How the ecosystem extent is changing: A national-level accounting approach and application

Adrián G. Bruzón 1*, Patricia Arrogante-Funes 1, Pablo Martínez de Anguita1, Carlos J. Novillo1, Fernando Santos-Martín 1

1Department of Chemical and Environmental Technology, ESCET, Rey Juan Carlos University, C/Tulipán s/n, Móstoles, 28933 Madrid, Spain

*Correspondence: adrian.bruzon@urjc.es

Keywords: Ecosystem accounting, extent account, natural capital, SEEA-EA, Spain.

Abstract

Assessing the spatial and temporal changes in ecosystems is essential to account for natural capital contribution to human well-being. However, various methods to quantify these changes challenge the development of reliable values which can be integrated into national statistical accounts. Following the international system of environmental-economic accounting framework, which recently adopts an ecosystem accounting standard. We present a novel approach to develop an ecosystem extent account from existing ecosystem classifications. This study shows the spatial and statistical extent account of 26 ecosystems (i.e. forests, grasslands, croplands, and urban, among others) between 1970 and 2015 at the national scale. Extent accounts were developed at a resolution of 25 meters and provided reliable information on how ecosystem types have changed over time in Spain. Our results reflect three main patterns in the extension account: (i) an increase in forest ecosystems, (ii) a considerable decrease in agroecosystems (especially annual croplands), and (iii) substantial development of urban areas. To the best of our knowledge, this method is the first attempt to develop a robust methodology to measure the extent of ecosystems at the national level. The proposed approach is crucial for a strong knowledge of ecosystem dynamics and their implications for ecosystem conditions and services at a national level. This has potential applications in urban planning, green infrastructure development, and multiple uses for territory management and policies, integrating natural capital into official statistics and mainstreaming ecosystems into national-level planning and monitoring processes.

1. Introduction

The importance of ecosystems and their services to human well-being and the economy is well established (Banerjee et al., 2020; IPBES, 2019; La Notte et al., 2019; MA, 2005; Mäler et al., 2008; Obst et al., 2016; Zagonari, 2016). Multiple international commitments, such as the United Nations (UN) Sustainable Development Goals (SDGs) and the Convention on Biological Diversity (CBD), advocate for a system capable of monitoring and quantifying ecosystem changes across spatial and temporal scales (Crossman et al., 2013; Maes et al., 2013). Over the past several decades, substantial efforts by international organisations and the scientific community have been dedicated to developing an ecosystem accounting framework within the System of Environmental-Economic Accounting (SEEA) of the UN Statistical Division (Esen and Hein, 2020; Obst, 2015; United Nations et al., 2014, 2021). In particular, the System of Environmental-Economic Accounting- Ecosystem Accounting (SEEA-EA) constitutes an integrated statistical framework for organising biophysical data, measuring ecosystem services, tracking changes in ecosystem extent and condition, and linking this information to economic and other human activities (United Nations et al., 2021). In March 2021, the United Nations Statistical Commission (UNSC) adopted the System of Environmental-Economic Accounting—Ecosystem Accounting (SEEA EA). More specifically, chapters 1-7 as an international statistical standard and chapters 8-11 as internationally recognised statistical principles and recommendations for ecosystem services and assets valuation. This new statistical framework will enable countries to measure their natural capital and understand nature's contributions to our prosperity and the importance of protecting it. It will mark a major step towards incorporating sustainable development in economic planning and policy decision-making and could important impact efforts to address critical environmental emergencies, including climate change and biodiversity loss. UNSC has encouraged nations to implement the SEEA-EA in their territory in the coming years (United Nations Statistical Commission, 2021).

Therefore, every nation's challenge is to design consistent methodologies for the development of ecosystem accounting according to the SEEA-EA standard. Furthermore, this methodology needs to facilitate integrated assessments and analytical modelling in a high-resolution spatial manner (Weber, 2007). Currently, several countries are testing different ecosystem accounts at different levels Spain (Campos et al., 2019; Vicente et al., 2016), Germany (Grunewald et al., 2020), Netherlands (Hein, Remme, et al., 2020), Australia (Keith et al., 2017), Finland (Lai et al., 2018), Bulgaria (Nedkov et al., 2016), Europe (Petersen, 2019; Weber, 2007), Mexico (Schipper et al., 2017) and Mauritius (Weber, 2014). Nevertheless, the absence of standardised methods to quantify these accounts is one of the principal challenges that research and policy institutions must face to provide reliable ecosystem accounts (Boyd and Banzhaf, 2007; Eppink et al., 2012).

The SEEA-EA framework describes the relationship between ecosystem and economic assets and seeks to integrate this data within the System of National Accounts (SNA) (Hein, Bagstad, et al., 2020; La Notte and Rhodes, 2020; La Notte et al., 2019). SNA is an international standard that provides a systematic compilation of information needed for a nationwide economic analysis and policymaking covering the entire economy robustly and simplified way. However, ecosystem information is intrinsically different from the environmental information traditionally included in the SNA (Daily and Matson, 2008). The SEEA-EA framework consists of biophysical and economic core accounts: extent, condition, services and assets (United Nations et al., 2021). The first step in this framework is to measure the extent of different ecosystem types and their transformations over time.

Further developments will consist of analysing the conditions of these ecosystems, the physical and monetary study of ecosystem services flows, and each ecosystem asset's economic value (Hein, Bagstad, et al., 2020). Extent account consists of knowing the changes over time of the different ecosystem types within an accounting area (Petersen, 2019). This extent account provides spatial data of each ecosystem type's opening and closing stock. In addition, it provides the necessary information to other ecosystem accounts as conditions and services (United Nations et al., 2021).

The ecosystem extent account requires delineating different ecosystem types within an accounting area (United Nations et al., 2021). The measure of two kinds of variables then shows the following: (i) the opening and closing stock of different ecosystem types in a spatially explicit manner (Petersen, 2019); and (ii) the ecosystem flows through time, expressing the relationship between land cover dynamics and ecosystem functions (Hellwig et al., 2019). Changes in ecosystem extension has direct consequences on ecosystem services and biodiversity, being necessary to include studies of gross and net change of the extension so that these accounting methods will be key for environmental assessment or climate change research (Fuchs et al., 2016).

The national-level extent of the ecosystem is an emerging line of research, and different countries, especially those in the European Union, are currently working on application cases, for example in United States (Dvarskas, 2019; Warnell et al., 2020), Germany (Grunewald et al., 2020), Netherlands (Hein, Remme, et al., 2020), Myanmar (Lee et al., 2020) or Czech Republic (Vačkářů and Grammatikopoulou, 2019). However, most of these initiatives are based on land cover maps, instead of ecosystem classification, as the starting point for building a national extent account. Nevertheless, the recognition between land cover and ecosystem types is essential because it defines a unique environmental asset that delineates the space in which economic activities, environmental processes, and assets are located (United Nations et al., 2021). Therefore, ecosystem and land cover classifications need to be integrated and further developed to produce suitable national extent accounts because the focus on land cover is not considered ecologically meaningful (UNEP-WCMC, 2017).

The development of an ecosystem extent account poses multiple challenges. One of the most common difficulties is accessing standardised long-term spatial data. To overcome this challenge, most studies have used land cover and land use cartography as a basic spatial unit and linked it with existing ecosystem classifications (Maes et al., 2013; Petersen, 2019). The SEEA-EA has recently recommended the use of the global ecosystem typology (IUCN ET) (Keith et al., 2020) as an international standard to improve the comparability and consistency of ecosystem accounts between different countries (Bogaart et al., 2019). Global ecosystem typology is a hierarchical classification system that defines ecosystems by their convergent ecological functions and distinguishes ecosystems with contrasting assemblages of species engaged in those functions, using simple, accessible, and clearly defined information (Keith et al., 2020).

In this study, we propose a method to develop an ecosystem extent account at the national level, in compliance with the requirements of SEEA-EA that monitor the opening and closing stock of ecosystem types and the flows between them. We think that the proposed approach is a step forward to understand historical ecosystem transformations, which is essential to monitor the impact of land conversion on environmental and ecological factors like food security, biodiversity, or climate change (Fuchs et al., 2015).

2. Methodology

2.1 Ecosystem accounting area

The entire territory of Spain was considered as the study area, which includes the Spanish Iberian Peninsula and the Balearic and the Canary Islands. This area has some different bioclimatic region, dominate by Mediterranean, but also include Alpine, Atlantic and Macaronesia. This mix of bioclimates makes Spain a privileged study area to observe how the ecosystems has changed, in a world increasingly alerted by climate change, sustainability, and environmental protection while the country is undergoing through multiple socio-ecological changes in the last decades (Santos-Martín, González García-Mon, et al., 2019)

2.2 National ecosystem classification

First, we used the national ecosystem classification developed by the Spanish National Ecosystem Assessment (SNEA, 2014) as the units to be analysed within the terrestrial national territory as the accounting area. The proposed classification of terrestrial ecosystems is strongly related to biogeography and is based on four environmental conditions: (i) human influence measured using land cover data; (ii) altitude, developed from the digital elevation model by the National Geographic Institute (iii) aridity from the thermal index, and (iv) macroclimatic conditions from the average annual temperatures and accumulated precipitation in fifteen-year trends to include consequences of global climate warming in the ecosystem classification (McCARTY, 2001). Hence, we include changes in ecosystem classification in two periods (1955–1970 and 2001–2015), with the aim of including the effect of climate change on ecosystems. In addition, the proposed classification of ecosystems through climatic regimes is in line with the topology developed by the IUCN at the level of ecosystem functional groups (Keith et al., 2020).

Second, we used the land cover, land cover change, and forestry (LULUCF) database designed by the Spanish Ministry of Ecological Transition and Demographic Challenge (Alonso Moya et al., 2020) to analyse spatial changes over time. LULUCF comprises a multi-source database of national and European information, such as Spanish Crop and Harvest Map, Corine Land Cover, Spanish Forest Map, and Spanish Geographic Information System of Agricultural Plots (Alonso Moya et al., 2020). We selected LULUCF as a basic spatial database to map and assess ecosystem extent account at a national level for the following reasons: (i) it offers the possibility to analyse ecosystem changes over an extended period (1970–2015); (ii) is the most accurate and update, high spatial resolution (25 meters) information for the entire national territory; (iii) it is composed of official data and will be periodically updated by the national statistical office in the future; (iv) it allows the ecosystem accounts to be related to climate change initiatives; and (v) can be used to report and assess on biodiversity and ecosystems services (Regulation (EU) 2018/841, 2018). This cartography follows the land cover classification developed by the Intergovernmental Panel On Climate Change (IPCC) Guidelines for National Greenhouse Gas Inventories (IPCC, 2006). This classification highlights land area information to estimate carbon stocks, emissions, and the removal of greenhouse gases associated with LULUCF activities (Arets et al., 2019). More information in Supplementary Materials. The combination of the Spanish ecosystem classification and LULUCF cartography resulted in 26 terrestrial ecosystem types used in this study (Table 1).

