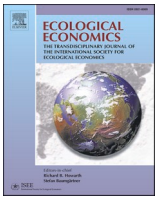
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Methodological and Ideological Options



A framework to allocate responsibilities of the global environmental concerns: A case study in Spain involving regions, municipalities, productive sectors, industrial parks, and companies

Brais Suárez-Eiroa [a,b,*], Emilio Fernández [c], David Soto-Oñate [d,e], Aida Ovejero-Campos [f], Pablo Urbieta [f], Gonzalo Méndez [f]

[a] ECOBAS, Universidade de Vigo, Post-Growth InnovationLab, Campus A Xunqueira, 36005, Pontevedra, Spain
[b] Centro de Investigación Mariña, Universidade de Vigo, Campus Lagoas-Marcosende, 36310 Vigo, Spain
[c] Centro de Investigación Mariña, Universidade de Vigo, Department of Ecology and Animal Biology, Campus Lagoas-Marcosende, 36310 Vigo, Spain
[d] ECOBAS, Universidade de Vigo, ERENEA, Campus Lagoas-Marcosende, 36310 Vigo, Spain
[e] Ostrom Workshop, Indiana University, Bloomington, IN 47408, United States
[f] Centro de Investigación Mariña, Universidade de Vigo, Department of Marine Geosciences and Territorial Planning, Campus Lagoas-Marcosende, 36310 Vigo, Spain

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ABSTRACT

This paper presents the responsible operating space (ROS) framework, a framework that includes both the common governance of the global environmental concerns and the comprehensive management of environmental sustainability. The ROS framework determines environmental objectives regarding the global concerns using a footprint perspective, and defines coherent integrated environmental objectives for the production and consumption processes at different territorial scales. In this case study, we firstly estimated the global ROS, as well as both the consumption-side ROS and the production-side ROS of Spain. Then, the national ROS was transferred to the 17 Spanish regions and was further extended to the 313 municipalities of one of the regions (Galicia). Finally, we transferred the ROS to 163 sectors, as well as to one industrial park and several companies. Our results demonstrate that, even assuming the methodological limitations and scarcity of data, it is already possible to allocate objectives and decentralize the management of global environmental concerns even at municipal and company levels. The role of circular economy under the ROS framework is also discussed on the basis of the potential coexistence of circularity with an unsustainable production-consumption system.

1. Introduction

Circular economy (CE) has gained momentum among scholars, policymakers and businesses (Prieto-Sandoval et al., 2018). Although CE policy agendas and business strategies commonly address the adoption of closing-the-loop production patterns with a strong focus on improving efficiency (Ghisellini et al., 2016; Kovacic et al., 2020), such a perspective has been shown to be counterproductive in the long term (Korhonen et al., 2018a; Millar et al., 2019). From an ecological economics perspective, CE should care about the fluxes of energy and matter between the socio-economic system and the natural system in absolute terms (Daly, 2007), thereby incorporating the notion of limits resulting from a finite natural system (Suárez-Eiroa et al., 2019).

The management of the natural limits covers territorial environmental concerns (TECs) and global environmental concerns (GECs). The GECs include the processes related to the Earth system resilience (Rocha et al., 2018) framed under the Planetary Boundaries (PBs) framework (Rockström et al., 2009; Steffen et al., 2015), as well as other GECs derived from the Sustainable Development Goals (SDGs) (Dong and Hauschild, 2017), such as microplastics concentration (Wright et al., 2013), land degradation and desertification (Briassoulis, 2019), air pollution (UN, 2015), or waste generation (Kaza et al., 2018).

TECs can be managed within territories (McLaughlin, 2018). However, management of GECs requires allocation of objectives to the different social actors from a top-down process. Since several global environmental concerns need to be addressed expeditiously, active

* Corresponding author at: Centro de Investigación Mariña, Universidade de Vigo, Office 56, Block C, Experimental Sciences building, Campus Universitario Lagoas-Marcosende, 36310 Vigo, Spain.
E-mail addresses: brsuarez@uvigo.es (B. Suárez-Eiroa), esuarez@uvigo.es (E. Fernández), davidsoto@uvigo.es (D. Soto-Oñate), aovejero@uvigo.es (A. Ovejero-Campos), pablo.fernandez.urbieta@uvigo.es (P. Urbieta), mendez@uvigo.es (G. Méndez).

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research is being conducted to address this operational need (e.g. Dao et al., 2018; Ryberg et al., 2018a). However, the responsibility of companies (e.g. Sandin et al., 2015; Ryberg et al., 2018b) and territories (e.g. Nykvist et al., 2013; Dao et al., 2018) on the GECs and the environmental performance of CE action plans of the different social actors have not been coherently integrated (Corona et al., 2019). Likewise, the actions among the different levels of implementation of CE remain barely connected (McDowall et al., 2017). Thus, there is a need to coherently integrate the CE strategies of the different social actors under the same framework.

