) are actually not grasslands but hedges or low shrubs along grasslands. The fourth cluster (54 sites) designates unmanaged woodlands in isolated, disturbed or private contexts. The fifth cluster (50 sites) relates to fallow or herbaceous path in croplands. This cluster contains most unique species (Table 1, third column). The sixth cluster (13 sites) refers to wet grass or shrub lands with typical hygrofilous species such as *Filipendula ulmaria*, *Scrophularia auriculata* or *Lythrum salicaria*.

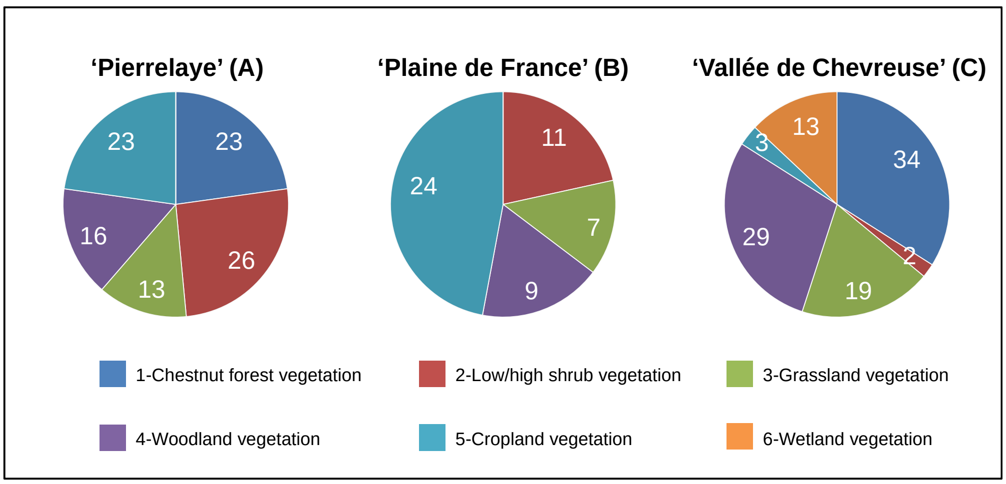

Figure 3: Distribution of plant clusters among sites of the three study areas. Values in the diagrams refer to numbers of sites.

The three study areas show three different profiles in terms of vegetation types (Figure 3). Study area A is the most balanced, plant clusters 1, 2 and 5 being present in the same proportions, ahead of plant clusters 3 and 4. This confirms the rationale for which the area was selected, i.e. for its landscape heterogeneity. Study area B is dominated by cluster 5 and cluster 1 is absent. It confirms the criteria of intensive agriculture to select this area. Plant cluster 6 is only found in study area C, confirming the hydrological criteria used to select it (valley). It also has the most wooded sites among all, clusters 1 and 4 are both most abundant in this area.

3.1.2. Proxy-based model clusters

Sites fell into two distinct clusters according to the proxy-based model estimates (Table 2). The first cluster comprised 169 sites, consisting of all sites that have land cover other than Broad-leaved forest. For almost all ES the capacity was low (0 or 1 out of 5, Table 2), except for Recreation, which had a medium capacity of 2.5. The second cluster only comprises sites classified as Broad-leaved forest (83 sites from areas A and C). This cluster shows high capacity for nearly all ES, except flood protection (Table 2).

Table 2: CORINE land cover types present in the study area, number of survey sites within each land cover type, and associated supply of the seven selected ES, as estimated using the ES matrix approach (Burkhard et al. 2012). Scores between 0 (no relevant capacity, no shading) and 5 (very high capacity, dark shading).

