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« Ecosystem service provisioning in the Grand-Est, France»

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Ecosystem service provisioning in the Grand-Est, France

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TECHNICAL SUMMARY

Ecosystem services are at the forefront of ecosystem management, and are a featured component of each research themes of the *Institut National de Recherche pour l'Agriculture, l'Alimentation, et l'Environnement* (INRAE). The national research program *Transition en Territoires de l'Agriculture, l'Alimentation et l'Environnement* (TETRAE) represents INRAE's long-term commitment to the sustainable management of agricultural, ecological, and urban environments. The project *Perceptions et valorisation des services écosystémiques en forêt* (PERCEVAL) is funded under the TETRAE program. Specifically, it seeks to assess potential markets for biodiversity and ecosystem services in forests in the Grand-Est region of France, and to develop a digital platform where economic partners and local stakeholders can access its findings to better inform their management decisions. In this document, we provide a baseline database of the supply or provisioning of ecosystem services in the Grand-Est region of France.

We estimate a set of eighteen indicators of seven ecosystem services, which include agriculture production potential, biodiversity, aboveground carbon storage, livestock grazing potential, net ecosystem productivity, pollination potential, and soil loss by water erosion. Our analysis uses a mix of land use and land cover data, established relationships between ecosystem services and land use/reflectance data, and published maps of ecosystem service supply from the scientific literature. We use information regarding the locations of agriculture, cities, and forests as well as topography to understand some of the potential drivers of ecosystem service supply in the Grand Est, and measure the interactions – how a change in one service leads to a change in another – between ecosystem services considered in the study. In full transparency, we provide support documentation for our study. This includes metadata, code, and data for estimating ecosystem services in the Grand Est.

In general, our findings are consistent with the scientific literature and what we would expect given our models and the data used to estimate them. While we would not recommend interpreting our results as absolute point measurements of ecosystem service supply at specific locations, we do believe that they do a good job at showing where ecosystem services are being supplied in the Grand Est. We discuss our results in the context of ecosystem management in Grand Est – specifically the importance of forests in the region – and how they fit into the broader question of what should be provided from the perspective of society. Finally, we provide a discussion of the limitations of our study.

Keywords: ecosystem services, GIS, Grand Est, interactions

JEL codes: C80, Q57, Y10

1. INTRODUCTION

Ecosystem services have been at the forefront of ecosystem management since the publication of the Millennium Ecosystem Assessment (Bennett et al., 2009; Fisher et al., 2009; Vihervaara et al., 2010). Indeed, ecosystem services form an integral part of the research agenda at the *Institut National de Recherche pour l'Agriculture, l'Alimentation, et l'Environnement* (INRAE), finding their way into each of its six research themes (agroecology, biodiversity, bioeconomy, climate change and risks, food and global health, society and regional strategies).

The national research program *Transition en Territoires de l'Agriculture, l'Alimentation et l'Environnement* (TETRAE) represents INRAE's long-term commitment to the sustainable management of agricultural, ecological, and urban environments according to multiple human-environmental criteria[1]. The program, co-financed by INRAE and several French regions, aims to develop partnerships between researchers, industry, and local stakeholders to stimulate open-science research on the development of more sustainable practices in the fields of agriculture, food, and the environment relevant to each region in France.

The project *Perceptions et valorisation des services écosystémiques en forêt* (PERCEVAL) is funded under the TETRAE program, and specifically focuses on the measurement and valuation of ecosystem services in the Grand-Est region of France[2]. The goal of project PERCEVAL is to assess potential markets for biodiversity and ecosystem services in forests in the Grand-Est region of France and to develop a digital platform where economic partners (foresters, industry businesses) and local stakeholders (associations, NGOs) can easily access and understand our findings to better inform their management decisions. To achieve these goals, the project measures the preferences of individuals – forest users and non-users – in regard to the management of forest ecosystem services (demand) and potential markets for them, as well as the spatial mapping of ecosystem services provided in the region (supply) and the creation of an open-access digital platform to access the results of the project. This document addresses the supply side of PERCEVAL.

[1] https://www.tetrae.fr/
[2] https://msh-lorraine.fr/perceval/

The objective of this technical document is to provide a baseline database of the supply of ecosystem services in the Grand-Est region of France. The Grand-Est represents the fifth largest region in the country in terms of area (57,441 km^2), and the sixth largest in terms of population 5,561,287 inhabitants in 2021) (Figure 1). It is composed of ten departments (Alsace – Haut-Rhin and Bas-Rhin, Ardennes, Aube, Haute-Marne, Marne, Meurthe-et-Moselle, Meuse, Moselle, Vosges), uniquely positioned at the border with Belgium, Germany, Luxembourg, and Switzerland. The population is concentrated in five cities (Metz, Nancy, Mulhouse, Reims, and Strasbourg), though much of it is located in the eastern part of the region. In terms of industry, the region ranks fifth in terms of the number of businesses (12,806)[3] and top three in France in terms of production depending on the sector (e.g., automobiles, metals, electrical and plastic processing)[4]. Ecologically speaking, the region is home to 6 regional parks (15% of the total land area of the region) and 27 regional wildlife reserves. Forests and agriculture make up about 80% of the area of the region. Thus, the region represents a valuable economic and ecological resources for both the public and private sectors.

We estimate a set of eighteen indicators of seven ecosystem services including agriculture production potential, biodiversity, aboveground carbon storage, livestock grazing potential, net ecosystem productivity, pollination potential, and soil loss by water erosion. Our methods are based on a mix of land use and land cover data, established relationships between ecosystem service provisioning and land use/reflectance (Amoatey et al., 2018; Myeong et al., 2006; Ricketts et al., 2008; Yao et al., 2014), and published maps of ecosystem service supply (Maes et al., 2015; Panagos et al., 2020; Schulp et al., 2014; Spawn et al., 2020). We compare the supply of each ecosystem service to the spatial distribution of agriculture, cities, and forests, and local topography to understand some of the potential drivers of ecosystem service supply in the Grand Est (Figure 2). Finally, we measure the interactions (tradeoffs and synergies) between ecosystem services by calculating the pairwise Pearson correlation coefficients for all services in the study. A detailed analysis at the national level, including all of the regions in France, can be found in Shanafelt et al. (2023).