This paper presents and analyzes the foundations and the practical capacity of the Responsible Operating Space (ROS) framework. The ROS framework allows to coherently stablish environmental objectives for the different social actors, integrating their economic plans within a territory to operationalize environmental sustainability. The paper is organized as follows. Section 2 reviews different methods to downscale the GECs existing in the scientific literature. Section 3 presents the ROS framework and the methodology to integrate the environmental objectives of the different social actors. Section 4 presents the results of applying the ROS framework to the case of Spain. Firstly, we estimated the global ROS, as well as both the consumption-side ROS and the production-side ROS of Spain. Secondly, the national ROS was transferred to the 17 Spanish regions and then to the 313 municipalities of one of the regions (Galicia). Thirdly, we transferred the ROS to 163 sectors, as well as to one industrial park and several companies. Section 5 discusses the main contributions of this paper, the limitations of the proposed methodology and the implications of the results for both CE implementation and policymakers.

2. Theoretical background

This section briefly outlines the theoretical foundations to allocate objectives to the different social actors regarding GECs. To start with, the Driver-Pressure-State-Impact-Response (DPSIR) framework (Smeets and Weterings, 1999) fostered a better understanding of the relationship between human activities and environmental impacts (Meyer and Newman, 2018), leading decision-makers to more appropriate responses to address environmental concerns (Balzan et al., 2019). Different indicators could be used to monitor impacts related to the same environmental category. For instance, we can use either temperature increase or greenhouse gases emissions to measure climate change (IPCC, 2018). Moreover, the analysis of environmental impacts from a footprint perspective results in a more comprehensive view of the responsibility of the different social actors on the GECs than the analysis of direct impacts (Fang et al., 2015; Häyhä et al., 2016).

Environmental impacts can be analyzed from the consumption or the production perspective. The consumption perspective involves territories and specific actors (governments, families, organizations, etc.). The production perspective refers to sectors, companies, and other strategic alliances such as industrial parks.

From the consumption perspective, the most common methodology to transfer GECs to a territory is the world per capita distribution method (e,g, Nykvist et al., 2013; Fanning and O'Neill, 2016; Dao et al., 2018). Nykvist et al. (2013) calculated four national boundaries for Sweden based on the planetary boundaries established initially by Rockström et al. (2009). In a similar effort, Cole et al. (2014) calculated eight limits for South Africa and analyzed the current situation of the country regarding these limits. Later, Fang et al. (2015) delved into the relationship between the planetary boundaries initially proposed by Rockström et al. (2009) and several consumption-based indicators. Fanning and O'Neill (2016) suggested four indicators to calculate national boundaries for Spain and Canada, assessing the environmental performance of these countries based on the computed limits, and Hoff et al. (2017) downscaled five planetary boundaries to Europe and analyzed its environmental performance based on the calculated limits. Häyhä et al. (2016) developed a framework to translate the planetary boundaries to the national scale based on biophysical, socio-economic, and ethical issues. Recently, Dao et al. (2018) proposed a methodology for downscaling the PBs to the national level and calculated the national boundaries and environmental footprints for Switzerland. This approach integrates the temporal responsibility of the countries, what is considered in our study as the starting point of the methodology proposed in Section 3.2.1.

From the production perspective, Sandin et al. (2015) proposed a methodology to estimate the targets of each sector for every environmental category. According to this approach, a reduction target is initially defined, and then modified by using two correction factors: the relative contribution of the sector in the future with respect to the relative contribution of the sector in the present, and a readjustment factor based on different ethical principles. In the methodology proposed by Wolff et al. (2017), the ecological budget of each company is allocated based on the relative contribution of the sector, what is known as the grandfathering principle (Kara et al., 2018), the consumer base of the company in person-year equivalent, the carrying capacity, and the population size. Ryberg et al. (2018a) show four sharing principles to establish sectorial and companies' boundaries within the European Union (EU). Firstly, they calculated the expenditure of a given product concerning the total spending in the world. This method allocates higher limits to products in which an average citizen spends more money, a variable that is not necessarily related to its environmental impact (Huppes, 2006). Secondly, a per capita distribution was used to transfer PBs to the EU, and then the expenditure of a product for the total spending in the EU was calculated. Thirdly, a per capita distribution is initially used to transfer PBs to the EU. To this aim, the product added gross value with respect to the added gross value of the EU was calculated, following a methodology previously applied in the USA energy sector (Algunaibet et al., 2019). Fourthly, they used the grandfathering principle to allocate PBs among the different sectors within the EU. The main limitations of the grandfathering principle lie on the fact that the initial allocation is based on the current situation, and therefore any new future situation would not be taken into consideration (Ryberg et al., 2018a). The introduction of a correction factor that allows considering the future situation, such as the one proposed in Sandin et al. (2015) could help to minimize this limitation.

All the previous scientific works shape a toolbox to transfer the GECs to territories and companies. Nevertheless, the methodologies related to the production and consumption perspectives remain unconnected, and hence it is necessary to explore new paths to coherently integrate the environmental objectives of the different social actors.

3. Materials and methods

3.1. The Responsible Operating Space framework

The Responsible Operating Space (ROS) framework has been developed under the environmental management perspective, i.e. aiming at adjusting human activities to preserve nature and ecosystem services that support the socioeconomic system. Five main features characterize the ROS framework:

- It covers both the territorial (TECs) and global (GECs) environmental concerns.
- It may address the same environmental category both as a TEC and GEC, e.g. water consumption or eutrophication.
- It determines environmental objectives regarding GECs using a footprint perspective, i.e. including direct and indirect impacts.
- It may need several indicators to monitor the same environmental category.
- It necessarily defines coherent and integrated environmental objectives for the production and consumption processes.