Cluster	Study area where cluster sites originates from	CORINE land cover types	Number of sites	Global climate regulation	Flood protection	Air quality regulation	Erosion regulation	Pollination	Wild foods	Recreation & aesthetic values
1 (169 sites)	A (37%) B (30%) C (33%)	Non-irrigated arable land	74	1	1	0	0	0	0	3
		Pastures	24	1	1	0	4	0	0	3
		Green urban areas	22	1	0	1	2	1	1	3
		Discontinuous urban fabric	21	0	0	0	0	0	1	0
		Complex cultivation patterns	13	1	1	0	0	0	0	2
		Land principally occupied by agriculture	4	2	1	1	3	0	3	2
		Inland marshes	4	2	4	0	0	0	0	0
		Water courses	3	0	5	0	0	0	0	4
		Industrial or commercial units	1	0	0	0	0	0	0	0
		Sport and leisure facilities	1	1	0	1	1	1	0	5
		Mineral extraction sites	1	0	0	0	0	0	0	0
		Road and rail networks and associated land	1	0	0	0	0	0	0	0
2 (83 sites)	A (47%) C (53%)	Broad leaved-forest	83	4	3	5	5	5	5	5

3.1.3. Phenomenological models clusters

Based on the phenomenological model results, sites were clustered into four groups (see suppl. material). For interpretation, phenomenological clusters are listed against CORINE land cover classes (Table 3).

Table 3: CORINE land cover for each sites of the four phenomenological clusters

CORINE land cover (2012)	Cluster 1 (66 sites)	Cluster 2 (88 sites)	Cluster 3 (51 sites)	Cluster 4 (47 sites)
Broad leaved-forest	0	34	3	46
Complex cultivation patterns	11	1	1	0
Discontinuous urban fabric	0	20	1	0
Green urban areas	2	18	2	0

Industrial or commercial units	1	0	0	0
Inland marshes	0	4	0	0
Land principally occupied by agriculture	2	0	2	0
Mineral extraction sites	0	1	0	0
Non-irrigated arable land	40	1	33	0
Pastures	10	5	8	1
Road and rail networks and associated land	0	1	0	0
Sport and leisure facilities	0	0	1	0
Water courses	0	3	0	0
Study area where cluster sites originate from	A (23%) B (74%) C (3%)	A (44%) B (2%) C (54%)	A (47%) C (53%)	A (49%) C (51%)

The first cluster comprises 66 sites representing a mix of agricultural land, mostly from the Plaine de France study area (B, Table 3). In this cluster ES provision is mostly below average, except for flood regulation and recreation, which is average (Fig 5). The second cluster comprises 88 sites. It is a mix between built-up areas (including green urban spaces) and forest. It has average supply of most of the ES, but higher erosion control and lower recreation capacity. The third cluster comprises 51 sites representing mixed agricultural land from Pierrelaye (A) and Vallée de Chevreuse (C), mostly non-irrigated arable land. The situation of the sites on the map (see figure 2) shows more proximity with broad-leaved forests and heterogeneous contexts than sites from cluster 1. Hence, recreation, wild food provision, pollination and air quality regulation are higher. The last cluster is mostly broad leaved-forest and comprises 47 sites mainly from Pierrelaye (A) and Vallée de Chevreuse (C). Most ES have high values, especially recreation, air quality regulation and carbon sequestration, the highest among all the clusters.

3.2 CONGRUENCE OF CLUSTERS

3.2.1 Generic patterns

Sites are rather scattered among the different clusters (Table 4), indicating that the three different methods have a low congruence. Most (though not full) congruence exists between Proxy-Cluster2, Pheno-Cluster4 and Plant-Cluster1, which are predominantly forested sites. Also Proxy-Cluster1, Pheno-Cluster1 and Plant-Cluster5 share a large proportion of sites, in agricultural land. Some clusters tend to be mutually exclusive: Pheno-Cluster1 has no similarity with Plant-Cluster1 or Plant-Cluster6, while Plant-Cluster5 and Pheno-Cluster4 exclude each other to a large extent, just as Plant-Cluster1 and Pheno-Cluster3.