[3] https://www.insee.fr/fr/accueil
[4] https://uimm-lorraine.com/

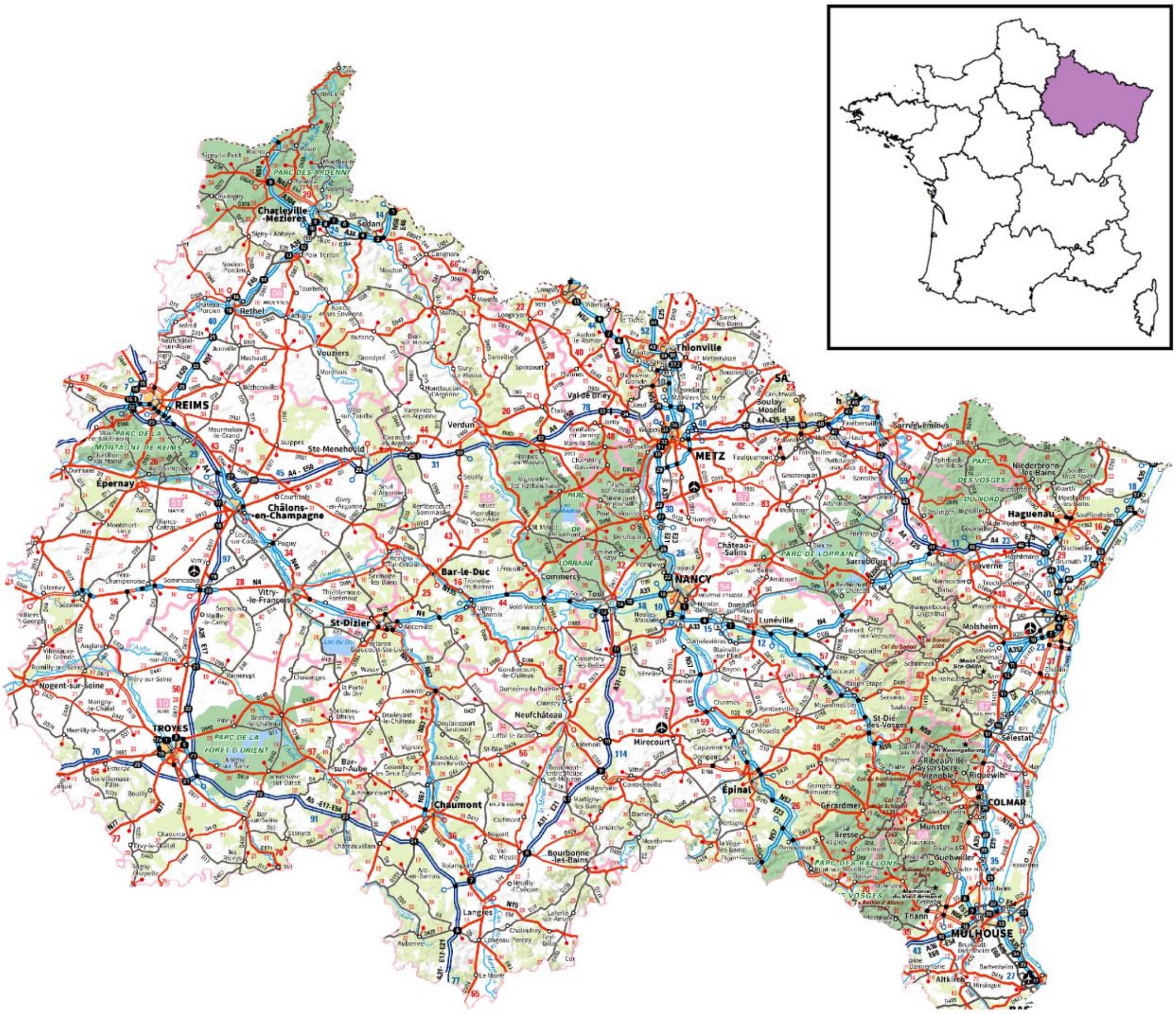

Figure 1. Map of the Grand Est region of France. Adapted from the Scan1000 product from the *Institut National de l'Information Géographique et Forestière* (IGN) (https://geoservices.ign.fr/scan1000).

The rest of the document is outlined as follows. In the next section, we outline our set of ecosystem services considered in the study and how we measure their interactions. The third section presents our primary findings, putting them in the context of the general land use and topology of the region. The final section discusses the results of our analysis, how they fit into the broader scientific literature of ecosystem services, what they mean for management of the Grand Est, and potential limitations of our study.

2. METHODS

Data

Data availability is one of the major limitations of the field (Bennett et al., 2009; Crossman et al., 2013; Egoh et al., 2012; Hou et al., 2013; Layke et al., 2012; Martinez-Harms and Balvanera, 2012). This study is no exception. We do not have access to local, plot-level data with which to test our hypotheses. We do, however, have access to coarser large-scale spatial data sets. We rely primarily on land use and land cover data to estimate ecosystem service provisioning. Land use and land cover data were downloaded from the French *Centre d'Etudes Spatiales de la Biosphère* (CESBIO) (Inglada et al., 2017) at the 10 m resolution. It includes seventeen land use types: annual summer crops; annual winter crops; broad-leaved forest; coniferous forest; natural grasslands; woody moorlands; continuous urban fabric; discontinuous urban fabric; industrial and commercial units; roads; bare rock; beaches, dunes, and sand; water bodies; glaciers and perpetual snow; intensive grasslands; orchards; and vineyards. Additionally, we use biodiversity data compiled from the National Inventory of Natural Heritage (INPN) (10 km resolution), reflectance data taken from the Google Earth Engine (<1 m resolution), and forest cover as provided by the European Commission Joint Research Centre (25 m resolution). References to download spatial data are located in Table 1 and can be accessed from the French governmental research data repository (https://doi.org/10.57745/NG3QSF).

Estimating the provisioning (supply) of ecosystem services

The scientific literature is abound with methodologies and frameworks to measure the supply of ecosystem services, including the InVEST model (Daily et al., 2009; Nelson et al., 2009), GUMBO (Boumans et al., 2002) and IMAGE (Schulp et al., 2012) frameworks, or the Soil Water Assessment Tool (SWAT) (Arnold et al., 1999; Lautenbach et al., 2013). We would direct the reader to reviews byCrossman et al. (2013), Egoh et al. (2012), Martinez-Harms and Balvanera (2012), and Schagner et al. (2013) for broad presentations of how to measuring individual ecosystem services. A summary spreadsheet of this literature can be found in the supplemental material of Shanafelt et al. (2023).