Territorial and global perspectives are essential to manage human

activities within the environmental ceiling. For instance, Randers et al. (2019) established a global limit for water consumption of 3000 km^3/y, and, at the same time, each territory, and particularly each riven basin, present limits for direct consumption both over temporal and spatial scales (Cole et al., 2014; Fanning and O'Neill, 2016). While the territorial perspective related to water consumption takes already part in the political agenda, the global perspective is not commonly addressed, forgetting that global interconnections (Liu et al., 2007) and cross-scale feedbacks (Rocha et al., 2018) within the socio-ecological system make it difficult to separate environmental impacts in any given region from pressures related to human activities occurring in other regions of the world. Thus, a footprint perspective provides a more realistic idea about the environmental impacts of the production-consumption process than the simple analysis of direct impacts (Fang et al., 2015; Häyhä et al., 2016), and is herewith considered the fundamental tool to address GECs. Moreover, there is currently a lack of consensus about the use of a single indicator to assess the performance of some environmental categories. For instance, Meyer and Newman (2018) used the nitrogen footprint to monitor the nitrogen cycle, while Dao et al. (2018) used nitrogen losses from agriculture for the same objective. Therefore, it can be necessary to use several indicators to monitor the same environmental impact category.

Important differences emerge when environmental objectives for the different social actors are considered for TECs and GECs. TECs are context-dependent, and hence they are directly transferred to the various social actors according to their location. GECs are transferred to the diverse social actors through a top-down process. The consumption-side ROS allocates GECs among different Territorial Units (TU), i.e. a territory that is under the jurisdiction of a government with enough granted competencies to perform this allocation. TUs can be nations, sub-national regions, municipalities, or supra-national entities. TUs are divided into three levels: supra-Key Territorial Units (supra-KTU), Key Territorial Units (KTU), and sub-Key Territorial Units (sub-KTU). The KTU is the main reference for this allocation procedure since its government is the only one that allocates the ROS to the productive sectors. Generally, this category is currently associated with nation-states.

The supra-KTUs refer to all the steps in the allocation procedure that are between the global and the KTU. The European Union could be an example of supra-KTU. Supra-KTU would therefore allocate the consumption-side ROS to lower-scale territories, which can be other supra-KTUs or the KTUs. Sub-KTUs are all subsequent steps in the allocation procedure after the KTU intervention. As in the case of supra-KTUs, they are only involved in the consumption side of the process.

So, the allocation procedure performs as follows. Firstly, the GECs can be transferred either directly to KTUs or supra-KTUs. Once the consumption-side ROS of the KTU has been calculated, it is transferred to the sub-KTUs and used to compute the production-side ROS of the KTU. Fig. 1 presents the relationship between the consumption and production processes, which is the foundation to calculate the production-side ROS from the consumption-side ROS.

The environmental impacts related to the consumed products can be generated in the territory or any other territory. Thus, indirect and direct environmental impacts are considered and included within the impacts of products produced in a territory for domestic consumption (the violet rectangle in Fig. 1). The rectangles in Fig. 1 are built from the footprint perspective. Particularly, this analysis can be implemented by applying the Environmental Extended Input-Output Analysis (EEIOA), i.e. the most extended methodology to calculate environmental impacts at the macro level (e.g. Kerkhof et al., 2009; Di Donato et al., 2015). Through this method, the production-side ROS of the KTU can be later transferred to sectors and companies.

From an ecological economics perspective, the definition of environmental objectives can be considered the starting point for the elaboration of economic plans, i.e. the CE strategies of the different social actors. Fig. 2 presents the way CE strategies are integrated under the ROS framework. Governments of the KTU and the sub-KTUs can design

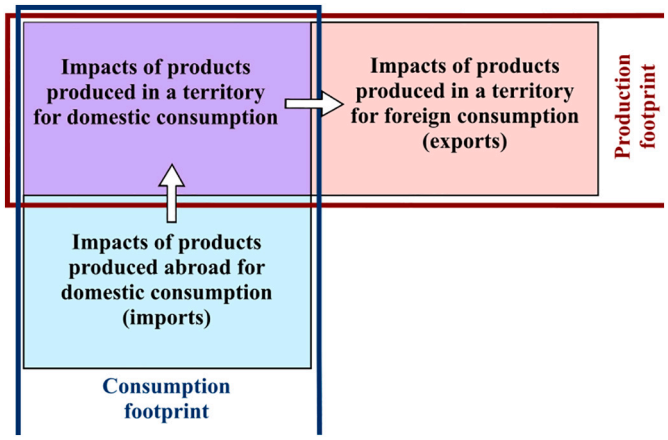


Fig. 1. Connection between the consumption-side and the production-side footprint. The arrows represent the way of calculating the production-side ROS of a territory departing from the consumption-side ROS of that territory. The vertical rectangle corresponds to the consumption-side footprint whereas the horizontal one corresponds to the production-side footprint. The overlap of both components, the violet rectangle, represents the common space between both economic sides, i.e., the environmental impacts associated with the products produced in a territory for domestic consumption. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

CE strategies to promote actions among the consumption actors: public administrations, social civic organizations, and households. Governments of the supra-KTUs could also make recommendations and define action guidelines, like the recent CE package of the European Union. Besides, sectors and companies can also design their own CE strategies based on their environmental objectives.