We utilised supporting spatial data to measure the information necessary to crosswalk Spanish ecosystem types and the LULUCF database. We used the map of biogeographic regions produced by the European Environmental Agency (2016) to obtain the climate conditions. We measured the thermal index based on bioclimatic areas (Rivas-Martínez, 1983). For maximum and minimum temperatures between 1955 and 2007, we used a dataset included in the project 'Climate of peninsular Spain 1950-2007' (Felicísimo et al., 2011). Otherwise, we utilised the annual mean of the daily temperature between 2008 and 2015 for MODIS1A (Wan et al., 2015). Regarding accumulated precipitation between 1955 and 2007, we used a dataset included in the project 'Climate of peninsular Spain 1950-2007' (Felicísimo et al., 2011) and Terraclimate for the Macaronesia region (Abatzoglou et al., 2018). Between 2008 and 2015, we utilised the annual accumulated global daily precipitation product (CHIRPS) (Funk et al., 2015). Finally, in 2015, we used the Copernicus riparian (Tamame et al., 2018) and coastal (Innerbichler et al., 2021) zones to support LULUCF cartography for these ecosystems. More information about the data sources in Supplementary Materials.

Table 1: Ecosystem classification proposed to develop extent accounts in Spain using different sources.

LULUCF (Arets et al., 2019)	Spanish Ecosystem classification (SNEA, 2014)
Forest land	Sclerophyllous Mediterranean forest
	Continental Mediterranean forest
	Mediterranean mountain forest
	Atlantic forest
	Alpine forest
	Insular forest
Grassland	Sclerophyllous Mediterranean grassland
	Continental Mediterranean grassland
	Mediterranean mountain grassland
	Atlantic grassland
	Alpine grassland
	Insular grassland
Shrubland	Sclerophyllous Mediterranean shrubland
	Continental Mediterranean shrubland
	Mediterranean mountain shrubland
	Atlantic shrubland
	Alpine shrubland
	Insular shrubland
Other Lands	Arid zones
	Coastal areas
	Other lands
Wetlands	Wetlands
	Rivers and lakes
Cropland	Perennial woody crops
	Annual crops
Settlements	Urban

2.3 Validation of the proposed ecosystem classification

To validate whether the proposed ecosystem classification was spatially accurate, we tested the agreement between Spanish ecosystem types (Table 1) with the Land Use and Land Cover Survey (LUCAS) program, which provided in-situ data for European land cover (Eurostat, 2015) as ground observational data in 2006, 2009, 2012, and 2015. First, we harmonised thematic classifications to achieve 26 common categories across all land cover databases. In particular, we adapted the LUCAS thematic classification for LULUCF landcover classification. This harmonisation process is a common practice for comparing land cover categories by developing methodologies (Pérez-Hoyos et al., 2017). Macaronesia and mountain ecosystems were excluded due to the lack of LUCAS data in these areas. In addition, for 2006 and 2009, the Balearic Islands were excluded for the same reason. Next, the test applied a site-specific analysis using a cross-tabulation table, where LUCAS is the reference data (Büttner and Maucha, 2006; Tsendbazar et al., 2016; Vilar et al., 2019). The cross-tabulation table is a typical method for quantifying the spatial agreement between the dataset and the chosen reference dataset (Vilar et al., 2019).

Second, we evaluated the agreement between the ecosystem types and LUCAS using three statistical measures: (i) overall accuracy, which is a measure of spatial agreement between datasets; (ii) Kappa statistic, which compares an observational probability of agreement with a random probability of agreement considering a substantial strength of agreement from 0.6 (Landis and Koch, 1977); and (iii) F_1-score which represents the harmonic mean of the precision and recall, for each ecosystem type to evaluate the fit between LUCAS and each ecosystem type (Powers, 2020).

2.4 Developing an algorithm to produce ecosystem extent accounts

To create an algorithm capable of producing ecosystem extent accounts for different periods, we follow the SEEA-EA approach (United Nations et al., 2021). The workflow of this algorithm is illustrated in Figure 1. More information about workflow in Supplementary Materials. From an ecosystem classification and a land cover mapping, we generated a map of ecosystem types, which is the basic input need it to assess extension changes in an accounting system. This algorithm gives us information on all transformations that occurred in the ecosystems between the periods studied. Including, on the one hand, information on the changes that each ecosystem has undergone (net change, turnover, stable stock, and extension change), and on the other hand, has given us information on the flows that occurred between different ecosystems.

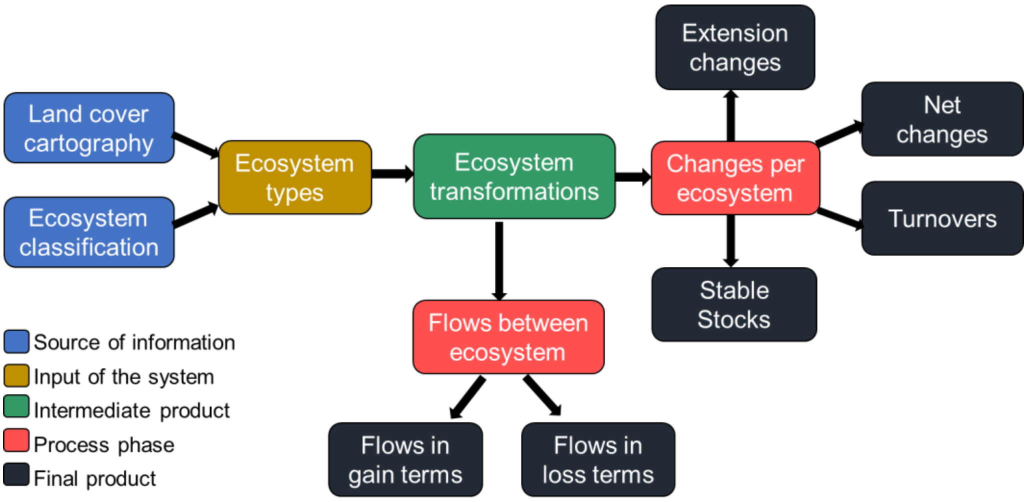

Figure 1: Workflow of ecosystem extent account algorithm

We based our four variables (net change, turnover, stable stock and extension change) on Robert Pontius Jr.'s concepts (2019) However, we adapted them to the SEEA-EA framework (United Nations et al., 2021). The algorithm measures these variables in area units and the percentage of the total area and provides the extent account in an accounting table. The extension change is the difference between the initial and final extents (Equation 1). The net change is the subtraction of the reduction to addition (Equation 2). The turnover is the sum of addition and reduction (Equation 3). The stable stock is the subtraction of the turnover to the initial extent (Equation 4), where *add* is additions in km^2, *red* is reductions in km^2, *x* is the ecosystem type, and *t* is the period considered.

$$Extension\ change\ = final\ extent_{xt} - \ initial\ extent_{xt} \qquad \text{(Equation 1)}$$
$$Net\ Change = \ add_{xt} - red_{xt} \qquad \text{(Equation 2)}$$
$$Turnover\ \ = add_{xt} + \ red_{xt} \qquad \text{(Equation 3)}$$
$$Stable\ Stock = initial\ extent\ _{xt} - turnover_{xt} \qquad \text{(Equation 4)}$$

Together these four measurements give us enough information to account for the extent of ecosystems. Firstly, the change in the extension gives us information about the square kilometres that an ecosystem has gained or lost in the study period. Secondly, the net change gives us information about the net gain or net loss that the ecosystem affects. Thirdly, the turnover expresses how there may be a change in one area between ecosystems, with a simultaneous and reverse change between the same ecosystem in another area. Finally, we measure the stable stock because it is essential to collect the ecosystem extension that did not change. We include these variables because studying only the net change does not give us enough information about the changes that are taking place in the ecosystems.

We used a statistical approach to distinguish between a systematic landscape transition and a seemingly random landscape transition (Pontius Jr et al., 2004). A non-random gain or a non-random loss for a particular ecosystem flow implies a systematic change process (Alo and Pontius Jr, 2008). This method uses a gain and loss cross-tabulation matrix to calculate the differences between the observed and expected changes. To develop the expected values, we based on the chi-square distribution, in which the proportion of the ecosystem that is transformed into another type of ecosystem is due to random chance. This method helps identify systematic processes within a pattern of ecosystem change (Pontius Jr et al., 2004). These flows are calculated automatically for the extent account algorithm, which is presented in tables of (i) a cross-tabulation matrix (measured in percentages of the total), (ii) gains, and losses flow (measured as subtraction of observed and expected changes), and (iii) significant coefficient of these flows and speed of this change (measure as gain and loses flows divided by the expected change). In this study, to ensure that the flows are systematic and not caused by a random process, we used the difference between the observed changes minus the expected changes divided by the value of expected changes, applying a confidence interval of 0.05. These ratios are similar to the ratios that form the basis of chi-square tests (Pontius Jr et al., 2004). The software used from spatial analysis was Arcgis Pro 2.8 and the Python library Arcpy (ESRI, 2020) and the software used from statistical analysis was Python library Numpy (Harris et al., 2020).

3. Results

3.1 Spatial analysis of ecosystem types over time

The spatial analysis of the Spanish terrestrial ecosystem types provides detailed data on the distribution and main changes of ecosystem types over time. Figure 2 shows an example of the spatial representation of ecosystem types in 1970 (Figure 2a) and 2015 (Figure 2b), including high-resolution maps of (I) 'Picos de Europa' National Park (an example of Atlantic and Alpine forests), (II) City of Madrid (an example of urban ecosystems), and (III) 'Teide' National Park (an instance of Macaronesia ecosystem).