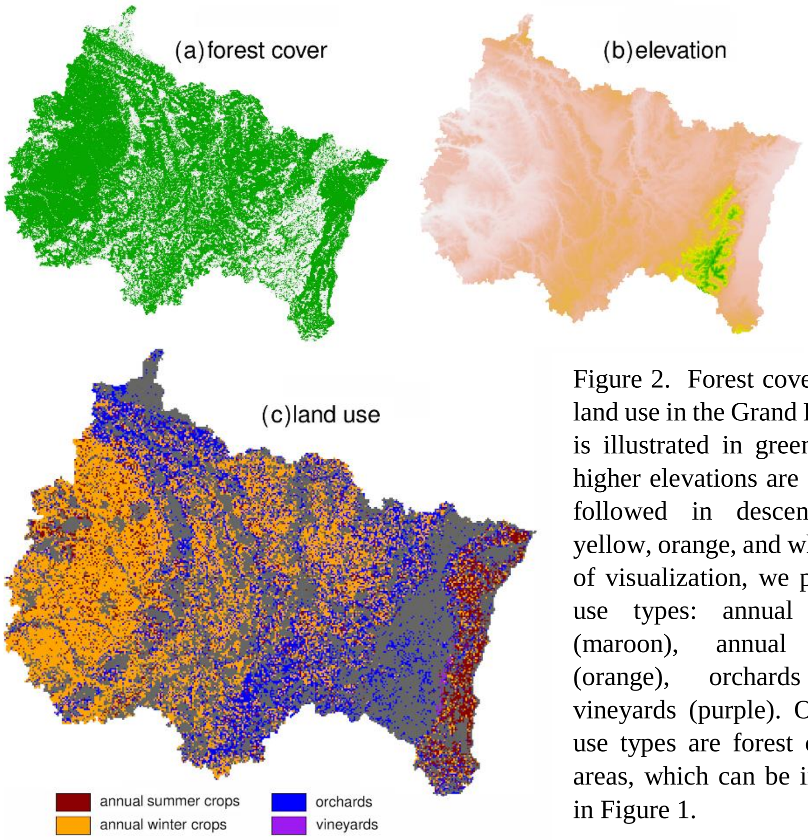

Figure 2. Forest cover, elevation, and land use in the Grand Est. Forest cover is illustrated in green. For elevation, higher elevations are shown by green, followed in descending order by yellow, orange, and white. For the sake of visualization, we present four land use types: annual summer crops (maroon), annual winter crops (orange), orchards (blue), and vineyards (purple). Other major land use types are forest cover and urban areas, which can be inferred here and in Figure 1.

Rather than taking one of the large modelling frameworks to estimate ecosystem service provisioning, we take them as inspiration and build our phenomenological models directly from the literature. Doing so allows us to increase the transparency of our work by enabling us to provide all base data and code for generating our ecosystem service supply maps. Furthermore, while many case studies measure a larger number of ecosystem services, we find that considering a smaller set allows us to go deeper into understanding the data, the models, and the underlying processes that go into the actual supply of ecosystem services in the Grand-Est. A summary of ecosystem services and the methods to measure them are located in Table 1.

We measure a set of seven ecosystem services at the regional spatial scale. As France possesses a large agricultural sector with a high degree of variation in its crops produced, we first consider ***agriculture production probability***, measured as a binary agriculture/not agriculture index. The presence or absence of agriculture was taken from the CESBIO land use data set, defined as either annual summer or winter crops, orchards, or vineyards. Different types of agriculture may matter when considering ecosystem service provisioning. We would expect different crops, as well as traditional versus organic practices, to have different management practices that affect service supply. However, due to the large diversity of agricultural products in France (346 different types of crop and livestock products were exported in 2018[5]), we believe that the proper distinction between agriculture types and products is better left for future work. Furthermore, we find that differentiating between agricultural types is more important when considering the economic valuation of the ecosystem service, where benefits and costs between crop types becomes more important.

Second, we measure four indicators of ***biodiversity***. While biodiversity is not an ecosystem service *per se*, it is known to be positively correlated with regulating services such as carbon sequestration, pest regulation, and soil mineralization (Cardinale et al., 2012; Millennium Ecosystem Assessment, 2005). However, fine-scale regional surveys of biodiversity are difficult to come by[6]. Therefore, we used taxonomic species richness of threatened or protected species as a proxy for biodiversity. Similar approaches have been applied in the United Kingdom for species of "conservation concern" (Anderson et al., 2009; Eigenbrod et al., 2009; Eigenbrod et al., 2010), and numbers of threated or protected species are often used as proxies for biodiversity in economic valuation studies (Bartkowski et al., 2015). Data were compiled from the National Inventory of Natural Heritage (INPN). The database is based on an atlas (grid) of 10 km spatial resolution, where species occurrences are aggregated by taxonomic groups to produce a series of biodiversity maps for protected species. We focused on taxonomic groups with different environmental requirements, theoretically representing different facets of biodiversity. Specifically, we produced maps for threatened amphibians, birds, and reptiles.

However, biodiversity of threatened or protected species is not necessarily correlated with common ones. In other words, biodiversity of threatened species may not be a good proxy of biodiversity of common species. Threatened species may exist as endemic, refugia populations with specific distribution patterns or may be oversampled compared to common species. Therefore, we supplemented our maps of threatened species with a well-established map of tree biodiversity in the Europe (Mauri et al., 2017). In its raw form, the data exist as occurrences of 242 species across the

[5] https://www.fao.org/faostat/en/#data/TCL
[6] It is possible to construct maps of species richness by taxonomic groups through the Inventaire National du Patrimoine Naturel (INPN), but it is at the departmental level.

European Union, compiled from existing European tree distribution datasets (Forest Focus and Biosoil) and previously unpublished National Forest Inventories datasets. We overlaid the raw data onto a 10 km by 10 km grid and aggregated species occurrences by species type within each grid cell to measure tree species richness at the 10 km resolution.

Thirdly, as it is one of the more well-studied ecosystem services in the literature (Crossman et al., 2013; Feld et al., 2009; Issa et al., 2020; Martinez-Harms and Balvanera, 2012; Seppelt et al., 2011), we measured a suite of models to estimate ***aboveground carbon storage***. First, we use two lookup table approaches based on land use type. The first assigns a categorical "low", "intermediate", or "high" carbon storage potential based on the type of land occupation (Egoh et al., 2008; Rouget et al., 2004).