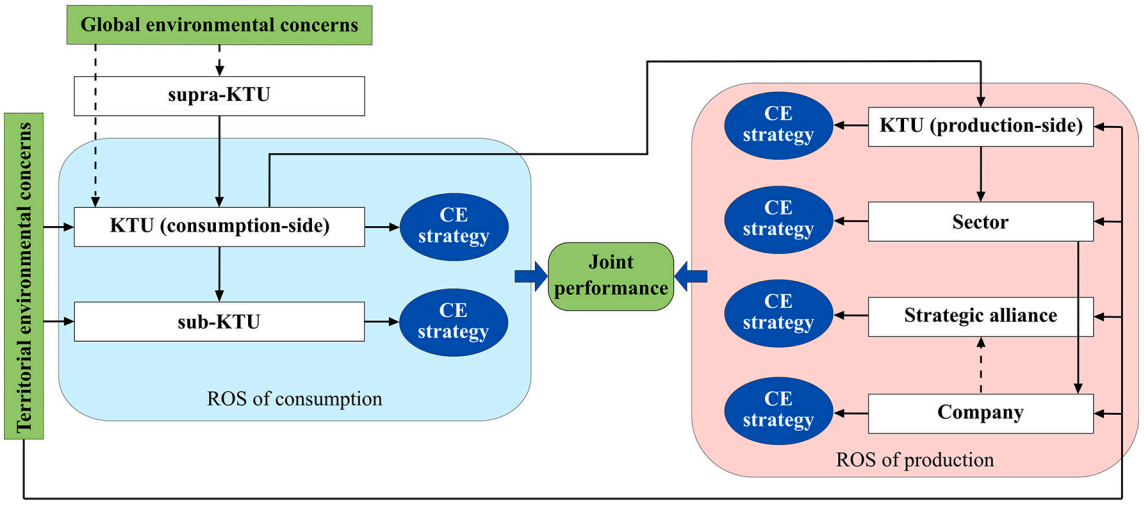
Under this framework, all CE strategies should be designed in cooperation with social actors because only the joint performance of all of them will determine the success of environmental sustainability within the KTU. Different CE strategies could be applied according to each territory and the current situation of the different social actors. Whenever the implemented strategies match the goals for the different social actors, the CE system would be successful.

3.2. Case study

The case study uses Spain as the KTU. We avoided the inclusion of a supra-KTU (such as the European Union) since it would not contribute much to the analysis. Spanish competencies are decentralized in three main governance levels: the national, the regional (17 Autonomous Communities), and the local (8131 municipalities). Other territorial levels (e.g. *provinces*) hold some administrative attributions, but the bulk of Spanish competencies are distributed among the national, regional, and local governments. Therefore, we used Autonomous Communities and the municipalities as sub-KTUs. Since the inclusion of territorial environmental impacts is not the focus of this investigation, the ROS computed here is exclusively focused on the transference of GECs to the different social actors within Spain.

3.2.1. ROS of Spain

The first step was the calculation of the consumption-side ROS of Spain. We defined the ROS of both the world and Spain by using 14 indicators according to the available data. A review of indicators and a proposal to quantify the global ROS is presented in the supplementary material file 1. In the literature review stressed the convenience for using different indicators for the same environmental category when distributing responsibilities among the different social actors. These indicators were used in this investigation as a starting point to test the proposed methodology, but neither the list of indicators nor the



b) Entities involved in the ROS framework by level in the operational structure

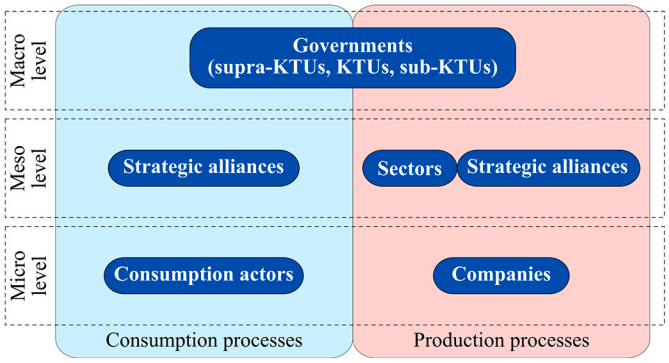


Fig. 2. Integration of the circular economy strategies of the different social actors under the ROS framework. The blue boxes contain governments, strategic alliances such as consumer cooperatives, and consumption actors. The red boxes contain productive sectors, strategic alliances such as industrial parks, and companies. Arrows with dotted lines show optional ways to transfer the ROS. KTU: Key Territorial Unit. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

estimated values are intended to be definitive.

Eqs. (1) and (2) were used to transfer the global ROS to the consumption-side ROS of Spain for annual and budget limits[1] respectively:

$$National\ ROS_i = \frac{National\ Pop}{World\ Pop} \times World\ ROS_i \tag{1}$$

$$National\,ROS_i = \frac{\sum_{BaseYear}^{Final\,year} National\,Pop_y}{\sum_{BaseYear}^{Final\,year} World\,Pop_y} \times World\,ROS_i - \sum_{BaseYear}^{Current\,year} National\,Foot_{y,i} \tag{2}$$

Where *National ROS* is the consumption-side ROS of the country, *National Pop* is the population of the country, *World Pop* is the population of the world, *World ROS* is the ROS of the world, *National Foot* is the consumption-side footprint of the country, and i and y refer to the environmental impact category and the year respectively. The base and the final years represent the first and the last year to account the accumulation respectively, and the current year makes reference to the year when calculations are being computed.