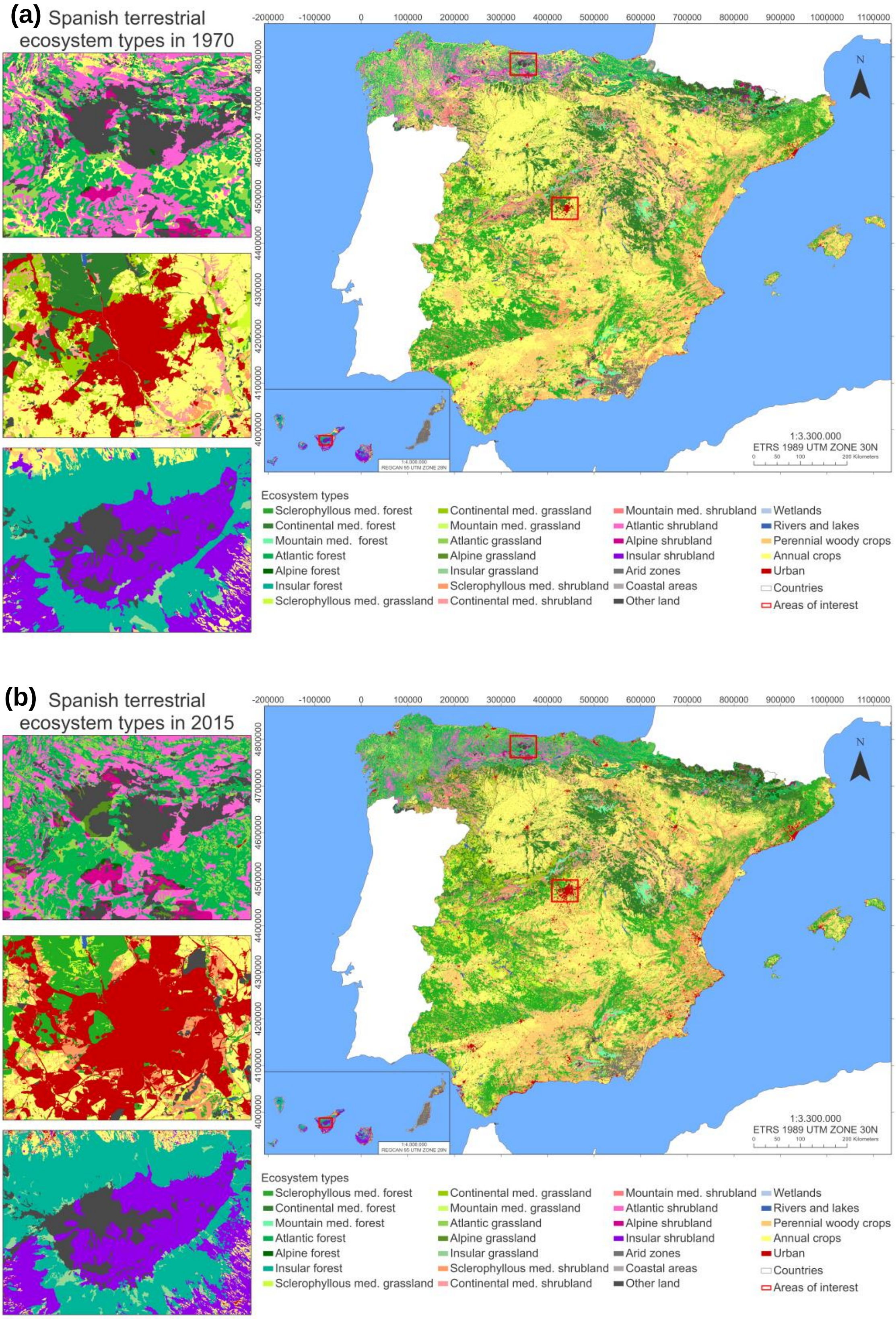

Figure 2: Spatial representation of Spanish ecosystem types. (a) 1970, (b) 2015. Detail areas from top right to bottom: (I) 'Picos de Europa' National Park. (II) The city of Madrid. (III) 'Teide' National Park.

3.2 Validation of the proposed ecosystem classification

The overall results of the validation analysis can be observed in Figure 3. The overall accuracy (a measure of spatial agreement between datasets) and Kappa statistic (which compares an observational probability of agreement with a random probability of agreement) showed a constant increase over time with a maximum in 2015 (0.69 accuracy and 0.65 Kappa). These results indicate a general improvement of 21% for accuracy and 35% for Kappa.

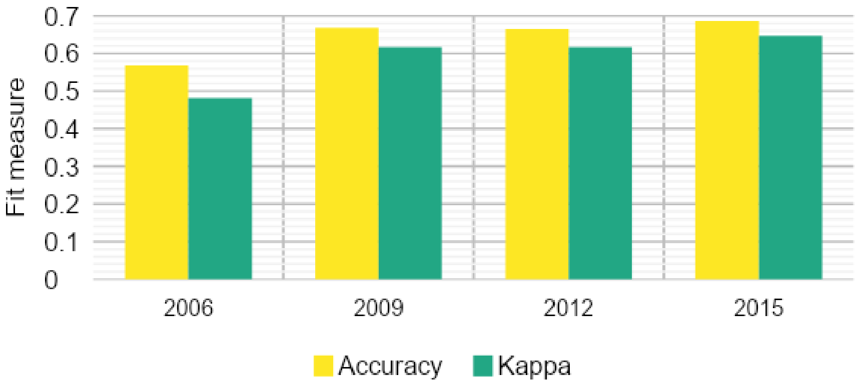

Figure 3: Accuracy and Kappa statistics results of proposed ecosystem types and LUCAS classes (as ground observational data) in the different available data periods.

Concerning the validation analysis by each type of ecosystem, the F_1-score ratio (which represents the harmonic mean of the precision and recall for each ecosystem type) results are illustrated in Appendix 2. Thus, forest and cropland ecosystems had the best scores, while wetlands, other lands, and arid ecosystems showed the worst values.

3.3 Spanish terrestrial ecosystem extent accounts

The ecosystem changes between 1970 and 2015 are presented in Figure 4 and Appendix 1. We observed a considerable number of changes among ecosystems in the past decades. In Figure 4, we found the initial and final ecosystem extent, represented by the thickness of the columns, and the flows between ecosystems and the stable stock, represented by the curves. We observed how stability among ecosystems is the main trend in the curves, especially in the most representative ecosystems in extension, forests, and croplands.

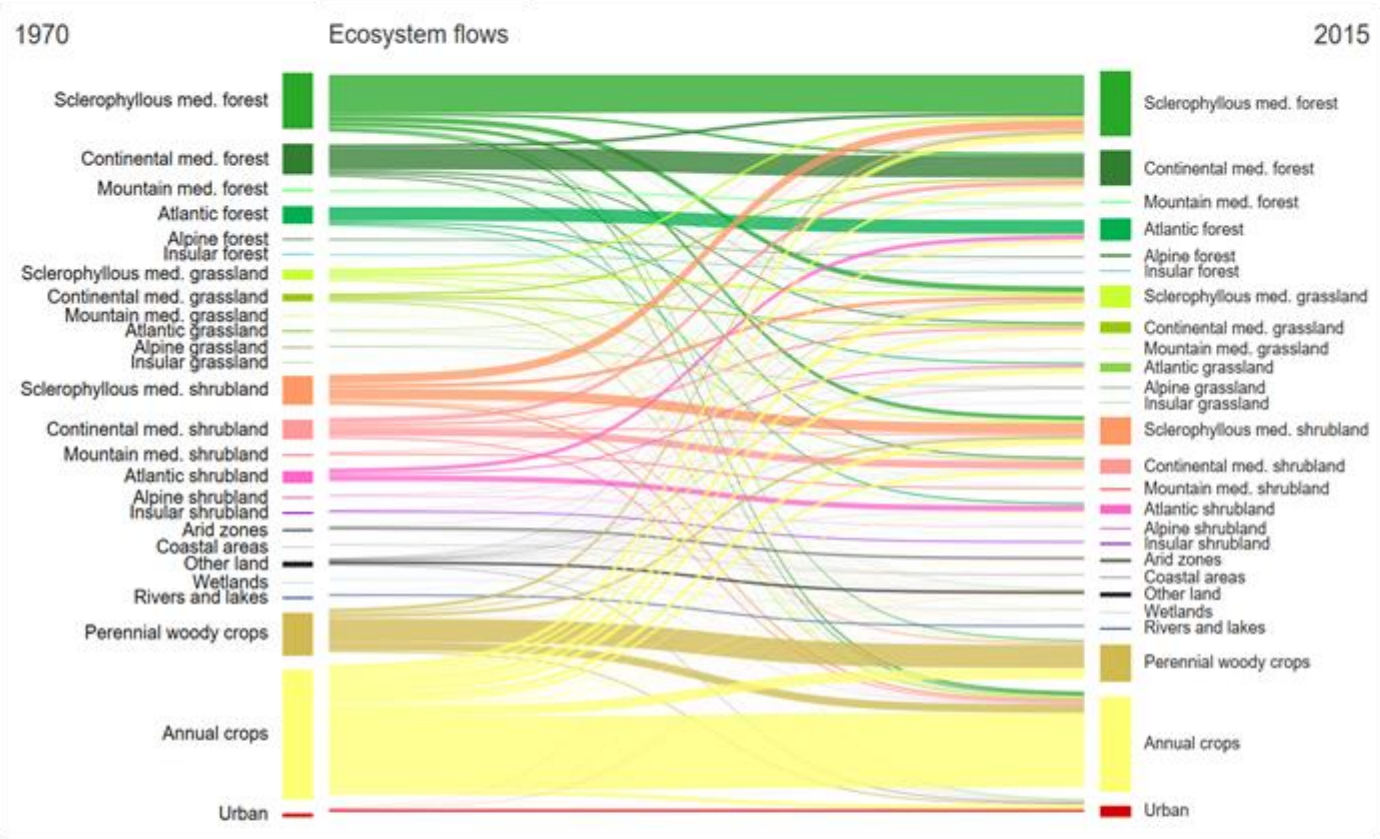

Figure 4: Representation of ecosystem extent changes and flows between 1970 and 2015 in Spain.

Conversely, grasslands and shrublands have the most ramifications, especially the Mediterranean sclerophyllous shrubland, which exhibits important changes. Finally, we observed how annual crops seem to have a greater loss of extension.

Appendix 1 summarises, in square kilometres, the changes that occurred in the extent of ecosystems between 1970 and 2015, in the form of an accounting table. This account includes information on additions, reductions, turnover, net change, and stable stock of ecosystems.

We present major ecosystem extent changes in percentage from the total accounting area for 1970–2015 (Figure 5). We observed three main trends in ecosystem extent changes based on these analyses. The first consists of those ecosystems that described a negative trend: annual croplands, perennial woody croplands, continental Mediterranean shrublands, and Atlantic shrublands. The second included ecosystems which represented a positive trend: sclerophyllous Mediterranean forests and grasslands, continental Mediterranean forests and grasslands, Atlantic forests and grasslands, and urban areas. The third category consists of ecosystems with a neutral trend: sclerophyllous Mediterranean shrublands, mountain Mediterranean forests, grasslands and shrublands, alpine forests, grasslands and shrublands, insular forests, grasslands and shrublands, arid zones, coastal areas, other lands, wetlands, rivers, and lakes. For example, we observed that the largest ecosystems in Spain are annual croplands, representing a percentage of 33.5% in 1970, decreasing to 24.5% in 2015. Additionally, we observed that the largest forest ecosystem (sclerophyllous Mediterranean) increased from 14.4% in 1970 to 16.7% in 2015. These two opposite trends in ecosystem extent, with a reduction of 9% of annual cropland and an addition of 2.2% of sclerophyllous Mediterranean forest, demonstrated the important transformation experienced by ecosystems in the last decades.