The second assigns an average quantity of carbon stored per hectare to a specific land use type (Bai et al., 2011; Chan et al., 2006; Maes et al., 2012; Naidoo et al., 2008; Spawn et al., 2020; Swetnam et al., 2011; Vallet et al., 2018). Our values of aboveground carbon storage are in line with that of Gibbs et al. (2007), who based their analysis on Intergovernmental Panel on Climate Change guidelines for national greenhouse gas emissions (Intergovernmental Panel on Climate Change, 2006). Our values of carbon storage per land use type in each case are reported in Table 2. We then complement these measures with a published map of aboveground carbon storage by Spawn et al. (2020), which is based on local, regional, and national data sets including national inventories. These methods are motivated by the literature and/or expert opinion and work well for large-scale studies, but can ignore local spatial heterogeneities across landscapes with similar land uses.

To account for some of this, we follow the approach of Dong et al. (2003), Myeong et al. (2006), Yao et al. (2014), and Amoatey et al. (2018) and relate aboveground carbon storage to reflectance data, measured by the normalized difference vegetation index or NDVI. We obtained NDVI data from the Google Earth Engine (Ermida et al., 2020; Jiang et al., 2008)[7] and transformed values of NDVI to carbon storage per pixel using the functions derived by Myeong et al. (2006), Yao et al. (2014), and Amoatey et al. (2018).

[7] As reflectance changes seasonally and annually, we specifically use the annual average between 2010 and 2020 for our analysis.

Table 1. Summary of ecosystem service provisioning models.

Ecosystem service	**Model description**
Agriculture	Binary if annual summer or winter crops, orchards, or vineyards [*]
Biodiversity	
National Inventory of National Heritage	Number of threatened species of amphibians, birds, and reptiles [ƚ]
Mauri et al. (2017)	Tree species richness [Ƒ]
Carbon storage (C)	
Amoatey et al. (2018) – Institutions	Power law relationship, $C = 4735 * exp\ (0.7075 * NDVI)$ [ꜧ]
Amoatey et al. (2018) – Parks and gardens	Power law relationship, $C = 3453.6 * exp\ (5.9194 * NDVI)$ [ꜧ]
Myeong et al. (2006)	Power law relationship, $C = 107.2 * exp\ (0.0194 * NDVI)$ [ꜧ]
Yao et al. (2014)	Power law relationship, $C = 6445.014 * (NDVI \wedge 2.390)$ [ꜧ]
Egoh et al. (2008)	Low/intermediate/high potential by land use type [§]
Gibbs et al. (2007)	Lookup table by land use type [§]
Spawn et al. (2020)	Aboveground carbon storage map [ꜧ]
Net ecosystem productivity	
Maes et al. (2015)	Net ecosystem productivity map
Pastureland	Binary if natural or intensive grassland [*]
Pollination (P)	
Ricketts et al. (2008)	Exponential function of distance to natural forest, $P = exp\ (-0.00053 * distance)$ [þ]
Schulp et al. (2014)	Map of percentage of suitable pollinator habitat [Ω]
Schulp et al. (2014)	Map of pollinator visitation probability [Ω]
Soil loss by water erosion	Mean annual soil loss map [ψ]
Panagos et al. (2020)	

[*] Taken from the CESBIO land use and land cover data (https://labo.obs-mip.fr/multitemp/) (10 m resolution).
[ƚ] As listed by the National Inventory of Natural Heritage (INPN) (https://inpn.mnhn.fr/) (10 km resolution).
[Ƒ] Compiled from species occurrence data, aggregated using a 10 km grid (10 km resolution).
[ꜧ] Associated data from the Google Earth Engine are reported at <1m resolution.
[§] Corresponding values of carbon storage by land use type can be found in the Supplemental Material (10 m resolution).
[þ] Natural forest data is provided by the European Commission Joint Research Centre forest cover data (https://forest.jrc.ec.europa.eu/en/) (25 m resolution).
[Ω] Calculated at the 1 km resolution.
[ψ] Estimated using the revised universal soil loss equation (RUSLE) (100 m resolution).

Fourth, we consider ***grazing probability*** as a binary pastureland/not pastureland variable. We consider a pixel to be pastureland if it is classified as natural or intensive grassland in the CESBIO land use data set. Like agriculture, it is difficult to differentiate grassland types by species and production, which is further compounded by the fact that farmers may routinely share their land between flocks. We leave this for future study.

Fifth, like aboveground carbon storage, we adopt three measures for ***pollination*** potential. First, we relate the probability of pollinator visitation to distance from forest (taken as a proxy for pollinator habitat), as proposed by Ricketts et al. (2004) and Ricketts et al. (2008). Specifically, we used a map of forest cover from the European Commission Joint Research Centre (JRC) to create a proximity map of the distances of the centers of each pixel to the nearest natural forest pixel (broad-leaved, coniferous, or mixed), and then fit the proximity data to a function defined in Ricketts et al. (2008) to estimate the mean visitation probability for temperate regions. Additionally, we supplemented our analysis with two published pollination maps by Schulp et al. (2014), which include the percentage of suitable pollinator

Table 2. Aboveground carbon storage conversion factors by CESBIO land use type.

Land use type	*Carbon storage classification (Egoh et al., 2008)*	Carbon storage (tons/hectare) *(Gibbs et al., 2007)*
annual summer crops (*culture été*)	intermediate	8
annual winter crops (*culture hiver*)	intermediate	8
broad-leaved forests (*foret feuillus*)	high	90
coniferous forests (*foret coniferes*)	high	130
natural grasslands (*pelouses*)	intermediate	9
woody moorlands (*landes ligneuses*)	intermediate	9
continuous urban fabric (*urbain dense*)	none	0
discontinuous urban fabric (*urbain diffus*)	none	0
industrial and commercial units (*zones ind et com*)	none	0
road surfaces (*surfaces routes*)	none	0
bare rock (*surfaces minerales*)	none	0
beaches, dunes, and sand (*plages et dunes*)	none	0
water bodies (*eau*)	none	0
glaciers and perpetual snow (*glaciers ou neige*)	none	0
intensive grasslands (*prairies*)	intermediate	9
orchards (*vergers*)	high	8
vineyards (*vignes*)	intermediate	90

habitat and the probability of pollinator visitation, both of which are based on Corine land cover and landscape green elements data.