The per capita distribution is a widely used sharing principle throughout the literature on downscaling GECs (Nykvist et al., 2013; Dao et al., 2018). Unlike other approaches based on restricting direct emissions (e.g. Kyoto Protocol), the per capita distribution is supported by the fact that environmental footprints allow to better measure the responsibilities of each social actor for the GECs (Fang et al., 2015).

The literature on downscaling GECs to lower scales is still growing. For this reason, we performed a comparison between the methodologies to transfer GECs with budget limits to the KTU proposed in Dao et al. (2018) and Eq. (2), whose results are shown in Section 4.1. This exercise allows advancing in this field, which is essential for the functionality of the ROS framework. The initial budget for the climate change category was calculated for four countries (Spain, Germany, South Africa, and Mexico) using 1990 as the base year. These countries were selected as they differ in their demographic trends since 1990. We further explored the effect of using different base years on the estimation of initial budgets. In this case, Spanish and Chinese budgets for climate change were compared with global budget estimates derived from different base years. These countries were selected as they differ in the speed they consume their budget since 1990.

Production-side and consumption-side environmental footprints are connected within the KTU. The production-side footprint can be calcu-

[1] Annual limits could be recalculated each year. For budget limits it is necessary to define the base and final years among which accumulation will be produced.

lated by subtracting environmental imports and adding environmental exports to the consumption-side footprint. Therefore, to calculate the production-side ROS of the KTU, we subtracted the share of environmental imports and added the share of environmental exports to the consumption-side ROS of the KTU. Eq. (3) was applied to calculate the production-side ROS of Spain:

$$National\ ROS\ production_i = National\ ROS_i \times \frac{(1 - Environmental\ Imports_i)}{(1 - Environmental\ Exports_i)} \quad (3)$$

Where *National ROS production* is the production-side ROS of the country, *National ROS* is the consumption-side ROS of the country, and *Environmental Imports* and *Environmental Exports* are the percentages of Spanish imports and exports of environmental impacts respectively. Imports and exports of environmental impacts were calculated by applying the EEIOA to the EXIOBASE 3 database (Stadler et al., 2018) for the year 2011.

Section 4.1 presents the results of the global ROS, both the consumption-side and the production-side ROS of Spain, and the comparison between the methodologies.

3.2.2. ROS of Spanish regions and municipalities

The consumption-side ROS of Spain was distributed among the 17 Spanish Autonomous Communities. Then, the regional ROS of one of the 17 Autonomous Communities, Galicia, was distributed among its 313 municipalities.[2] Galicia was selected as an example to test the methodology because both population data and geographical extension lie close to the average values obtained for Spanish Autonomous Communities, and hence it can be considered as a representative Spanish region. Eqs. (4) and (5) were used to transfer the consumption-side ROS of Spain to the regional ROS and the regional ROS to the municipal ROS respectively:

$$Regional\ ROS_{r,i} = National\ ROS_i \times \frac{Regional\ Foot_{r,i}}{National\ Foot_i} \times k_{r,i} \quad (4)$$

$$\sum_{r=1}^{R} Regional\ ROS_{r,i} = National\ ROS_i$$

$$Municipal\ ROS_{m,r,i} = Regional\ ROS_{r,i} \times \frac{Municipal\ Foot_{m,r,i}}{Regional\ Foot_{r,i}} \times k_{m,r,i} \quad (5)$$

$$\sum_{m=1}^{M} Municipal\ ROS_{m,r,i} = Regional\ ROS_{r,i}$$

Where *National ROS* is the consumption-side ROS of the country, $Regional\ ROS_{r,\ i}$ is the ROS of the region *r*, $Municipal\ ROS_{m,\ r,\ i}$ is the ROS of the municipality *m* in a given region, *National Foot* is the consumption-side footprint of the country, $Regional\ Foot_{r,\ i}$ is the consumption-side footprint of the region, $Municipal\ ROS_{m,\ r,\ i}$ is the consumption-side footprint of the municipality, $k_{r,\ i}$ is the correction factor for each region, $k_{m,\ r,\ i}$ is the correction factor for each municipality, and *i* refers to the environmental impact category.

This sharing principle, known as the grandfathering principle (e.g. Wolff et al., 2017; Ryberg et al., 2018a), was used to allocate the ROS to the different sub-KTUs according to their current contribution. Consequently, the same relative efforts are initially required for all sub-KTUs, which is the main motivation behind the use of this sharing principle at this level. A correction factor *k* is used to readjust the initial distribution of responsibilities for the GECs. The Spanish and the Galician governments are responsible for the final allocation of the ROS to the different Autonomous Communities and municipalities respectively.