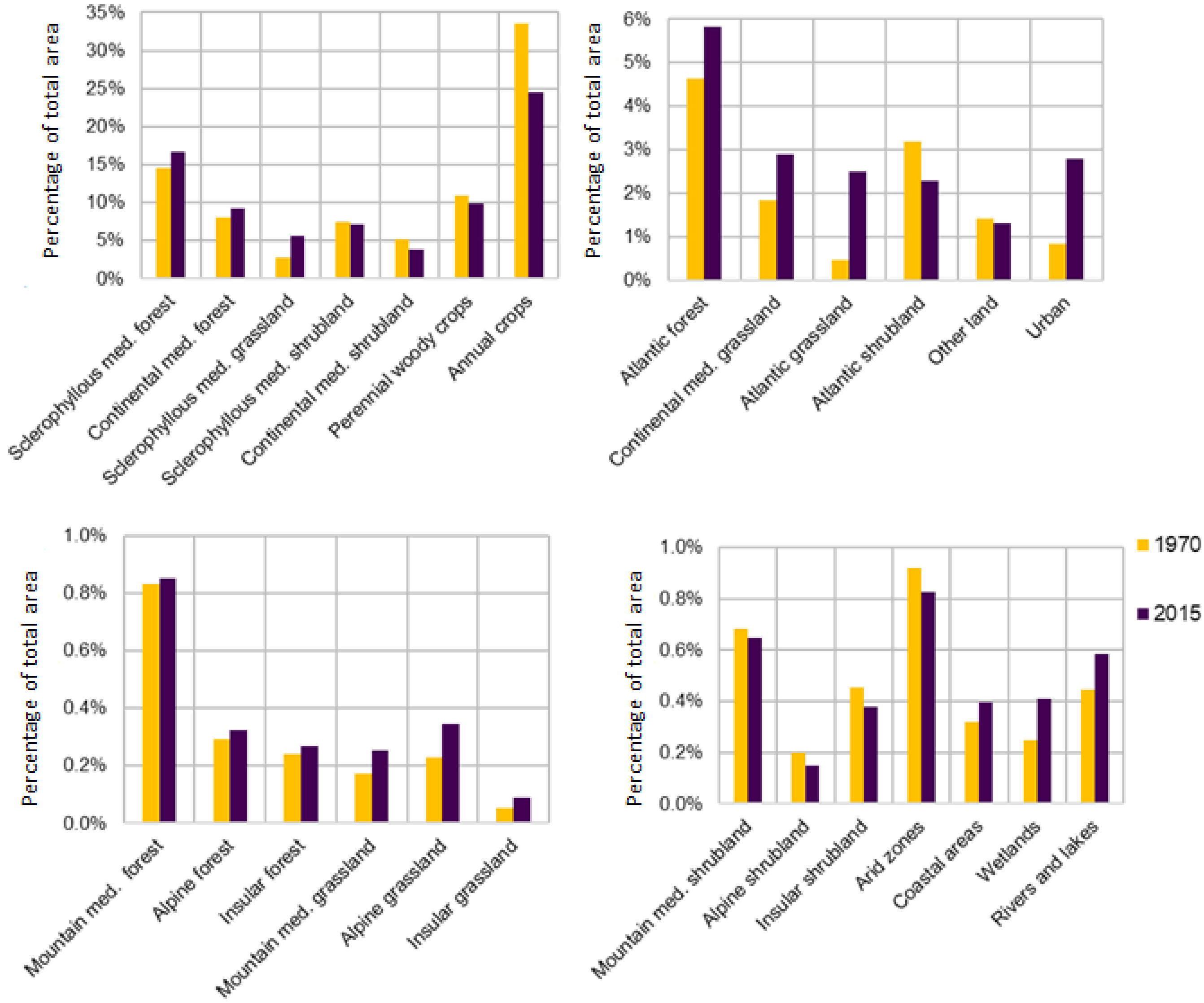

Figure 5: Ecosystem extent changes in the percentage of total area from 1970 to 2015 in Spain.

To complement the above results on major trends, we also measured turnover, net change, and stable stock to clarify these trends. Concerning net changes, we observed that most of the ecosystems showed small positive net changes (between 2.9% of the Sclerophyllous Mediterranean grassland to 0.02% of the Mountain Mediterranean forest). In contrast, some showed negative net changes, particularly annual cropland, perennial woody cropland, and continental Mediterranean shrubland (Figure 6).

The turnover shown in Figure 7 for the Spanish ecosystem between 1970 and 2015 indicates the existence of a strong turnover, which means the simultaneous additions and reductions of an ecosystem extent over another occurring more frequently than net changes. On average, the turnover is 5.8 times higher than the net change. We observed the highest turnovers in the

Mediterranean sclerophyllous area and annual croplands ecosystems. In this case, we showed how high turnover values are not synonymous with growth or decreased ecosystem extent.

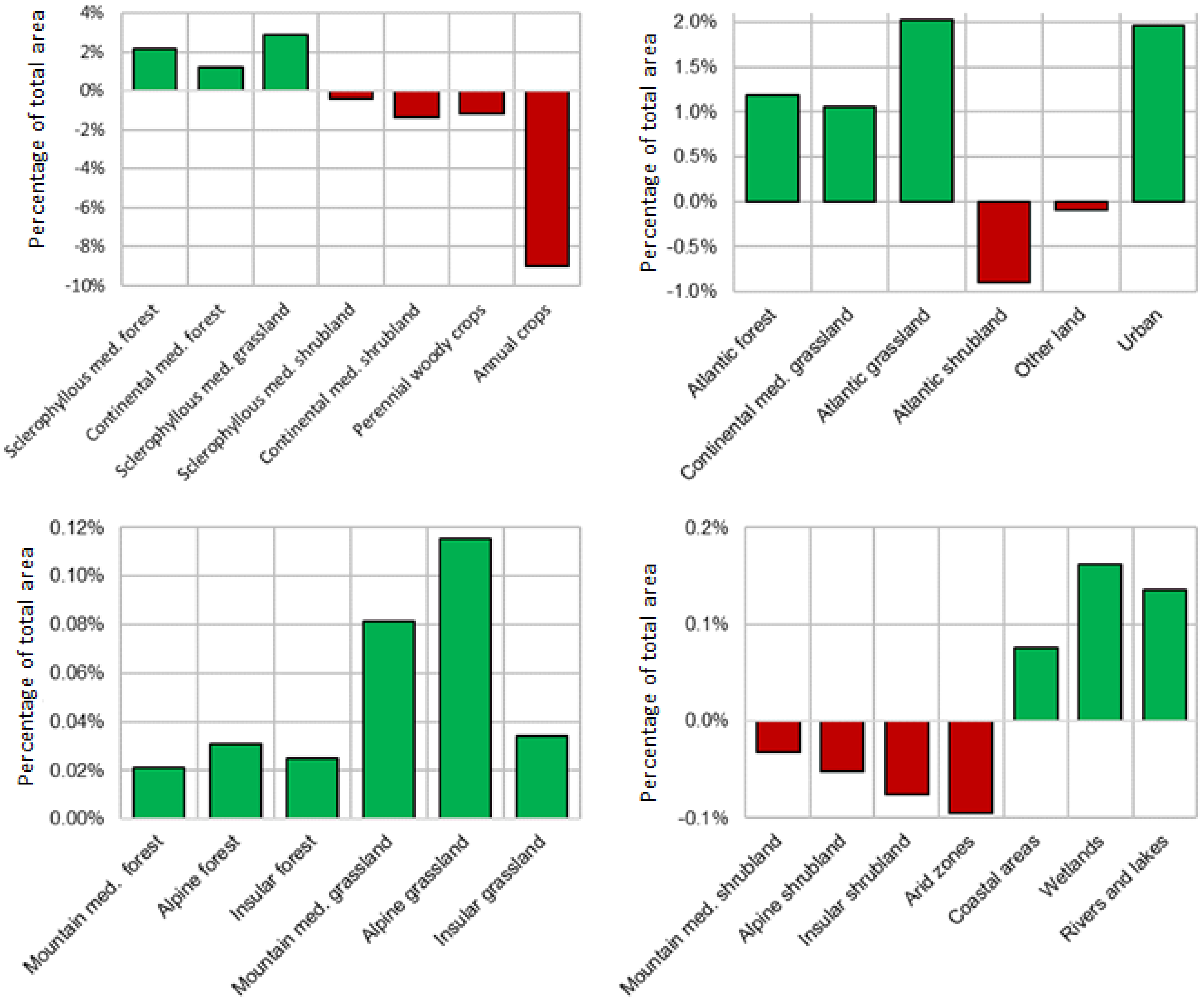

Figure 6: Ecosystem net change rates in the percentage of total area from 1970 to 2015 in Spain

The differences between turnover and net change are substantial in many ecosystems. For example, we observed how sclerophyllous Mediterranean shrubland has a 22.4 times higher turnover than the net change or Mountain Mediterranean forest, which has an 18.4 times higher turnover than the net change. However, some ecosystems have a turnover similar to net changes, highlighting urban, Atlantic grasslands, rivers, and lakes. In summary, it is observed how the trends witnessed in Figures 2 and 4 are corroborated with these analyses. We kept a high turnover rate between ecosystems, with some marked net changes.

The stable stock is shown in Figure 8. We observed how, in some ecosystems, the main extent process is the turnover. In this case, it was grassland ecosystems, especially in Atlantic grasslands or urban areas. Nevertheless, in major ecosystems, the main process of extent is reflected in the stability of ecosystems, such as forest and aquatic ecosystems.

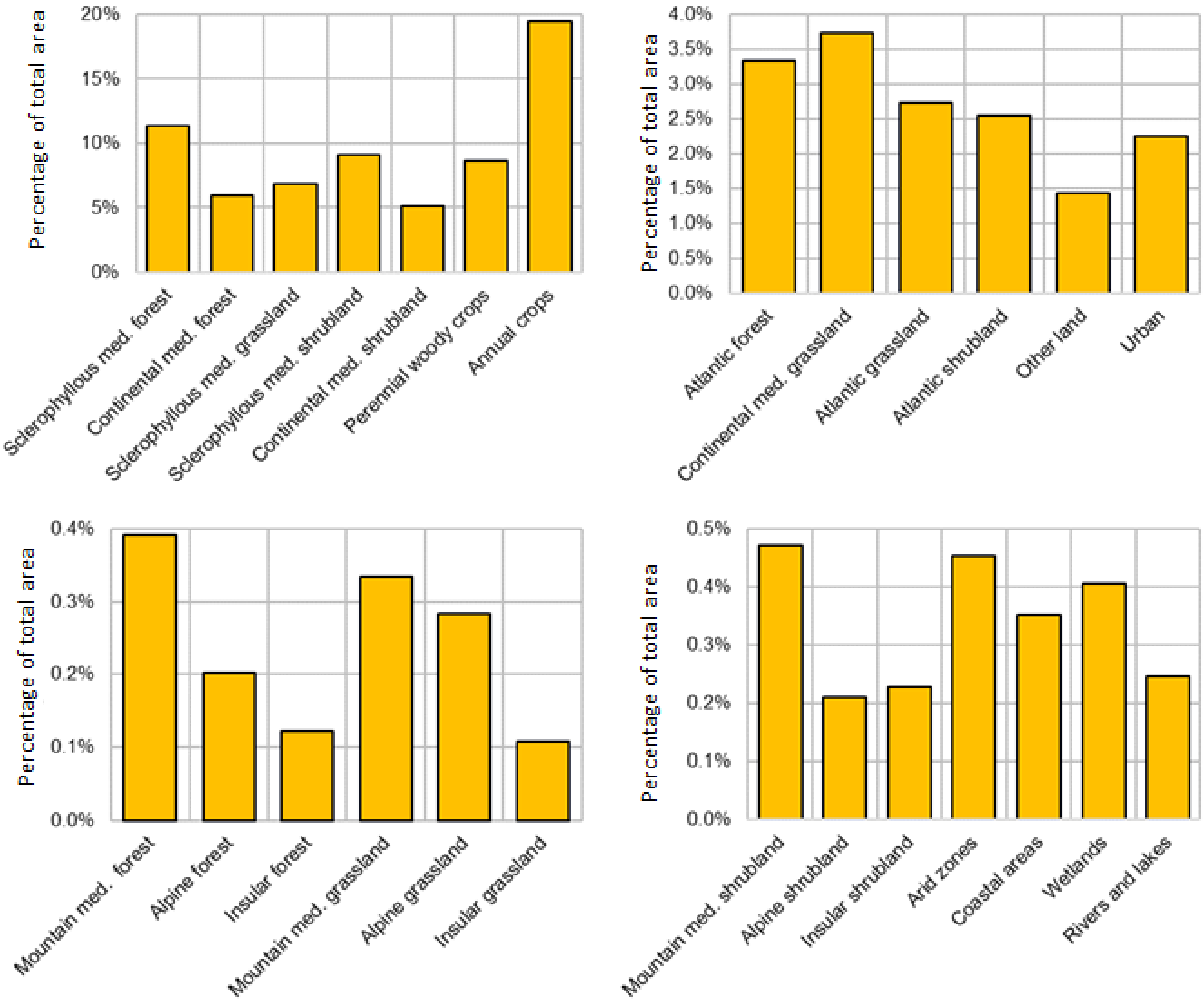

Figure 7: Ecosystem turnover rates in percentage from the total area from 1970 to 2015 in Spain