Table 4: Number of sites shared by clusters (Jaccard index)

		Plant clusters						**Proxy-based clusters**	
		1 Chestnut forests vegetation	2 Low and high shrubs vegetation	3 Grasslands vegetation	4 Woodlands vegetation	5 Cropland vegetation	6 Wet grass/shrub lands vegetation	1 Other ES	2 Broad-leaved forest ES
Pheno. clusters	1 Agricultural land ES	0 (0)	18 (0.21)	9 (0.09)	11 (0.11)	28 (0.32)	0 (0)	66 (0.39)	0 (0)
	2 Mixed built-up/forest areas ES	30 (0.26)	11 (0.09)	8 (0.07)	27 (0.23)	5 (0.04)	7 (0.07)	54 (0.27)	34 (0.25)

	3 Heterogeneous agriculture land ES	1 (0.01)	6 (0.07)	18 (0.25)	8 (0.08)	15 (0.17)	3 (0.05)	48 (0.28)	3 (0.02)
	4 Broad-leaved forest ES	26 (0.33)	4 (0.05)	4 (0.05)	8 (0.09)	2 (0.02)	3 (0.05)	1 (0.001)	46 (0.55)
Proxy-based clusters	1 Other land cover ES	13 (0.06)	29 (0.16)	34 (0.20)	36 (0.19)	47 (0.27)	10 (0.06)		
	2 Broad-leaved forest ES	44 (0.46)	10 (0.09)	5 (0.04)	18 (0.15)	3 (0.02)	3 (0.03)		

3.2.2. Pheno. clusters vs proxy-based clusters

The similarity is seemingly low between proxy-based clusters and phenomenological clusters, (Table 4, rightmost two columns, Jaccard indices), but this is in part explained by the difference in cluster number and size: Pheno-Cluster1 is entirely embedded in Proxy-Cluster1 (66 sites), and Pheno-Cluster3 is predominantly embedded in Proxy-Cluster1 too (48 sites out of 51); Pheno-Cluster4 is almost entirely embedded in Proxy-Cluster2 (46 sites out of 47); only Pheno-Cluster 2 is clearly divided over both Proxy-Clusters.

3.2.3. Proxy-based clusters vs plant clusters

In terms of land use, Proxy-Cluster2 represents a homogeneous land use class (broad-leaved forests), but its sites are nevertheless rather dispersed when clustered according to their plant composition (Table 4, bottom row). This indicates that on the ground plant composition is more nuanced than reflected by the proxy-based ES. Moreover, 49 sites that do have a plant composition representative of forests (Plant-Cluster1, 13 sites, and Plant-Cluster 4, 36 sites) are not grouped with this forest-related ES cluster, but with Proxy-Cluster1. Hence, the congruence between plant composition and ES bundles according to Proxy-based models is relatively low.

3.2.4. Pheno. clusters vs plant clusters

The similarity between plant clusters and phenomenological model clusters is also low given the Jaccard values (Table 4). However, the way sites are predominantly distributed appears logical. For instance, Pheno-Cluster4 (Broad-laved forest) is mostly composed with elements from Plant-Cluster1 (Chestnut forests vegetation). Plant-Cluster1 also shares sites with Pheno-Cluster2 (mixed urban/forest). And, Pheno-cluster3 (ES from Heterogeneous agriculture land) is composed of sites from Plant-Cluster3 (grassland vegetation) and Plant-Cluster5 (Cropland vegetation). Despite this overall congruence, it is not uniform and a certain level of scatter remains in all clusters, showing that heterogeneity in the landscape, as represented by plant composition, is not fully reflected in the ES results from phenomenological models either.

3.3 COMPARISON OF ES MODEL ESTIMATES

3.3.1. ES levels in study areas

As the three study areas were chosen in reference to specific types of landscapes (Fig. 1), it is valuable to see whether the ES bundles estimated by the two model types reflect this difference, too. Fig. 4 depicts the mean ES supply for the sites in each of the study areas for both ES model types.