Since the grandfathering principle is used to perform the initial distribution of responsibilities among regions and municipalities, knowledge of their current environmental impact is needed. The environmental impacts of Spain were calculated from EEIOA and the EXIOBASE 3 database.[3] EXIOBASE 3 includes environmental and consumption data of 200 products and 163 sectors for 44 countries and 5 rest-of-the-world regions between 1995 and 2011. Supplementary material file 2 updates the characterization of factors to adapt the database to the selected indicators.

Environmental impacts for regions and municipalities were estimated by applying an IO-expenditure EEIOA. This method combines the environmental intensity of the goods and services resulting from the EEIOA and expenditure data to calculate the environmental impacts (Di Donato et al., 2015). The environmental intensity of goods and services was calculated using the EXIOBASE 3 database. The consumer expenditure was extracted from INE (2019). Consumer expenditure data for regions are available for 103 goods and services, and consumer expenditure data for municipalities is only available for the 12 main groups of consumption from the COICOP classification. Therefore, environmental intensities from EXIOBASE 3 were aggregated as shown in Supplementary material file 2. All data used for these calculations corresponded to 2011 as EXIOBASE 3 includes data until that year.

Consumer expenditure data for municipalities is only available in five population size classes (> 100,000, 50,000–100,000, 20,000–50,000, 10,000–20,000, ⟨10,000). Therefore, all municipalities belonging to the same interval were assumed to have the same pattern of consumption. Despite the lack of resolution of the data set, the proposed methodology could be applied.

Using this data set the environmental pressure from households was computed. The environmental pressure related to other consumer actors (i.e. governments and civil society organizations) was calculated for Spain by applying EEIOA to EXIOBASE 3 database, and then it was allocated to regions and municipalities applying a per capita distribution. National population data were extracted from UNEP (2017), whereas regional and municipal population data were obtained from INE (2020), and further readjusted to fit the UNEP (2017) data set.

To test for the effect of the correction factor for each sub-KTU (*k*) on the final results, we defined three scenarios. This factor refers to the times the contribution of a sub-KTU can be increased with respect to its current contribution. For instance, if a sub-KTU is currently generating 10% of the impacts of the KTU, and policymakers decide that it should contribute by 8%, *k* would be 0.8. When the correction factor is applied in this way, a fraction of the ROS of the KTU is removed from the initial distribution, and should be reallocated. Thus, if the sub-KTU of the previous example had an initial ROS of ten units of impact, it will have just eight units after the readjustment. Since the condition is that the sum of the ROS of all the sub-KTU equals the ROS of the KTU, the remaining two units of impacts should be reallocated to other sub-KTUs. For this reason, we use *k-rest* to explain how the remaining ROS is reallocated among the other sub-KTUs. *k-rest* represents the share of the remaining ROS that was reallocated to a particular group of sub-KTUs. Therefore, the sum of the values of *k-rest* is necessarily 1.

The three defined scenarios allocate responsibilities using different criteria. Scenario 1 uses the estimated population increase for the year 2030 as the base for the correction factor. It assumes that reduction policies may need to take into account population projections. Scenario 2 uses population density, and it is based on the idea that a high population density is usually related to more compact cities, which are likely

[2] A map of Spain with the distribution of the Autonomous Communities is presented in Appendix A, as well as a map of Galicia with the distribution of the municipalities. The autonomous cities Ceuta and Melilla were excluded.

[3] Nitrogen and phosphorus fertilizers have been excluded from the calculus due to the lack of data for the consumption of fertilizers from a footprint perspective. Besides, those environmental categories in which the national ROS is zero has also been excluded from calculations since the necessary reduction of all social and productive sectors is 100%.

in a better position to manage the ecological transition. The correction factor in Scenario 3 is based on the per capita income. The rationale behind the use of this variable is that regions and municipalities with a higher gross rent per capita are likely to have more possibilities to implement the necessary ecological transition policies. The resulting values from each scenario were divided into ten deciles. As population densities of the studied municipalities are in the range of 14 to 6465 inhabitants per km^2, municipalities with a population density in the first decile will be assigned a given value of *k*. Table 1 shows the different values for *k* and *k-rest* for each decile according to each scenario.

For instance, in scenario 1, *k* in decile 10 is 0.6, which means that regions and municipalities with a higher expected population decrease would be reallocated with a lower ROS than the initial one. The remaining ROS is reallocated among deciles 1 to 4: 40% to regions or municipalities within decile 1, 30% to decile 2, 20% to decile 3, and 10% to decile 4. The distribution within each decile was performed using a per capita criterion; the population of each region and municipality included in decile 1 was divided by the total population of the region and the municipality included in decile 1.

The main aim of the selected scenarios is to allow discussing the reliability of the proposed methodology when different variables to readjust the initial allocated ROS of regions and municipalities are selected. The discussion about the optimal way to distribute responsibilities among regions and municipalities is out of the scope of this paper. A more detailed explanation about the scenarios and data sources used in the calculations are included in the supplementary material file 1. A summary of regional and municipal results is presented in Section 3.2, and the detailed results are shown in the supplementary material file 3.

3.2.3. ROS of Spanish sectors, industrial parks, and companies

The production-side ROS of Spain was distributed among Spanish sectors, as well as among six companies used as examples. Then, we simulated an industrial park composed of four companies. The economic data corresponding to these sectors were extracted from the Eurostat (2019) database, and the data for the companies is based on real economic data extracted from Sabi (2020).[4] All economic data refer to the year 2011. Economic data for sectors and companies are presented in Table 2.