Our sixth and seventh ecosystem services are regulating ecosystem services: soil loss by water erosion (erosion prevention) (Panagos et al., 2020; Panagos et al., 2015) and net ecosystem productivity (Maes et al., 2015). The former is based on the universal soil loss equation (USLE) (Batjes, 1996; Nelson et al., 2009; Wishmeier and Smith, 1978), which relates soil properties, topology, land management and vegetation cover, and precipitation to predict potential soil loss by water erosion. Specifically, we used the published map of Panagos et al. (2020), who adopted the updated revised universal soil loss equation (RUSLE) to estimate mean annual soil loss rates (tons/hectare/year) across the European Union in 2016.

Net ecosystem productivity (NEP) is defined as an ecosystem's net accumulation of carbon, which depends on the balance between gross primary production and losses via plant and animal respiration, leaching, plant emissions, methane fluxes, and disturbances (Chapin et al., 2012). For ecosystems that experience little or no disturbances, NEP is given by the difference between carbon gains from plant primary production (photosynthesis) and carbon losses by respiration and leaching. We used the published net ecosystem productivity map of Maes et al. (2015), which was prepared as part of an European Commission Joint Research Council report to measure spatial-temporal trends in ecosystem services across the European Union. Specifically, Maes et al. (2015) used reflectance data as a proxy for net ecosystem productivity, defining it as the difference between net primary productivity and decomposition rates of dead organic matter (taken as a proxy for heterotrophic respiration). They then adopted the "Phenolo" algorithm of Ivits et al. (2013) to convert spatial maps of the reflectance data to plant primary productivity, adjust for decomposition of dead organic matter, and normalize net ecosystem productivity to a dimensionless scale of 0 to 1.

In order to compare our maps of ecosystem services and calculate the correlations between them, we aligned our raster layers to be the same spatial extent and resolution. Layers were resampled using a bilinear nearest-neighbor aggregation up to the resolution of the coarsest layer (numbers of threatened species, 10 km resolution), and then all layers were cropped to the same spatial extent using the '*raster*' and '*sp*' packages in R v.3.6.2. In doing so, we implicitly transform our binary presence/absence measures of agriculture and grazing to a probability of presence/absence based on their distance to a pixel where that service is present. Scripts for generating the ecosystem service maps – including aligning raster layers to be the same spatial extent and resolution – and the final data layers can be accessed from the French national governmental research data repository (https://doi.org/10.57745/NG3QSF).

Measuring the interactions between ecosystem services

We calculate the interactions between our ecosystem services by calculating the Pearson correlation coefficients between each pair of ecosystem services using data across the entire Grand Est region. Other methods for estimating interactions exist in the literature, such as principle component analysis (PCA), production possibility frontiers, and regressions (Feld et al., 2009; Lee and Lautenbach, 2016; Vallet et al., 2018). However, we prefer correlation coefficients as they are widely used and accepted in the literature, transparent, and provide reasonable estimates of interactions (Chan et al., 2006; Raudsepp-Hearne et al., 2010; Vallet et al., 2018). Calculation of the correlation coefficients were conducted in R 3.6.2. Scripts for the correlation analysis and visualization of the data are found on the French national governmental research data repository (https://doi.org/10.57745/NG3QSF).

3. RESULTS

Distribution of ecosystem services in the Grand Est

Maps of the spatial distribution of ecosystem services across the Grand Est region are presented in Figures 3-5. We would like to stress that while it is tempting to interpret our results in absolute terms, it is better to view them as potential average values of services supplied across the region or as a tool to understand *where* ecosystem services are being supplied in the region. As we will discuss in the next section, our results do come with a certain amount of uncertainty associated with the models and data used to estimate them.

Agriculture is concentrated in the west and far east of the region. The western section is predominantly made up of annual winter crops mixed with a smattering of annual summer crops, and bordered by a band of orchards extending south from the Ardennes. The eastern section is mostly annual summer crops, though it includes all agricultural types in the CESBIO data. Areas between the two are characterized by annual winter crops and orchards, with orchards being especially prevalent in the Vosges.

Grazing is an inverse of agriculture, being primarily concentrated in the northwest in the Ardennes and along a belt from the northeast to south central part of the region. Much of this is due to the nature of the land use data – a parcel of land can belong to only one land use type, not two or more simultaneously.

In terms of our measures of ***biodiversity***, we find overlap in the numbers of threatened amphibians and birds. Overall, numbers of all three taxonomic groups of protected species are more-or-less evenly distributed across the region (though reptiles exhibit a more homogenous distribution). Surprisingly, numbers of threatened species are lower in the high-agriculture section in the western part of the region and exhibit their highest values in national and regional parks of the Ardennes and Ballons des Vosges. This suggests potential sampling bias in the data, with sampling effort being higher in parks than on private agricultural areas. Tree diversity is highest along a north-south band in the central part of the region aligning with the distribution of communal and public forests. Interestingly, the highest

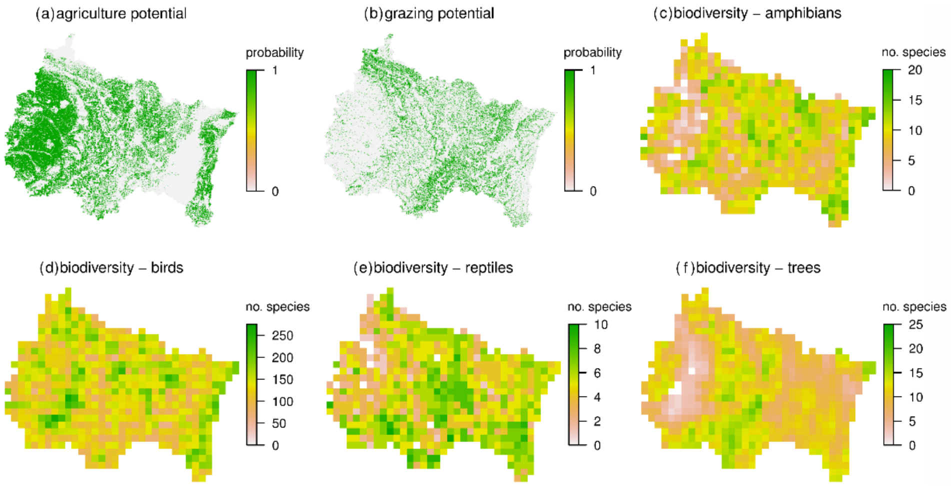

Figure 3. Distribution of agriculture and grazing (probabilities of occurrence), and biodiversity (number of threatened or protected amphibians, birds, and reptiles; number of tree species) in the Grand Est region of France.

values of tree biodiversity are not in the Vosges, which may be due to this area being at higher elevations with lower forest cover, or the possibility of more homogeneous species distributions (such as resinous ones).