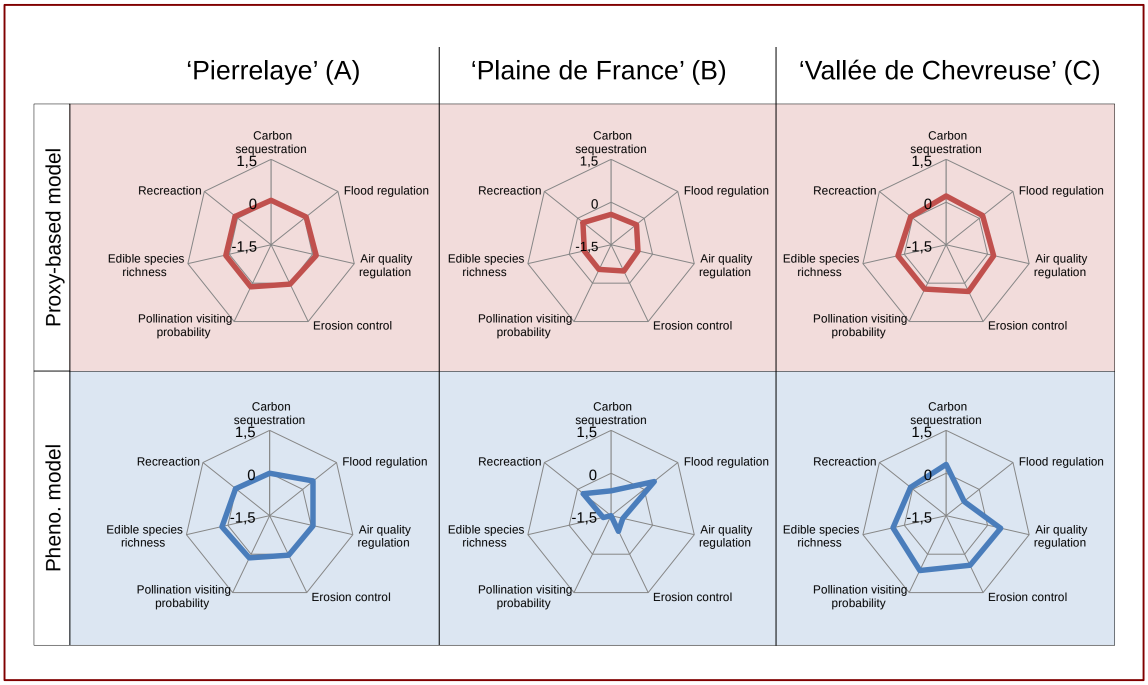

Figure 4: Mean ES supply in the sites within each of the study areas A (Pierrelaye), B (Plaine de France), C (Vallée de Chevreuse) according to the proxy-based model and the phenomenological model (normalized values: zero indicates the average value for a service within the study area is equal to the average value for that service among all sites. Negative (positive) values indicate the average value within the study area scores below (above) the overall average for that ES).

For proxy-based models, the round shape of the spider diagrams for all three study areas suggests the areas do not differ in the bundle of services provided, but only slightly in the amount of services provided. Study area B shows below average ES values, which is explained by the absence of sites classified as broad-leaved forest (even though, according to the vegetation data, 9% of the sites contains woodland vegetation (Fig. 3)). Although study areas A and C do possess broad-leaved forest sites, they show average supply because they are also a mix of several land cover types.

For phenomenological models, the contrast is stronger between the three study areas. Also here study area B has the lowest overall ES provision compared to A and C, but the bundles are not as balanced as with proxy-based model. Study areas A and B stand out in terms of flood regulation, while this service is lower in area C, where edible species richness, pollinators visiting probability, erosion control, air quality regulation and carbon sequestration values are the highest.

3.3.2. ES levels in site clusters

To compare ES values of the two modelling approaches in all of the clusters, mean ES values were calculated over the sites per cluster, for each of the two modelling approaches (analogous to the mean ES values for the study areas in the previous section). (Fig. 5 panels a, b, c). Like with study areas, proxy-based model estimates (red lines) are consistently round in shape, suggesting all clusters provide the same ES bundle, just in different quantities. The phenomenological model estimates (blue lines) on the other hand, often show a different shape indicating that the models are more pronounced in what bundle of ES is provided by different clusters.