The environmental data for the different productive sectors have been calculated by applying the EEIOA to the EXIOBASE 3 database for the year 2011. Both the Eurostat and EXIOBASE databases provide information for each sector within each region, thereby allowing performing the corresponding assignments. Therefore, the results showed in Section 4.3 correspond to real scenarios that would derive from the application of the proposed methodology by decision makers.

A summary of the results for three sectors, as well as the results for companies and the industrial park, are presented in Section 4.3, and the detailed results are shown in the suplementary material file 3.

It would be of interest to apply the grandfathering principle to calculate the ROS of sectors since the same relative effort is required for all of them. However, no future situations are taken into consideration by using this approach. Nevertheless, it should be taken into account that transitions are usually slow; hence there is scope for adaptation in face of the emergence of new sectors. Additionally, during an ecological transition, the emergence of new sectors is expected to be more environmentally friendly, thus the necessary ROS is likely to be smaller. Moreover, the introduction of a correction factor can help to overcome this limitation. Therefore, Eq. (6) was applied to transfer the national ROS of production to Spanish sectors:

$$Sectorial\,ROS_{s,i} = National\,ROS\,production_i \times \frac{Sectoral\,Foot_{s,i}}{National\,Foot\,production_i} \times k_{s,i} \quad (6)$$

$$\sum_{s=1}^{S} Sectoral\ ROS_{s,i} = National\ ROS\ production_i$$

Where $Sectoral\ ROS_{s,\,i}$ is the ROS of the sector s, $Sectoral\ Foot_{s,\,i}$ is the footprint of the sector *s*, $National\ Foot\ production_i$ is the production-side footprint of the country, $k_{s,\,i}$ is the correction factor for each sector, and *i* refers to the environmental category.

Since the effect of the correction factor was already discussed in the calculation of the ROS for sub-KTUs, we did not perform any additional tests. After the calculation of the sectorial ROS, the ROS of industrial parks and companies can be assessed. Although industrial parks would be the next logical implementation level, we propose first to calculate the ROS of the companies from a sector. The ROS of the three companies from the automotive sector was calculated by applying Eq. (7):

$$Company\ ROS_{c,s,i} = Sectoral\ ROS_{s,i} \times \frac{Company\ GVA_{c,s}}{Sector\ GVA_s} \quad (7)$$

$$\sum_{c=1}^{C} Company\ ROS_{c,s,i} = Sectoral\ ROS_{s,i}$$

Where $Company\ ROS_{c,\,s,\,i}$ is the ROS of the company c, $Company\ GVA_{c,\,s}$ is the GVA of the company *c*, $Sector\ GVA_s$ is the GVA of the sector *s*, and *i* refers to the environmental category. The results express the ROS of the companies in terms of *added environmental impact* to the total environmental impact of production. We transformed the results into the ROS of the companies in terms of life cycle assessment (LCA) by applying Eq. (8):

$$Company\,ROS_LCA_{c,s,i} = Company\,ROS_{c,s,i} \times \frac{Company\ Operating\ Revenues_{c,s}}{Company\,GVA_{c,s}} \quad (8)$$

Where $Company\ ROS_LCA_{c,\,s,\,i}$ is the ROS of the company *c* of sector *s* in terms of LCA, $Company\ ROS_{c,\,s,\,i}$ is the ROS of the company calculated previously, $Company\ GVA_{c,\ s}$ is the GVA of the company, $Company\ Operating\ Revenues_{c,\ s}$ is the operating revenue of the company, and *i* refers to the environmental category.

Finally, we simulated an industrial park composed of four companies from different sectors: the fishing sector, the waste sector, the energy sector, and the automotive sector. We assumed that all the production of the companies takes place within the industrial park. We also assumed that the waste company takes part in the supply chain of the remaining companies: 20% of its operating revenues come from the energy company, 50% from the fishing company, and 30% from the automotive company. This will result in duplicities, which were addressed using Eq. (9) to calculate the ROS of the industrial park:

$$Ind\ Park\ ROS_LCA_{p,i} = \sum Company\ ROS_LCA_{c,s,i} - \sum Company\ ROS_{c,s,i} \times \frac{Econ_Duplicities_{c,s}}{Company\ GVA_{c,s}} \quad (9)$$

Where $Ind\ Park\ ROS_LCA_{p,\ i}$ is the ROS of the industrial park *p*, $Company\ ROS_{c,\ s,\ i}$ is the ROS of the company *c* of sector *s*, *Company*

[4] Sabi database contains real economic information on companies operating in Spain and Portugal. For confidentiality reasons, data were rounded. These data are used later for calculating the environmental budget of the company, and hence the results are representative of real companies of each sector.

Table 1
k and *k-rest* values used in each scenario.

		Decile									
		1	2	3	4	5	6	7	8	9	10
Scenario 1	*k*					1	1	0.9	0.8	0.7	0.6
	k-rest	0.4	0.3	0.2	0.1						
Scenarios 2 and 3	*k*	0.8	0.85	0.9	0.95	1	1				
	k-rest							0.1	0.2	0.3	0.4

Scenario 1 uses the population increase estimated for the year 2030. Scenario 2 uses population density. Scenario 3 uses gross rent per capita.
Note: The correction factor *k* refers to the times the contribution of the sub-KTU can be increased related to their current contribution. k-rest values represent the fraction of the ROS subtracted to the first group of municipalities which is assumed by the sub-KTUs located in the remaining deciles.