Therefore, in terms of extension, between 1970 and 2015, Spanish forests have increased their presence with a high rate of stable stock, while grasslands and shrublands have a high turnover and small stable stocks, with little positive net change rates in grasslands and negative net change rates in shrublands. Otherwise, the agroecosystems have different behaviours, with a high weight of negative net changes in the turnover of annual croplands. In contrast, turnover does not translate into great negative net changes in permanent woody croplands. Finally, urban areas have the most similar turnover and net change, with small stable stocks, possibly implying that the transformation of an urban area is unlikely.

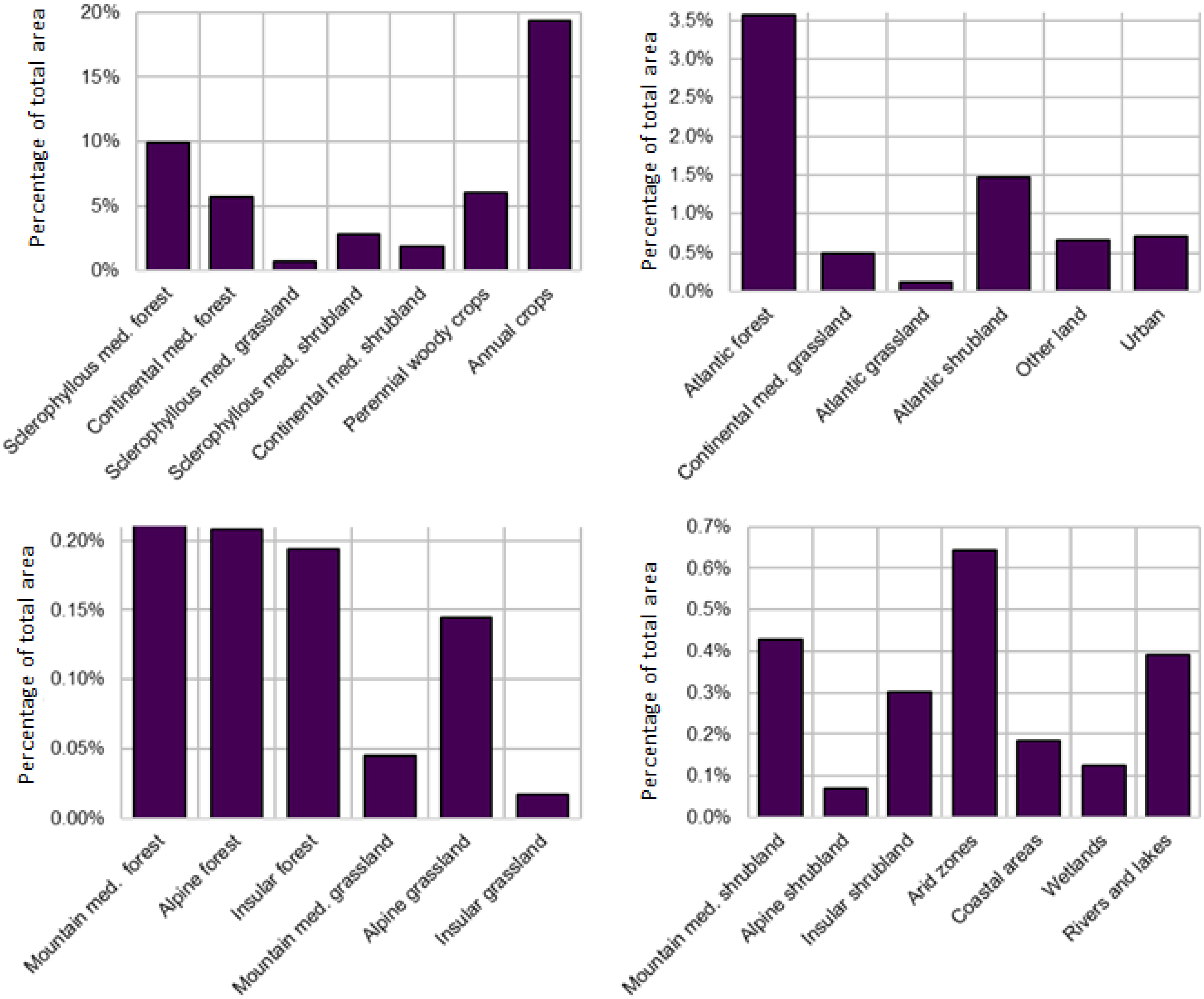

Figure 8: Ecosystem stable stock in percentage from the total area from 1970 to 2015 in Spain

Regarding these trends, Table 2 presents systematic gain flows to determine the ecosystems in which the major net extension gains occur. In this way, we see more gains than expected between different covers of the same type of ecosystem. For example, we see how in Mediterranean sclerophyllous grassland, the most significant gains have been made with Mediterranean sclerophyllous forests and shrublands. Exceptions to these transitions are the exchanges between sclerophyllous Mediterranean forests and continental Mediterranean forests, the transformation of annual crops into continental Mediterranean and Atlantic grasslands, or the transformation of alpine forests into Atlantic forests. Regarding the other positive net change, urban growth, we observed how it had gained more than expected annual croplands and perennial woody croplands, which occurred to a lesser extent in other lands and coastal areas. Concerning the speed of these transitions, we observed that the fastest transitions occurred between Atlantic grasslands (16.42) and shrublands (13.64) to Atlantic forests. Simultaneously, the slowest transition occurred between the continental Mediterranean scrub (0.12) and continental Mediterranean grassland.

Table 2: Most systematic gain transitions between Spanish terrestrial ecosystems. Value of the systematic change in percentage of the total (column observed less expected), the significance of the change (column of the difference divided by the expected)

Gain transitions		Observed minus expected	Difference divided by expected
Ecosystem in 2015	Ecosystem in 1970		
Sclerophyllous med. grassland	Sclerophyllous med. forest	0.72	0.99
	Sclerophyllous med. shrubland	0.65	1.72
	Arid zones	0.02	0.43
Sclerophyllous med. forest	Sclerophyllous med. shrubland	1.74	2.93
	Sclerophyllous med. grassland	0.58	2.67
	Continental med. forest	0.24	0.37
Atlantic grassland	Annual crops	0.58	0.73
	Atlantic shrubland	0.4	5.28
	Atlantic forest	0.31	2.77
Urban	Annual crops	0.38	0.53
	Perennial woody crops	0.16	0.68
	Other land	0.1	3.22
	Coastal areas	0.04	5.45
Continental med. forest	Continental med. shrubland	0.95	4.7
	Continental med. grassland	0.4	5.7
	Sclerophyllous med. forest	0.2	0.36
Atlantic forest	Atlantic shrubland	1.03	13.64
	Atlantic grassland	0.18	16.42
	Alpine forest	0.02	2.93
Continental med. grassland	Continental med. shrubland	0.53	0.12
	Continental med. forest	0.25	1.27
	Annual crops	0.17	0.21

Concerning the more than expected loss transition, we observed multiple sources of significant loss in annual crops, with the most important being the exchange between the two types of croplands. In addition, we observed various sources of substantial loss in annual crops, with the most significant being the exchange between annual croplands and perennial croplands. In contrast, we observed that the other transitions of losses of annual crops have been towards different types of grasslands, urban areas, and Mediterranean continental shrublands. Conversely, the losses of continental Mediterranean shrubland occurred more than expected in the other ecosystems of a continental Mediterranean and sclerophyllous Mediterranean shrubland. Finally, permanent woody crops, in addition to the exchange with annual crops, have undergone more than expected transformation in urban areas and the sclerophyllous Mediterranean shrublands.

Table 3: Most systematic loss transitions between Spanish terrestrial ecosystems. Value of the systematic change in percentage of the total (column observed less expected), the significance of the change (column of the difference divided by the expected)

Loss transitions		Observed minus expected	Difference divided by expected
Ecosystem in 1970	Ecosystem in 2015		
Annual crops	Perennial woody crops	0.93	0.50
	Atlantic grassland	0.92	1.95
	Sclerophyllous med. grassland	0.61	0.57
	Urban	0.56	1.06
	Continental med. grassland	0.45	0.82
	Continental med. shrubland	0.18	0.25
Continental med. shrubland	Continental med. forest	0.84	2.66
	Continental med. grassland	0.56	5.78
	Sclerophyllous med. shrubland	0.22	0.90
Perennial woody crops	Annual crops	0.89	0.67
	Sclerophyllous med. shrubland	0.33	0.85
	Urban	0.24	1.57

4. Discussion

4.1 Main patterns of ecosystem extent accounts

We found three main patterns in extent account changes: (i) an increase in forest ecosystems, (ii) an important decrease in agroecosystems (especially annual croplands), and (iii) a constant development of urban areas. Compared to global and different national scales, these trends show similarities and differences. For example, forest ecosystems experience a decrease at the global level, while croplands show an increase (Li et al., 2018). However, the pattern is contrary in developed countries in Western Europe or North America, such as in Spain (Chao et al., 2018; Nowosad et al., 2019). The other net changes in ecosystems in Spain are similar to the global view, such as a significant increase in urban areas in the 2000s or the shrubland's irregular negative net changes (Li et al., 2018).

More specifically, if we compare our results with other European countries, we find a trend similar to our results, with a net increase in forest ecosystems and a net decrease in agroecosystems around the 39 countries of Europe since 2000 (Petersen, 2019). One of the principal differences is grassland ecosystems, which show a negative trend in Europe, while our results describe a growing pattern. Moreover, we identify small net changes that hide a high turnover between ecosystems, which misinterpret processes and flows of the ecosystems (Yuan et al., 2016) in Spain. These discrepancies in the values could be due to the later development of the Spanish economy compared to other developed countries (Santos-Martín, González García-Mon, et al., 2019). Afforestation improved after Spain joined the European Economic Community in 1986 because of policy and market changes that enhanced the transformation of the agricultural sector (Fernández-Nogueira and Corbelle-Rico, 2018).