In terms of ES bundle size, models largely agree: e.g. when the Proxy-based model estimates a small bundle, so do the Pheno models (e.g. for Proxy-Cluster1 (Fig.5, panel a) and Pheno-Cluster1 (panel b)). The estimate for flood regulation deviates most often between the two model types (e.g. Proxy-Clusters 1 and 2, Pheno-Clusters 1 and 4, and Plant clusters 1, 2 and 5). Interestingly, the models estimate a much larger ES bundle in Plant-Cluster1 than in Plant-cluster4 while both clusters represent forested habitats. Plant-Cluster3 and 6 (grassland / shrubland vegetation types (panel c)) show overall disagreements in the size of ES bundles. In each case, phenomenological models estimated average to above average values for most services, whereas proxy-based methods mostly estimated below average values.

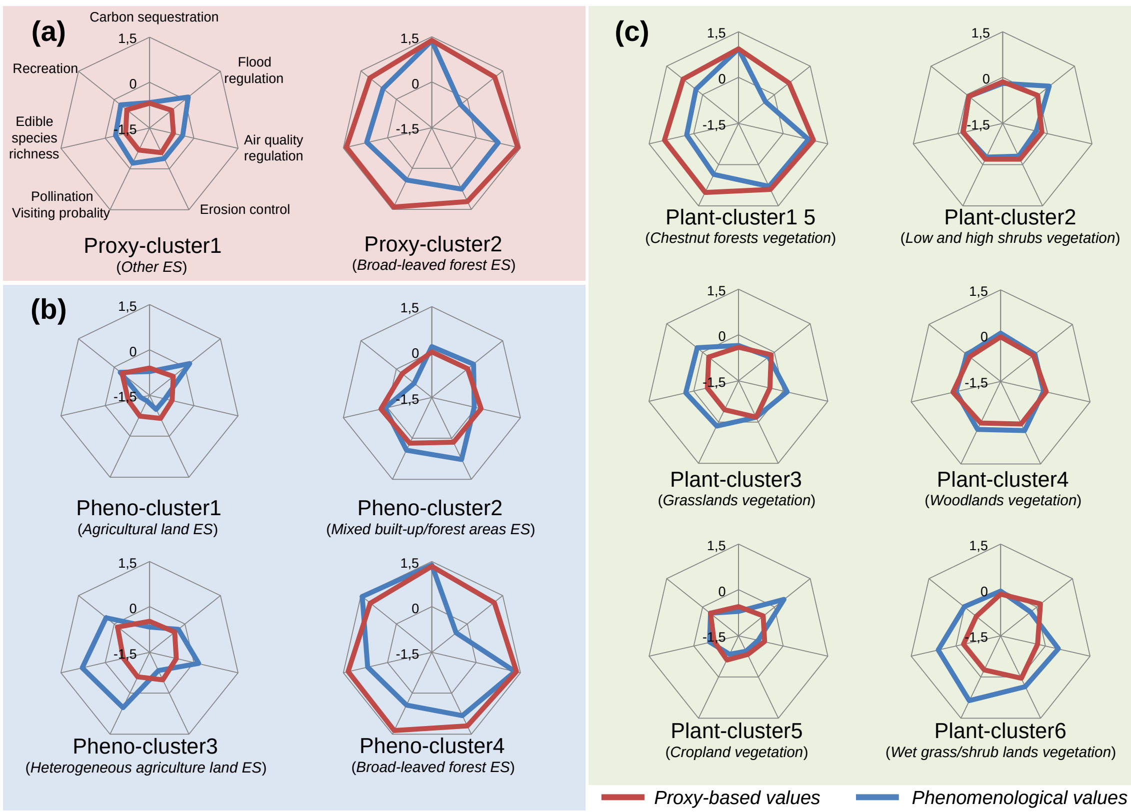

Figure 5: Mean ES values per cluster for proxy-based (a), phenomenological (b) and plant clustering (c). Both ES values are depicted (red = proxy-based model values; blue = phenomenological model values). A value of zero represents the mean value of the ES across all sites (n=252), as estimated by a particular method. Positive (negative) values indicate ES supply in the cluster is estimated to be higher (lower) than the mean.