The spatial distribution of ***aboveground carbon storage*** is fairly consistent across all models tested, though we observe gross differences in the quantitative values of carbon stored between models (more to come in the Discussion). In particular, carbon storage is highest in the communal and public forests in the Ballons des Vosges regional park.

Not unsurprisingly, ***net ecosystem productivity*** parallels the distribution of carbon storage, being concentrated in forests, grasslands, and pasturelands. ***Soil loss by water erosion*** exhibits its highest rates in grasslands in the northwest and south-central parts of the region, and in the higher elevations of the mountains.

Pollination, in terms of visitation probability, is quite high across the region, greatest in forests and lowest near agricultural areas. In terms of the percentage of suitable habitat, it is concentrated in the central and southeast sections of the region and a small area in the northwest.

Correlations between ecosystem services

Correlation coefficients are often used synonymously with the terms "interactions" or "associations", and there some debate regarding what makes an interaction or association, what are the different types of interactions (tradeoff versus a synergy), how they form, and what level of correlation is meaningful (Lee and Lautenbach, 2016; Vallet et al., 2018). When presenting our findings, we will call our "interactions" for what they are – correlations – and interpret their values in a purely positive/negative mathematical way. We consider our correlations to be statistically significant if they have p-values at less than the ten percent confidence level. A presentation of the correlations between ecosystem services in their entirety can be found in Figure 6.

For within-service correlations, all measures of biodiversity are positively correlated with each other, except numbers of threatened birds and tree diversity, which are negatively correlated. The latter aligns with intuition. We would expect more diverse forests to provide better habitat for avian species, and an inverse relationship between tree diversity and the number of threatened bird species. (In contrast, we would expect a positive relationship between tree diversity and the number of *common* bird species.)

All measures of aboveground carbon storage are strongly correlated with each other, with the lowest value being a correlation of 0.76 (the reflectance models exhibit correlations close to 1). The pollination models also show positive correlations, though the correlation between our two pollinator visitation probabilities is not statistically significant.

For between-service correlations, agriculture is negatively correlated with all services except numbers of threatened birds and the Schulp et al. (2014) pollinator visitation rate (which are positively correlated). Numbers of threatened amphibians is positively correlated with all ecosystem services except the aforementioned agriculture (negative correlation), the Egoh et al. (2008), Gibbs et al. (2007), and Spawn et al. (2020) carbon models, Schulp et al. (2014) pollinator visitation, and soil loss (not significant). Numbers of threatened bird species is negatively correlated with every service except agriculture and threatened amphibians and reptiles (positive correlation), and the Ricketts et al. (2008) and Schulp et al. (2014) suitable habitat pollination models (not significant). Numbers of threatened reptiles show the same correlations as amphibians, except positive correlations for all carbon models,

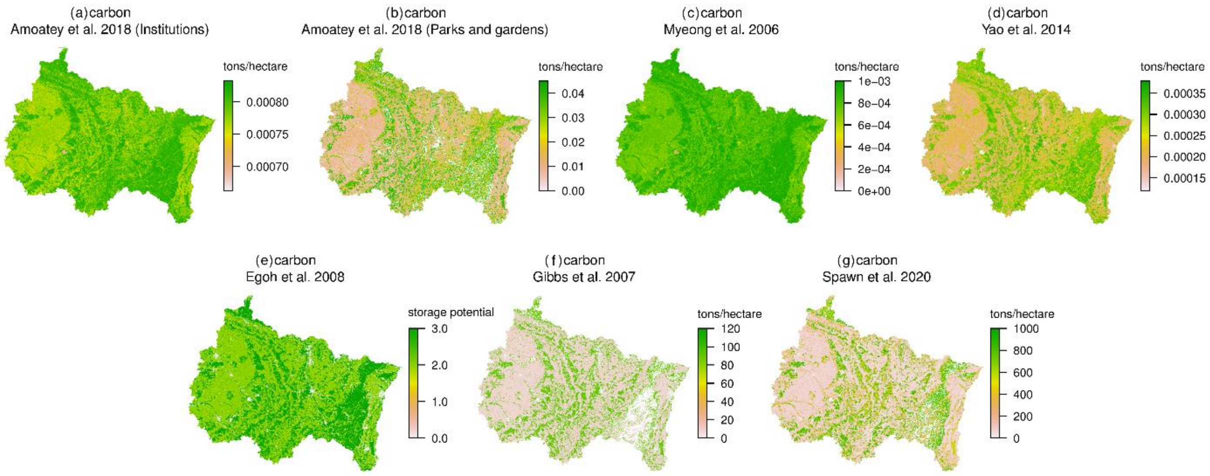

Figure 4. Distribution of aboveground carbon storage in the Grand Est, according to a set seven models. Units are represented as tons of carbon stored per hectare.

no statistically-significant relationship with grazing, and a negative correlation with the Schulp et al. (2014) pollinator visitation model. Tree diversity is negatively correlated with agriculture and numbers of threated bird species, and positively correlated with all others except Schulp et al. (2014) pollinator visitation and soil loss (not significant).

In addition to the above correlations, carbon models exhibit positive correlations with grazing, net ecosystem productivity, and the Ricketts et al. (2008) and Schulp et al. (2014) suitable habitat pollination models, and either negative or statistically-insignificant correlations with Schulp et al. (2014) pollinator visitation and soil loss. Grazing is positively correlated with net ecosystem productivity, all pollination models, and soil loss. Net ecosystem productivity is positively correlated with pollination, but is not statistically-significantly correlated with soil loss. Finally, all pollination models are positively correlated with soil loss.