Furthermore, climate and other socioeconomic factors, such as population density and accessibility to cities, are related to the dynamics of ecosystems (Hellwig et al., 2019). However, these factors do not affect all ecosystems similarly. On the one hand, species sensitive to anthropogenic changes are abundant. On the other hand, disturbance-tolerant and generalist species may prosper, although technological development and policy implementation could reduce the human impact (Luck, 2007). Warmer temperature regimens and more frequent droughts impact natural ecosystems that are less adapted to these conditions (Novillo et al., 2019). These changes have consequences in the atmospheric behaviour, for example, in North Atlantic Jet's trajectory (Trouet et al., 2018), triggering changes in ecosystems and favouring the expansion of sclerophyllous ecosystems over continental, Atlantic, and alpine ecosystems.

Regarding changes in the agroecosystem, our results show that the only significant agricultural decrease flow occurs in grasslands because of the abandonment of croplands in Europe since the second half of the twentieth century (Lasanta et al., 2017). These cropland negative net changes could cause problems in food security, which could be compensated by intensifying existing production (d'Amour et al., 2017). However, this intensification could result in numerous damaging environmental impacts on soils, water, and biodiversity, increased crop ecological footprints, and the development of social conflicts and inequalities (Paul et al., 2017; Santos-Martín, Zorrilla-Miras, et al., 2019)

The abandonment of croplands impacts biodiversity, which has a negative effect on migrant farmland species and is beneficial for non-migratory forest species (Gradinaru et al., 2020). However, the abandonment process affects cropland ecosystem services, decreasing fire regulation capacity while increasing fire frequency (Bajocco et al., 2012).

Concerning the urbanisation process, and in line with our results, Li et al. (2018) and Petersen (2019), on a global and European scale, respectively, reported a substantial development during the first decade of the twenty-first century. Our results in Spain are explained by the expansion of the Spanish economy based on the increased infrastructure and housing construction, which is linked to the liberalisation of land laws in 1998 (Santos-Martín, González García-Mon, et al., 2019). As in global change, the urbanisation process occurred in croplands (van Vliet, 2019) with substantial direct impacts on global biodiversity hotspots and carbon pools due to reductions in local primary productivity. However, this process has not been carried out on more natural ecosystems, such as forests, typically occurring in other areas worldwide (Li et al., 2018). Furthermore, depending on the climate and population density zonal characteristics, this urbanisation process can increase or decrease greenhouse gas emissions per capita (Seto et al., 2012).

4.2 Applications, uncertainties, and challenges in the compilation of the extent accounts.

The potential applications of ecosystem extent accounting in policy and decision-making are diverse (Nagendra and Ostrom, 2012). The main objective is to establish structured, natural capital information consistent with economic data and institutionally embedded and sustained by national government institutions (Ruijs et al., 2019). Furthermore, they should promote conceptual coherence to develop a comprehensive, convergent, and viable measurement system that integrates ecosystem values into national planning (Bordt, 2018), facilitating applications across a range of spatial and organisational scales (Keith et al., 2020). For instance, information on how ecosystems change over time allows multiple actors to make informed decisions about restoration priorities, management plans for natural protected areas, urban and peri-urban connectivity design, and so on. One key element of ecosystem accounting is that data need to be managed by a single authoritative source that integrates information across levels of governance, scales, and resources (Vardon et al., 2016). These accounts supply ecosystems' location and geophysical context to provide insights into the effects of different land use and environmental characteristics correlated with different policy and management actions (Petersen, 2019). To achieve the usefulness of the extent and other ecosystems accounts for the public and private sector, these accounts need to demonstrate their applicability and complement other environmental monitoring systems, such as climate change or air and water monitoring systems (Hein, Remme, et al., 2020). One key issue in integrating this information is to approve an international ecosystem classification, such as UICN Global Ecosystem Typology (Keith et al., 2020), EUNIS classification (Moss, 2014), and global ecological land units (Sayre et al., 2014). This connection between national and international standard methods is critical for providing a reliable global monitoring system for ecosystems (Bogaart et al., 2019).

Our validation process has similar results to other studies (Büttner and Maucha, 2006; Tsendbazar et al., 2016; Vilar et al., 2019). However, in our case, the principal source of misclassifications is the differences in each dataset's descriptions to classify the categories (Vilar et al., 2019). For instance, in wetlands, LULUCF defines these ecosystems as land covered or saturated by water for all or part of the year (IPCC, 2006), which includes rivers and lakes under this category. Nevertheless, LUCAS separates rivers and lakes into different categories (Ballin et al., 2018). A potential explanation for these results is that wetland ecosystems have few examples points in LUCAS (17 samples for 2015), the first source of confusion in this category, including rivers and lake ecosystems. Similarly, very few examples of arid zones and coastal areas exist in LUCAS, and these are mostly confused with the sclerophyllous Mediterranean shrubland. Finally, a main source of misinterpretation was found between grassland and shrubland ecosystems concerning other lands.

Otherwise, different spatial resolutions may also confuse. A higher resolution implies better characterisation because smaller features are best represented at higher resolutions in heterogeneous landscapes (Pérez-Hoyos et al., 2017). In our case, the heterogeneous spatial pattern is observed in other lands, and arid zones are often confused with extended and homogeneous categories, such as sclerophyllous Mediterranean shrubland and annual cropland. This phenomenon has been reported in other studies (Herold et al., 2008; Tsendbazar et al., 2016).

Extent accounts need to find a position against other environmental monitoring systems already in place, such as the LULUCF reporting (Hein et al, 2020). At the same time, it became clear that combining different data sources can provide important new insights. For example, the good fix observed in forest and cropland ecosystems is related to the multi-source approach of LULUCF cartography, which improves the suitability of the categories based on local data (Pérez-Hoyos et al., 2020; Sturari et al., 2017; Xu et al., 2017). This case is based on the Spanish forest map and the geographic information system of agricultural plots (SIGPAC). Additionally, the extent account could be used to model and map forest stands using data from the National Forest Inventories. As a result, more spatially detailed estimates of stocks, harvest and regrowth are reported compared to the forest inventory output by itself (Hein, Remme, et al., 2020).

Finally, we hope that the proposed approach and application of ecosystem extent account in Spain will be shared and tested in other countries. Furthermore, our method consists of an automatic technical system for applications related to urban planning, green infrastructure development, and natural capital assessment, among multiple uses for territory management and policies.

5. Conclusions

In the present study, we developed a national ecosystem extent account approach based on an automatic system to ensure the replicability and usefulness of our research for different scales, stakeholders, and nationwide utilities.

We considered a long time series (1975-2018) to assess ecosystem extent changes in Spain. Our results showed different patterns like an increase in forest ecosystems or an important decrease in the agroecosystem.

It is vital to promote accounting initiatives for ecosystems and natural capital to monitor the effects occurring in ecosystems through a reproducible, comparable, and standardised methodology. Currently, there are some initiatives at different scales to develop ecosystem accounting, especially the United Nations SEEA-EA framework. However, it isn't easy to apply an ecosystem accounting system useful to different stakeholders without sufficient institutional support.

We conducted an extent account based on the LULUCF spatial database to test the proposed approach. Using LULUCF as the basic spatial unit and the types of ecosystems in the Spanish National Ecosystem Assessment demonstrated high accuracy with ground observation data used for ecosystem identification purposes. In addition, we checked how a multi-source Spanish dataset with a high spatial resolution merged with European spatial scale data improved fit measures.

These results are the first step in developing the other ecosystem accounts proposed in the SEEA-EA framework. Therefore, we considered that our ecosystem account method and the results obtained at a national level are crucial for a strong knowledge of ecosystem dynamics and their implications for ecosystem conditions at a national level.

Funding

The preparation of this paper was supported by the EU MAIA project (Project Number H2020-SC5-2018-1. Grant Number 817527). We also acknowledge funding from the Rey Juan Carlos University under predoctoral contracts program (Ref.: PREDOC21-029).

6. Bibliography

Abatzoglou, J. T., Dobrowski, S. Z., Parks, S. A., and Hegewisch, K. C. (2018). TerraClimate, a high-resolution global dataset of monthly climate and climatic water balance from 1958–2015. *Scientific data*, *5*(1), 1-12.

Alo, C. A., and Pontius Jr, R. G. (2008). Identifying systematic land-cover transitions using remote sensing and GIS: the fate of forests inside and outside protected areas of Southwestern Ghana. *Environment and Planning B: Planning and Design*, *35*(2), 280-295.

Alonso Moya, M. J., Roldán Martínez, Á., Lerner Cuzzi, M., and Fernández Ramiro, M. (2020). Cartografía del Sistema Español de Inventario de Emisiones (SEI). Serie Cartográfica LULUCF. *XI Jornadas Ibéricas de Infraestructuras de Datos Espaciales (JIIDE2020)*.

Arets, E. J. M. M., Van Der Kolk, J. W. H., Hengeveld, G. M., Lesschen, J. P., Kramer, H., Kuikman, P. J., and Schelhaas, M.-J. (2019). Greenhouse gas reporting of the LULUCF sector in the Netherlands. *Greenhouse gas reporting of the LULUCF sector in the Netherlands: Methodological background, update 2019*.

Bajocco, S., De Angelis, A., Perini, L., Ferrara, A., and Salvati, L. (2012). The impact of land use/land cover changes on land degradation dynamics: a Mediterranean case study. *Environmental management*, *49*(5), 980-989.

Ballin, M., Barcaroli, G., Masselli, M., and Scarnó, M. (2018). Redesign sample for Land Use/Cover Area frame Survey (LUCAS) 2018 : 2018 edition. *Eurostat: Statistical working papers*.

Banerjee, O., Bagstad, K. J., Cicowiez, M., Dudek, S., Horridge, M., Alavalapati, J. R., . . . Rutebuka, E. (2020). Economic, land use, and ecosystem services impacts of Rwanda's Green Growth Strategy: An application of the IEEM+ ESM platform. *Science of The Total Environment*, *729*, 138779.

Bogaart, P., Chan, J. Y., Horlings, H., Keith, D., Larson, T., Sayre, R., . . . Soulard, F. (2019). SEEA EEA Revision Expert Consultation Working group 1: Spatial units. *SEEA EEA Revision Expert Consultation Working group 1: Spatial units*.

Bordt, M. (2018). Discourses in ecosystem accounting: a survey of the expert community. *Ecological Economics*, *144*, 82-99.

Boyd, J., and Banzhaf, S. (2007). What are ecosystem services? The need for standardized environmental accounting units. *Ecological economics*, *63*(2-3), 616-626.

Büttner, G., and Maucha, G. (2006). The thematic accuracy of Corine land cover 2000. Assessment using LUCAS (land use/cover area frame statistical survey). *European Environment Agency: Copenhagen, Denmark*, *7*.

Campos, P., Caparrós, A., Oviedo, J. L., Ovando, P., Álvarez-Farizo, B., Díaz-Balteiro, L., . . . Herruzo, A. C. (2019). Bridging the gap between national and ecosystem accounting application in Andalusian Forests, Spain. *Ecological Economics*, *157*, 218-236.