5.Discussion

5.1. SPATIAL CONGRUENCE OF VEGETATION AND ES BUNDLES

The phenomenological model estimates of ES allowed a more nuanced clustering of sites (four clusters) than proxy-based model estimates of ES (two clusters). This indicates that phenomenological models created more pronounced bundles than the proxy-based model, which only highlighted broad-leaved forests as an important ES supply source (Figs. 4 & 5). Clustering based on plant composition was most detailed, yielding six distinct clusters. To decide on an appropriate number of clusters we looked for clear breakpoints in each of the dendrograms (Norusis and SPSS, 2011; see suppl. material). That is, we assessed the distance between two clustering nodes, in relation to the agglomeration values used to build the dendrogram. In our study, the clustering of sites was very different for each of the three datasets, with only a clear breakpoint for proxy-based model ES estimates. For the plant clusters and to a lesser degree, the phenomenological clusters, clear breakpoints were lacking. For these datasets, we therefore interpreted the dendrograms using expert knowledge of the study area, to decide on relevant cut-off values. There are also other ways of deciding on the optimal number of clusters to use, such as using equal distance between clusters, a similar number of sites per cluster etc. (Hennig et al. 2015). There is no consensus, however, on what is the best method (Dimitriadou *et al.*, 2002; Milligan and Cooper, 1985). Therefore, we decided to follow an approach that helped to best represent the specifics of the study area in the clusters identified.

There can be multiple explanations why sites that do have similar plant composition, do not cluster together based on ES bundles. First, while ES are provided by ecosystems and hence have a relation to plant composition (Díaz et al., 2007), these relations can be complex, and do not rely on vegetation alone (e.g. topography, soil type, hydrology, and management intensity also play a role) (De Groot, Wilson, and Boumans, 2002). For the phenomenological models, which take some of these additional factors into account, this can be one explanation for estimating different ES bundles on sites that are otherwise similar in land cover type. The flood regulation service shows this effect (see Figures 4 and 5). Besides land cover, the phenomenological flood regulation index also takes soil type and related water holding capacity as well as catchment type into account (Stürck et al., 2014). The seemingly contradicting results – forested sites (Plant-clusters1 and 4, Fig. 5) having lower mean flood regulation values than intensive cropland sites (Plant-cluster 5, Fig 5) – is a result of the spatial composition of the peri-urban landscape. On the contrary, carbon sequestration capacities were very similar between the ES models (Fig. 5). This was also shown in larger scale studies: land cover is a dominant input in estimating carbon sequestration capacity (Schulp et al., 2014a). Hence, for certain services the 'biophysical realism gap' (*sensu* Lavorel et al. 2017) seemed to increase indeed with the use of proxy-based models compared to the more detailed phenomenological models.

Second, the land cover data used in the ES models mismatched with the actual, on the ground plant composition. Indeed, the proxy-based model cluster that represented broad-leaved forest sites (Proxy-Cluster2) did also include non-forest sites and failed to include actual forest sites (Table 4). Such mismatch can be due to a lack of thematic resolution (the number of land cover classes) in the land cover data, or due to coarser spatial resolution of the land cover data. Eigenbrod et al. (2010) detected such a mismatch between primary data and proxy-based ES models using larger mapping units at a national scale. As we show here, the use of standard land cover data such as CORINE data faces limitations when applied in heterogeneous landscapes, including peri-urban contexts. With a minimum mapping unit of 1ha, smaller patches are not represented in CORINE, leading to noise in the ES estimated for such areas. For sites representing forested areas (both model types) and agricultural areas (phenomenological models) the congruence with plant composition was relatively high (Table 4). This can be explained as the forest and agricultural patches are typically larger, increasing the chances of being correctly classified in generic land use data sets. On the contrary, the cluster analysis of plant composition revealed specific contexts such as herbaceous paths/fallow vegetation or small woodland/shrub vegetation patches in broader intensive cropland areas that showed a low congruence with ES bundles (Table 4). As a consequence, in homogenous intensive cropland landscapes, both proxy-based and phenomenological methods showed low ES supply (Fig. 4). However, those small landscapes elements do enhance heterogeneity in the landscape, so we could expect some ES supply too (e.g. wild food provision, pollination or erosion regulation) as it has been shown elsewhere (Björklund, Limburg, & Rydberg, 1999; Swift, Izac, & van Noordwijk, 2004,Verhagen et al., 2016b). Hence, for studies taking place at smaller and fine-grained scale, readily-available proxy-based methods and land use data without further contextualisation are likely to perform poorly, and it may be important to use land cover data with appropriate detail (e.g. Derkzen et al., 2015). Note however that, as pointed out by Gómez-Baggethun and Barton (2013), the valuation of ES can serve different urban planning contexts (e.g. awareness raising, accounting, priority-setting, instrument design), which may require different levels of spatial detail and accuracy. The (un)suitability of ES mapping tools for the peri-urban context will have to be evaluated based on the context in which it will be applied and for what purpose.