4. DISCUSSION

Our measures of ecosystem services supply and their correlations are consistent with trends in the general scientific literature and what we would expect from the models and the data. Rather than discussing each of these in turn (which we believe would be exhaustive and not fruitful), we would

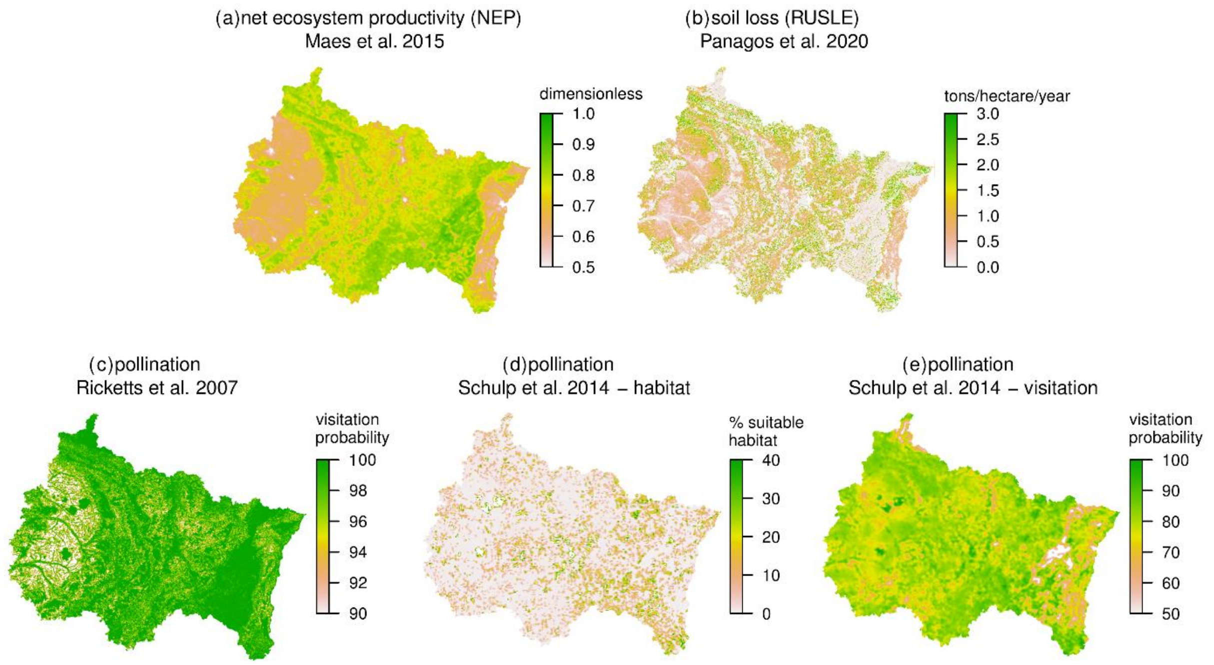

Figure 5. Spatial distribution of net ecosystem productivity (dimensionless), soil loss by water erosion (tons/hectare/year), and pollination (visitation probability; percent suitable habitat).

direct the reader to Vallet et al. (2018), who provides a detailed review of tradeoffs and synergies between ecosystem services in the literature, and Shanafelt et al. (2023), who provides a deep discussion of the models and data, and what this means for the resulting estimation of ecosystem services and their interactions. Instead, we would prefer to highlight some key findings and how they fit into the greater literature and forest policy. For example, there exists a general negative relationship between agriculture and biodiversity (Mattison and Norris, 2005; Phalan et al., 2011; Reidsma et al., 2006). We find a positive correlation between agriculture and the number of threatened bird species, and negative correlations between agriculture and the number of threatened amphibian reptile species. One way to interpret this is that viable habitat for each type of species is suitable or not for agriculture compared to other land uses. If, for example, the most viable habitat for bird species in the Grand Est lied in lowlands used by agriculture, then we would expect a positive correlation between the two. In a recent paper Shanafelt et al. (2023) found the opposite correlations between agriculture and numbers of threatened birds (negative) and amphibians (positive) at the national level. Deeper investigation regarding the ecological and habitat requirements of these types of species in the Grand Est is warranted in future studies.

Interestingly, we find similar trends in our correlations as a similar study at the national level (Shanafelt et al., 2023), though due a much smaller sample size we see fewer statistically-significant correlations. We would expect different regions to have different bio-physical properties and processes, land uses, and management regimes, and subsequently different levels of ecosystem service provisioning and correlations between them. Understanding which correlations between ecosystem services are robust to a variety of landscapes and sample sizes, and which must be studied at finer scales with site-specific case studies, offers insights into how we can make general statements about the trends of the interactions between ecosystem services. Many interactions between ecosystem services are straightforward to measure and understand (such as timber production and carbon storage) and we can make general statements about the nature of their interactions across landscapes. Others are much less intuitive (e.g., regulating services), and it is difficult to make generalizations about the interactions between them. What these services may be, and what social, physical, and ecological properties of their local environments drive their interactions, is valuable knowledge going ahead.

Our maps of ecosystem service supply do a decent job of indicating where different ecosystem services are being provided in the Grand Est, but should by no means be used to indicate what *should* be provided at those locations. Answering this question requires understanding the portfolio of what people use and value – one of the primary goals of the PERCEVAL project. For example, we find that agriculture is negatively correlated with almost all other ecosystem services in our study. That is, areas with a high probability of being agriculture exhibit lower values of other ecosystem services on average. It is easy to say that agriculture is “bad” for ecosystem service provisioning, but this may be quite erroneous in the context of human society. Agriculture is an integral part of the French economy, and the “best” scenario from a social welfare perspective may be not to change land from agriculture to something else, but rather to improve agricultural management to lessen or reverse its impact on other ecosystem services. We find similar stories when balancing economic development and species conservation, where the extensive economic value of human infrastructure necessitates finding a middle ground between economic and ecological goals (Melstrom et al., 2021).