Chao, Y., Ciais, P., and Li, W. (2018). Smaller global and regional carbon emissions from gross land use change when considering sub-grid secondary land cohorts in a global dynamic vegetation model. *Biogeosciences*, *15*(4), 1185.

Crossman, N. D., Burkhard, B., Nedkov, S., Willemen, L., Petz, K., Palomo, I., . . . Boyanova, K. (2013). A blueprint for mapping and modelling ecosystem services. *Ecosystem services*, *4*, 4-14.

d'Amour, C. B., Reitsma, F., Baiocchi, G., Barthel, S., Güneralp, B., Erb, K.-H., . . . Seto, K. C. (2017). Future urban land expansion and implications for global croplands. *Proceedings of the National Academy of Sciences*, *114*(34), 8939-8944.

Daily, G. C., and Matson, P. A. (2008). Ecosystem services: from theory to implementation. *Proceedings of the national academy of sciences*, *105*(28), 9455-9456.

Dvarskas, A. (2019). Experimental ecosystem accounting for coastal and marine areas: A pilot application of the SEEA-EEA in Long Island coastal bays. *Marine Policy*, *100*, 141-151.

Eppink, F. V., Werntze, A., Mäs, S., Popp, A., and Seppelt, R. (2012). Land management and ecosystem services how collaborative research programmes can support better policies. *GAIA-Ecological Perspectives for Science and Society*, *21*(1), 55-63.

Esen, S. E., and Hein, L. (2020). Development of SEEA water accounts with a hydrological model. *Science of the Total Environment*, *737*, 140168.

ESRI. (2020). Arcgis Pro 2.8.

European Environmental Agency (2016). Biogeographical regions in Europe.

Eurostat. (2015). LUCAS 2012 (Land Use/Cover Area Frame Survey). Quality Report. In.

Felicísimo, Á. M., Muñoz, J. s., Villalba, C. J., and Mateo, R. G. (2011). *Impactos, vulnerabilidad y adaptación al cambio climático de la biodiversidad española: Flora y vegetación: Proyecciones de las áreas de distribución potencial de la flora amenazada y las especies forestales de la España penínsular por efecto del cambio climático*. Oficina Española de Cambio Climático, Ministerio de Medio Ambiente y Medio ….

Fernández-Nogueira, D., and Corbelle-Rico, E. (2018). Land Use Changes in Iberian Peninsula 1990-2012. *Land*, *7*(3), 99.

Fuchs, R., Herold, M., Verburg, P. H., Clevers, J. G., and Eberle, J. (2015). Gross changes in reconstructions of historic land cover/use for Europe between 1900 and 2010. *Global change biology*, *21*(1), 299-313.

Fuchs, R., Schulp, C. J., Hengeveld, G. M., Verburg, P. H., Clevers, J. G., Schelhaas, M. J., and Herold, M. (2016). Assessing the influence of historic net and gross land changes on the carbon fluxes of Europe. *Global change biology*, *22*(7), 2526-2539.

Funk, C., Peterson, P., Landsfeld, M., Pedreros, D., Verdin, J., Shukla, S., . . . Hoell, A. (2015). The climate hazards infrared precipitation with stations—a new environmental record for monitoring extremes. *Scientific data*, *2*(1), 1-21.

Gradinaru, S. R., Ioja, C. I., Vanau, G. O., and Onose, D. A. (2020). Multi-Dimensionality of Land Transformations: From Definition to Perspectives on Land Abandonment. *Carpathian Journal of Earth and Environmental Sciences*, *15*(1), 167-177.

Grunewald, K., Schweppe-Kraft, B., Syrbe, R.-U., Meier, S., Krüger, T., Schorcht, M., and Walz, U. (2020). Hierarchical classification system of Germany's ecosystems as basis for an ecosystem accounting-methods and first results. *One Ecosystem*(2).

Harris, C., Millman, K., van der Walt, S., Gommers, R., Virtanen, P., Cournapeau, D., . . . Berg, S. (2020). Smith 474 NJ. *Kern R, Picus M, Hoyer S, van Kerkwijk MH, Brett M, Haldane A, del R'ıo JF, Wiebe M, Peterson P, G'erard-475 Marchant P, et al. Array programming with NumPy. Nature*, *585*(7825), 357-362.

Hein, L., Bagstad, K. J., Obst, C., Edens, B., Schenau, S., Castillo, G., . . . Bordt, M. (2020). Progress in natural capital accounting for ecosystems. *Science*, *367*(6477), 514-515.

Hein, L., Remme, R. P., Schenau, S., Bogaart, P. W., Lof, M. E., and Horlings, E. (2020). Ecosystem accounting in the Netherlands. *Ecosystem Services*, *44*, 101118.

Hellwig, N., Walz, A., and Markovic, D. (2019). Climatic and socioeconomic effects on land cover changes across Europe: Does protected area designation matter? *PloS one*, *14*(7).

Herold, M., Mayaux, P., Woodcock, C. E., Baccini, A., and Schmullius, C. (2008). Some challenges in global land cover mapping: An assessment of agreement and accuracy in existing 1 km datasets. *112*(Generic), 2538-2556. https://doi.org/https://doi.org/10.1016/j.rse.2007.11.013

Innerbichler, F., Kreisel, A., and Gruber, C. (2021). Coastal Zones Nomenclature Guideline. *European Environment Agency*.

IPBES. (2019). Report of the plenary of the intergovernmental science-policy platform con biodiversity and ecosystem services on the work of its seventh session. Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the intergovernmental science-policy platform on biodiversity and ecosystem services. Retrieved from.

IPCC. (2006). 2006 IPCC guidelines for national greenhouse gas inventories. *Institute for Global Environmental Strategies, Hayama, Kanagawa, Japan*.

Keith, D., A.Ferrer-Paris, J., R.Nicholson, E., and Kingsford, R. (2020). IUCN Global Ecosystem Typology 2.0 : descriptive profiles for biomes and ecosystem functional groups. *Gland, Switzerland : IUCN, 2020*. https://doi.org/https://doi.org/10.2305/IUCN.CH.2020.13.en

Keith, H., Vardon, M., Stein, J. A., Stein, J. L., and Lindenmayer, D. (2017). Ecosystem accounts define explicit and spatial trade-offs for managing natural resources. *Nature ecology and evolution*, *1*(11), 1683-1692.

La Notte, A., and Rhodes, C. (2020). The theoretical frameworks behind integrated environmental, ecosystem, and economic accounting systems and their classifications. *Environmental Impact Assessment Review*, *80*(Generic), 106317. https://doi.org/https://doi.org/10.1016/j.eiar.2019.106317

La Notte, A., Vallecillo, S., Marques, A., and Maes, J. (2019). Beyond the economic boundaries to account for ecosystem services. *Ecosystem Services*, *35*(Generic), 116-129. https://doi.org/https://doi.org/10.1016/j.ecoser.2018.12.007

Lai, T.-Y., Salminen, J., Jäppinen, J.-P., Koljonen, S., Mononen, L., Nieminen, E., . . . Oinonen, S. (2018). Bridging the gap between ecosystem service indicators and ecosystem accounting in Finland. *Ecological Modelling*, *377*, 51-65.

Landis, J. R., and Koch, G. G. (1977). The measurement of observer agreement for categorical data. *Biometrics*, 159-174.

Lasanta, T., Arnáez, J., Pascual, N., Ruiz-Flaño, P., Errea, M. P., and Lana-Renault, N. (2017). Space–time process and drivers of land abandonment in Europe. *149*(Generic), 810-823. https://doi.org/https://doi.org/10.1016/j.catena.2016.02.024

Lee, C. K. F., Nicholson, E., Duncan, C., and Murray, N. J. (2020). Estimating changes and trends in ecosystem extent with dense time-series satellite remote sensing. *Conservation Biology*.

Li, W., MacBean, N., Ciais, P., Defourny, P., Lamarche, C. l., Bontemps, S., . . . Peng, S. (2018). Gross and net land cover changes in the main plant functional types derived from the annual ESA CCI land cover maps (1992â€"2015).

Luck, G. W. (2007). A review of the relationships between human population density and biodiversity. *Biological Reviews*, *82*(4), 607-645.

MA. (2005). Millennium ecosystem assessment. *Washington, DC: New Island*, *13*, 520.

Maes, J., Teller, A., Erhard, M., Liquete, C., Braat, L., Berry, P., . . . Santos, F. (2013). Mapping and Assessment of Ecosystems and their Services. *An analytical framework for ecosystem assessments under action*, *5*, 1-58.

McCARTY, J. P. (2001). Ecological consequences of recent climate change. *Conservation biology*, *15*(2), 320-331.

Moss, D. (2014). EUNIS habitat classification: a guide for users. *European Topic Centre on Biological Diversity*.

Mäler, K.-G., Aniyar, S., and Jansson, Å. (2008). Accounting for ecosystem services as a way to understand the requirements for sustainable development. *Proceedings of the National Academy of Sciences*, *105*(28), 9501-9506.

Nagendra, H., and Ostrom, E. (2012). Polycentric governance of multifunctional forested landscapes. *International Journal of the Commons*, *6*(2).

Nedkov, S., Doncheva, S., and Markov, B. (2016). Mapping of ecosystems in Bulgaria based on MAES typology. Seminar of Ecology-2016, with international participation: Proceedings,

Novillo, C. J., Arrogante-Funes, P., and Romero-Calcerrada, R. (2019). Recent NDVI trends in Mainland Spain: Land-cover and phytoclimatic-type implications. *ISPRS International Journal of Geo-Information*, *8*(1), 43.