5.2. ES MAPS AND PRIMARY VEGETATION DATA

The use of primary data can improve ES mapping methods and ultimately guide urban policies (Maes et al., 2012). While we did not infer ES from plant composition, we did assess the congruence between estimated ES bundles and vegetation types, focussing on spontaneous flora. The data we used were collected for the purpose of understanding the composition of semi-natural/spontaneous vegetation in peri-urban regions. By following a stratified sampling method based on a nested landscape approach, these data are representative of the semi natural vegetation land cover in Paris peri-urban area. Of course other types of vegetation cover such as planted green spaces (Derkzen et al., 2015) or crops (Swift, Izac, and van Noordwijk, 2004) also provide ES. However, we were not able to convey such data here.

We used cluster analysis to assess the congruence in the spatial distribution of ES bundles compared to vegetation types. The clustering based on plant data did result in the grouping of sites, which could be considered to have different vegetation types. As a step forward, one could consider using finer vegetation maps to assess this congruence or even direct ES quantification. For a different purpose (biodiversity conservation) using different methods (phytosociology), the French National Botanical Conservatory for the Parisian Basin (CBNBP) carried out a study of the vegetation throughout the entire Île-de-France region, mixing botanical surveys and extrapolations from aerial photographs analysis (Ferreira et al., 2015). About 100 communities have been mapped at 1/10 000 scale to help decision making in conservation projects. Trait-based methods can be used to evaluate ES supply at such refined scales (Díaz et al., 2007). However, the data intensity of those approaches (Lavorel et al., 2011) is still a limitation. Moreover, as the work aimed at biodiversity it ignored some more common vegetation types: 97 of our 252 sites (38%) are not mapped as plant communities by the CBNBP. This shows that even when detailed vegetation databases exist, they may still not be suitable for landscape-scale ES quantification. Hence, for ES assessments in landscapes such as (peri-) urban areas where smaller fractions of vegetation matter for ES provision, a land cover map with intermediate level of spatial resolution and thematic (vegetation type) detail is relevant.

6. Conclusions

Peri-urban landscapes are often pivotal in ES provision to the urban community (e.g. Gómez-Baggethun and Barton, 2013). To understand ES provision, mapping ES at such fine-grained, smaller scale level is frequently conducted (Malinga et al., 2015), but often using generic data and methods, especially to quantify several ES. Using seven ES, we have demonstrated that ES bundles estimated by such methods using generic data have low congruence with actual on-the-ground vegetation data, in particular in areas where the land use is heterogeneous. Because they integrate some biological mechanisms, phenomenological models showed more refined bundles of ES than proxy-based methods, for which they may be a better indicator of ES bundles in peri-urban areas. However, while large vegetation covers such as forests are correctly reflected, the mismatch is still considerable for finer vegetation cover types such as grasslands or small woodlands. With the increasing attention for resilient and healthy cities (UN, 2015), it is important to realise that readily available land cover data and ES mapping methods may not adequately capture the spatial and thematic (vegetation) detail relevant in peri-urban regions. The purpose of ES assessments in peri-urban and urban planning determines the degree of detail necessary. However, investments in mapping (peri-)urban green space in more detail, and using ES quantification methods that adequately reflect the heterogeneity present, may well be offset by the returns in human well-being from a more effective green space management.

Supplementary material

The following data are presented as supplementary material:

- Synthesis table on phenomenological ES models
- Dendrograms from the three cluster analysis : Plant data, Proxy-based ES values, Phenomenological ES models

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