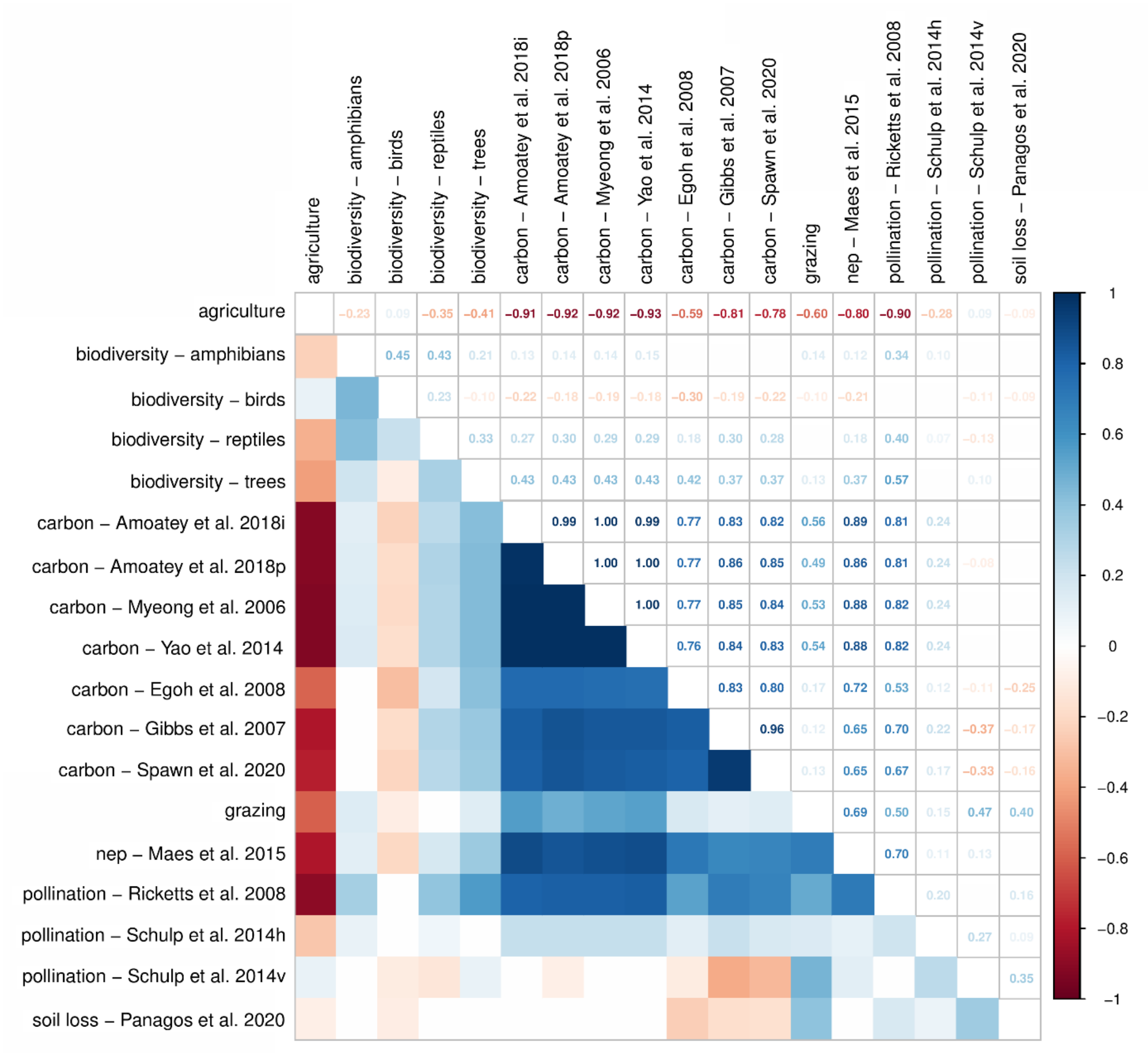

Figure 6. Correlation coefficients for all pairwise combinations of ecosystem services in our study. Values of the correlation are presented by color in the lower triangular part of the plot, and numbers in the upper triangular. Only pairwise correlations with p-values greater than 0.1 are presented.

However, while we cannot use our study to say how ecosystem services should be provided in the Grand Est, there are some interesting take-away messages about what is important for ecosystem service provisioning in the Grand Est. For instance, the importance of forests – communal and public forests – cannot be discounted. Many of the "hotspots" of individual ecosystem services lie not only within the regional or national parks of the region, but within communal and public forests. The Office National des Forêts (ONF) provides information on the boundaries of public forests, making it possible to extrapolate public and private forest holdings. It would be interesting to explicitly test the contribution of public and private forest (separately) to the spatial provisioning of our ecosystem services. Additionally, our findings have implications for how altering land use to change the provisioning of one or more services may affect the provisioning of others (Lee and Lautenbach, 2016; Vallet et al., 2018). Correlations between ecosystem services are often taken as indicators of their tradeoffs and synergies – a positive or negative correlation implies that an increase in one results in an increase or decrease in the other. In the context of our study, if agriculture was highly valued by society, then altering land use to maximize food provisioning (converting land to agriculture) would likely result in lower carbon storage and tree diversity, and potentially threaten bird diversity compared other land use types. Whether this solution is "optimal" depends on how society values those other services.

Question of data availability, quality, quantity, and uncertainty are certainly key limitations to the field (Crossman et al., 2013; Egoh et al., 2012; Hou et al., 2013; Layke et al., 2012; Martinez-Harms and Balvanera, 2012), and our study is no exception. Many of our models rely on aggregated or proxy data for estimating ecosystem services, which could potentially drive the nature of the interactions between ecosystem services. For example, agriculture, grazing, and the Gibbs et al. (2007) and Egoh et al. (2008) carbon models are based on the CESBIO land use data. Agriculture and grazing are measured directly from the presence or absence of each, and the carbon models assign a value of carbon based on land use type, with forests and grasslands storing more carbon than agriculture. Therefore, we would expect a negative interaction between agriculture and these carbon models. Furthermore, while few studies in the literature address issues of uncertainty (see Shanafelt et al. (2023) for a brief review), those that do have found discrepancies between land cover-based proxy methods and fine-scale point measurements of ecosystem services in the field (Eigenbrod et al., 2010; Roussel et al., 2017). The most apparent example of this in our study is our set of carbon models – we find gross differences in their estimation of the quantity of carbon stored in the Grand Est, which is mainly due to the data driving their measurement. The reflectance data methods of Amoatey et al. (2018), Myeong et al. (2006), Yao et al. (2014) are calibrated to desert and urban ecosystems – we would not expect them to give particularly accurate measurements in forests or temperate climates. The Gibbs et al. (2007) estimates are based on global averages per land use type. The Spawn et al. (2020) is derived from a mix of local, regional, and national data sets including national inventories. Therefore, it is worth reinforcing that our findings should not be used as absolute point measures of what is being supplied at specific sites in the Grand Est, but rather potential average values or, better yet, as a tool to understand where ecosystem services are being provided in the region.

Understanding how to manage ecosystem to maximize the full portfolio of human and ecological needs is a complex question at the heart of the PERCEVAL project. It requires an understanding of how ecosystem services are provided across the region and how those services may interact with other, but also the social side of the problem – what do people value as different groups with heterogeneous preferences, and how do ecosystem services fit into the portfolio of the benefits of society as a whole. While there is certainly scope to improve our measurements of ecosystem services with on-the-ground field estimates, we have provided a first approximation of ecosystem service provisioning in the Grand Est. The remaining parts of the PERCEVAL project will address the second part of this question.

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