Nowosad, J., Stepinski, T. F., and Netzel, P. (2019). Global assessment and mapping of changes in mesoscale landscapes: 1992–2015. *78*(Generic), 332-340. https://doi.org/https://doi.org/10.1016/j.jag.2018.09.013

Obst, C., Hein, L., and Edens, B. (2016). National accounting and the valuation of ecosystem assets and their services. *Environmental and Resource Economics*, *64*(1), 1-23.
Obst, C. G. (2015). Reflections on natural capital accounting at the national level: Advances in the system of environmental-economic accounting. *Sustainability Accounting, Management and Policy Journal*.
Paul, C., Weber, M., and Knoke, T. (2017). Agroforestry versus farm mosaic systems–Comparing land-use efficiency, economic returns and risks under climate change effects. *Science of the Total Environment*, *587*, 22-35.
Petersen, J.-E. (2019). Natural capital accounting in support of policymaking in Europe. *EEA report*, *26/2018*.
Pontius Jr, R. G. (2019). Component intensities to relate difference by category with difference overall. *International Journal of Applied Earth Observation and Geoinformation*, *77*, 94-99.
Pontius Jr, R. G., Shusas, E., and McEachern, M. (2004). Detecting important categorical land changes while accounting for persistence. *Agriculture, Ecosystems and Environment*, *101*(2-3), 251-268.
Powers, D. M. (2020). Evaluation: from precision, recall and F-measure to ROC, informedness, markedness and correlation. *arXiv preprint arXiv:2010.16061*.
Pérez-Hoyos, A., Rembold, F., Kerdiles, H., and Gallego, J. (2017). Comparison of global land cover datasets for cropland monitoring. *Remote Sensing*, *9*(11), 1118.
Pérez-Hoyos, A., Udías, A., and Rembold, F. (2020). Integrating multiple land cover maps through a multi-criteria analysis to improve agricultural monitoring in Africa. *88*(Generic), 102064. https://doi.org/https://doi.org/10.1016/j.jag.2020.102064
Regulation (EU) 2018/841 of the European Parliment and of the Council of 30 May 2018 on the inclusion of greenhouse gas emissions and removals from land use, land use change and forestry in the 2030 climate and energy framework, and amending Regulation (EU) No 525/2013 and Decision No 529/2013/EU, 1-25 (2018).
Rivas-Martínez, S. (1983). Pisos bioclimáticos de España. *Lazaroa*, *5*(1983), 33-43.
Ruijs, A., Vardon, M., Bass, S., and Ahlroth, S. (2019). Natural capital accounting for better policy. *Ambio*, *48*(7), 714-725.
Santos-Martín, F., González García-Mon, B., González, J. A., Iniesta-Arandia, I., Garcí-a-Llorente, M., Montes, C., . . . Benayas, J. (2019). Identifying past social-ecological thresholds to understand long-term temporal dynamics in Spain. *Ecology and Society*, *24*(2).
Santos-Martín, F., Zorrilla-Miras, P., García-Llorente, M., Quintas-Soriano, C., Montes, C., Benayas, J., . . . Paracchini, M. L. (2019). Identifying win-win situations in agricultural landscapes: an integrated ecosystem services assessment for Spain. *Landscape Ecology*, *34*(7), 1789-1805.
Sayre, R., Dangermond, J., Frye, C., Vaughan, R., Aniello, P., Breyer, S., . . . Derrenbacher, W. (2014). A new map of global ecological land units—an ecophysiographic stratification approach. *Washington, DC: Association of American Geographers*, *46*.
Schipper, A., Tillmanns, M., Giesen, P., and van der Esch, S. (2017). Compiling biodiversity accounts with the GLOBIO model.
Seto, K. C., Güneralp, B., and Hutyra, L. R. (2012). Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. *Proceedings of the National Academy of Sciences*, *109*(40), 16083-16088.
SNEA. (2014). Spanish national ecosystem assessment. Fundación Biodiversidad. Ministerio de Medio Ambiente, y Medio rural y Marino.
Sturari, M., Frontoni, E., Pierdicca, R., Mancini, A., Malinverni, E. S., Tassetti, A. N., and Zingaretti, P. (2017). Integrating elevation data and multispectral high-resolution images for an improved hybrid Land Use/Land Cover mapping. *European Journal of Remote Sensing*, *50*(1), 1-17. https://doi.org/10.1080/22797254.2017.1274572
Tamame, M., Lorenzo, A., and Lindmayer, A. (2018). Riparian

Zones Nomenclature Guideline. *European Environment Agency*.
Trouet, V., Babst, F., and Meko, M. (2018). Recent enhanced high-summer North Atlantic Jet variability emerges from three-century context. *Nature communications*, *9*(1), 1-9.
Tsendbazar, N. E., de Bruin, S., Mora, B., Schouten, L., and Herold, M. (2016). Comparative assessment of thematic accuracy of GLC maps for specific applications using existing reference data. *44*(Generic), 124-135. https://doi.org/https://doi.org/10.1016/j.jag.2015.08.009

UNEP-WCMC, I. (2017). Experimental Ecosystem Accounts for Uganda. *United Nations Environment Programme, Cambridge*.

United Nations, European Union, Food and Agriculture Organization of the United Nations, International Monetary Fund, Organisation for Economic Co-operation and Development, and World Bank. (2014). System of Environmental-Economic Accounting 2012--Central Framework.

United Nations, European Union, Food and Agriculture Organization of the United Nations, International Monetary Fund, Organisation for Economic Co-operation and Development, and World Bank. (2021). System of Environmental-Economic Accounting—

Ecosystem Accounting (SEEA EA). White cover publication, pre-edited text subject to official editing.

United Nations Statistical Commission. (2021). Report of the Commission on its fifty-second session.

van Vliet, J. (2019). Direct and indirect loss of natural area from urban expansion. *Nature Sustainability*, *2*(8), 755-763.

Vardon, M., Burnett, P., and Dovers, S. (2016). The accounting push and the policy pull: balancing environment and economic decisions. *Ecological Economics*, *124*, 145-152.

Vačkářů, D., and Grammatikopoulou, I. (2019). Toward development of ecosystem asset accounts at the national level. *Ecosystem Health and Sustainability*, *5*(1), 36-46.

Vicente, D., Rodríguez-Sinobas, L., Garrote, L., and Sánchez, R. (2016). Application of the system of environmental economic accounting for water SEEAW to the Spanish part of the Duero basin: Lessons learned. *Science of the Total Environment*, *563*, 611-622.

Vilar, L., Garrido, J., Echavarría, P., Martínez-Vega, J., and Martín, M. P. (2019). Comparative analysis of CORINE and climate change initiative land cover maps in Europe: Implications for wildfire occurrence estimation at regional and local scales. *International Journal of Applied Earth Observation and Geoinformation*, *78*, 102-117.

Wan, Z., Hook, S., and Hulley, G. (2015). MOD11A1 MODIS/Terra Land Surface Temperature/Emissivity Daily L3 Global 1km SIN Grid V006 [Data set]. NASA EOSDIS Land Processes DAAC.

Warnell, K. J. D., Russell, M., Rhodes, C., Bagstad, K. J., Olander, L. P., Nowak, D. J., . . . Hirabayashi, S. (2020). Testing ecosystem accounting in the United States: A case study for the Southeast. *Ecosystem Services*, *43*, 101099.

Weber, J.-L. (2007). Implementation of land and ecosystem accounts at the European Environment Agency. *61*(Generic), 695-707. https://doi.org/https://doi.org/10.1016/j.ecolecon.2006.05.023

Weber, J. L. (2014). *Experimental Ecosystems Natural Capital Accounts: Mauritius Case Study: Methodology and Preliminary Results, 2000-2010*. Indian Ocean Commission.

Xu, Y., Ma, F., Meng, D., Ren, C., and Leung, Y. (2017). A co-training approach to the classification of local climate zones with multi-source data. 2017 IEEE International Geoscience and Remote Sensing Symposium (IGARSS),

Yuan, Y., Li, B., Gao, X., Liu, H., Xu, L., and Zhou, C. (2016). A method of characterizing land-cover swap changes in the arid zone of China. *Frontiers of Earth Science*, *10*(1), 74-86.

Zagonari, F. (2016). Using ecosystem services in decision-making to support sustainable development: Critiques, model development, a case study, and perspectives. *Science of the Total Environment*, *548*, 25-32.

Appendices

Appendix 1: Spanish ecosystem extent account table between 1970 and 2015. In square kilometres and percentage of total extent

Ecosystems	Sclerophyllous med. forest	Continental med. forest	Mountain med. forest	Atlantic forest	Alpine forest	Insular forest	Sclerophyllous med. grassland	Continental med. grassland	Mountain med. grassland	Atlantic grassland	Alpine grassland	Insular grassland	Sclerophyllous med. shrubland
Initial Extent	**73324.9**	**41020.5**	**4209.0**	**23493.3**	**1484.7**	**1228.5**	**13839.4**	**9251.3**	**867.0**	**2348.9**	**1156.0**	**274.9**	**38005.6**
Reductions	23157.1	12124.8	936.3	5417.9	432.0	247.9	10030.3	6790.9	639.2	1785.7	423.8	189.0	23985.3
Additions	34231.6	18155.3	1044.1	11416.0	586.8	375.0	24694.6	12130.8	1052.4	12042.3	1007.2	360.7	21941.1
Net changes	11074.5	6030.6	107.7	5998.1	154.8	127.1	14664.3	5339.9	413.1	10256.6	583.4	171.8	-2044.2
Net_%	2.2	1.2	0.0	1.2	0.0	0.0	2.9	1.1	0.1	2.0	0.1	0.0	-0.4
Total turnover	57388.8	30280.1	1980.4	16833.8	1018.9	623.0	34725.0	18921.7	1691.6	13828.0	1431.1	549.7	45926.3
Turnover_%	11.3	6.0	0.4	3.3	0.2	0.1	6.9	3.7	0.3	2.7	0.3	0.1	9.1
Stable Stock	50167.7	28895.8	3272.7	18075.4	1052.6	980.6	3809.1	2460.4	227.7	563.2	732.2	86.0	14020.3
Stable_%	9.9	5.7	0.6	3.6	0.2	0.2	0.8	0.5	0.0	0.1	0.1	0.0	2.8
Final Extent	**84399.6**	**47051.1**	**4316.7**	**29491.5**	**1639.5**	**1355.6**	**28503.8**	**14591.2**	**1280.1**	**12605.5**	**1739.6**	**446.7**	**35961.4**

Ecosystems	Continental med. shrubland	Mountain med. shrubland	Atlantic shrubland	Alpine shrubland	Insular shrubland	Arid zones	Coastal areas	Other land	Wetlands	Rivers and lakes	Perennial woody crops	Annual crops	Urban
Initial Extent	**26338.3**	**3447.1**	**16145.5**	**1012.9**	**2290.7**	**4650.7**	**1626.6**	**7144.3**	**1246.5**	**2264.6**	**55481.5**	**170070.1**	**4232.5**
Reductions	16503.1	1275.5	8722.7	662.4	765.8	1389.9	698.0	3851.7	615.6	277.2	24711.0	72194.7	709.8
Additions	9544.7	1114.2	4203.7	404.8	385.9	912.8	1081.5	3392.0	1435.2	965.4	18871.6	26555.7	10632.3
Net Additions	-6958.4	-161.3	-4519.0	-257.7	-379.9	-477.1	383.5	-459.7	819.6	688.2	-5839.4	-45639.0	9922.5
Net_%	-1.4	0.0	-0.9	-0.1	-0.1	-0.1	0.1	-0.1	0.2	0.1	-1.2	-9.0	2.0
Total turnover	26047.8	2389.7	12926.3	1067.2	1151.7	2302.7	1779.5	7243.7	2050.8	1242.6	43582.7	98750.4	11342.0
Turnover_%	5.1	0.5	2.6	0.2	0.2	0.5	0.4	1.4	0.4	0.2	8.6	19.5	2.2
Stable Stock	9835.2	2171.6	7422.8	350.5	1524.9	3260.8	928.5	3292.6	630.9	1987.4	30770.4	97875.4	3522.7
Stable_%	1.9	0.4	1.5	0.1	0.3	0.6	0.2	0.7	0.1	0.4	6.1	19.3	0.7
Final Extent	**19379.9**	**3285.8**	**11626.5**	**755.3**	**1910.8**	**4173.9**	**2010.8**	**6684.6**	**2066.2**	**2954.3**	**49642.1**	**124431.2**	**14155.6**

Appendix 2: F1-score results of proposed ecosystem types and LUCAS classes (as ground observational data) by ecosystem and periods of available.