Table 2
Production data corresponding to five Spanish economic sectors, six companies, and one industrial park.

	Code	GVA (thous €)	Operating revenues (thous €)	% over the PIB of the sector
Fishing sector	S1	1,230,000	–	–
Automotive sector	S2	8,498,000	–	–
Construction sector	S3	73,557,000	–	–
Energy sector	S4	25,819,000	–	–
Waste sector	S5	6,222,000	–	–
Company 1 from S2	C1	1,000,000	10,000,000	11.77%
Company 2 from S2	C2 and C1_IP	100,000	500,000	1.18%
Company 3 from S2	C3	10,000	150,000	0.12%
Company from the energy sector	C2_IP	150,000	800,000	0.58%
Company from the waste sector	C3_IP	800	5000	0.16%
Company from the fishing sector	C4_IP	1000	7000	0.08%

Table 3
Results of the global and the Spanish ROS.

	Global ROS	Spanish ROS of consumption	Spanish ROS of production
Climate change (kg CO_2eq) [base year 2019]	5.03E+14	2.12E+12	1.89E+12
Ocean acidification (kg CO_2) [base year 2019]	3.05E+14	1.29E+12	1.21E+12
Stratospheric Ozone Depletion (kg CFC-11 eq)	0.00E+00	0.00E+00	0.00E+00
N total losses (kg N)	6.20E+10	3.81E+08	3.07E+08
N agricultural losses (kg N)	5.07E+10	3.12E+08	2.47E+08
N fertilizers application (kg N)	8.30E+10	5.11E+08	–
P total losses (kg P)	1.10E+10	6.77E+07	4.91E+07
P fertilizers application (kg P)	3.85E+10	2.37E+08	–
Freshwater use (Mm^3)	2.80E+06	1.72E+04	1.57E+04
Anthropized land (km^2)	4.75E+07	2.92E+05	1.79E+05
Deforested land (km^2) [base year 1990]	0.00E+00	0.00E+00	0.00E+00
Biodiversity loss (PDF)	7.11E-02	4.37E-04	2.78E-04
Air pollution (PM2.5 eq)	4.77E+10	2.93E+08	2.18E+08
Technical waste generation (kg)	0.00E+00	0.00E+00	0.00E+00

$ROS_LCA_{c,\ s,\ i}$ is the ROS of the company from the LCA perspective, $Econ_Duplicities_{c,\ s}$ are the economic duplicities, $Company\ GVA_{c,\ s}$ is the gross added value of the company, and *i* refers to the environmental impact category. In this case, we needed to correct three duplicities through the second part of Eq. (9). The economic duplicities are computed for those companies whose income is received in part by other companies of the parks.

4. Results

4.1. The Spanish ROS

The consumption-side and the production-side ROS of Spain are presented in Table 3. We defined the ROS for climate change and ocean acidification as a budget limit with 2019 as the base year. Therefore, the values represent the budget of emissions from 2019 to 2100. These data will be inputs in the reduction strategy. From the consumption perspective, and considering a linear reduction strategy (the die-off rate of emissions is constant), the estimated year to become carbon neutral is 2029, with an annual reduction of 10% from the initial emission value of 373 $MtCO_2$ eq. The concept of carbon-neutral includes both the reduction of emissions and the increase of carbon sinks. We also present the deforested land as a budget limit with 1990 as the base year. This means that the sum of the deforested land since 1990 (a pressure indicator) should be zero at most. The rest of the environmental impact categories are expressed through annual limits. Thus, the goals presented in Table 3 are to be reached each year.

As explained in Section 3.2, we performed a comparison between different methodological approaches for the calculation of budget limits for the KTU (Fig. 3). The budget estimated for Spain, Germany, South Africa, and Mexico is shown in Fig. 3a. In the case of Germany and Spain, the outcome of the Dao et al. (2018) method yields higher values than when the method proposed in this study is used. However, the budget corresponding to South Africa and Mexico is higher when the methodology proposed in our research is applied. These contrasting results derive from the different approaches adopted by the two tested methods. The Dao et al. (2018) method used a per capita distribution in the base year, whereas the methodology proposed in Eq. (2) considers the population projections for the region. This is a significant difference since developed countries usually present a lower rate of population increase than less developed countries. Thus, if a budget is calculated for a region based on the population at a given time, future needs could be either underestimated or overestimated.

We then compared the results derived from the use of different base years (Fig. 3b). As expected, on a global scale, the consumed budget is independent of the selected base year since this budget is based on a physical limit, and the consumed budget represents the distance from the current situation to that limit. By contrast, the budget consumed by countries differs depending on the base year since, in this case, this value not only refers to physical limits but also addresses the concept of temporal responsibility. The overall Spanish tendency shows that the more recent the base year, the lower the responsibility. This is the general pattern followed by countries with high levels of development in the nineties, which is usually associated with larger environmental impacts. Assuming the current trend, the selection of 1990 as the base year would result in Spain reaching 100% of its budget in 2008. If 2018 was selected, Spain would reach 100% of its budget in 2022. The case of

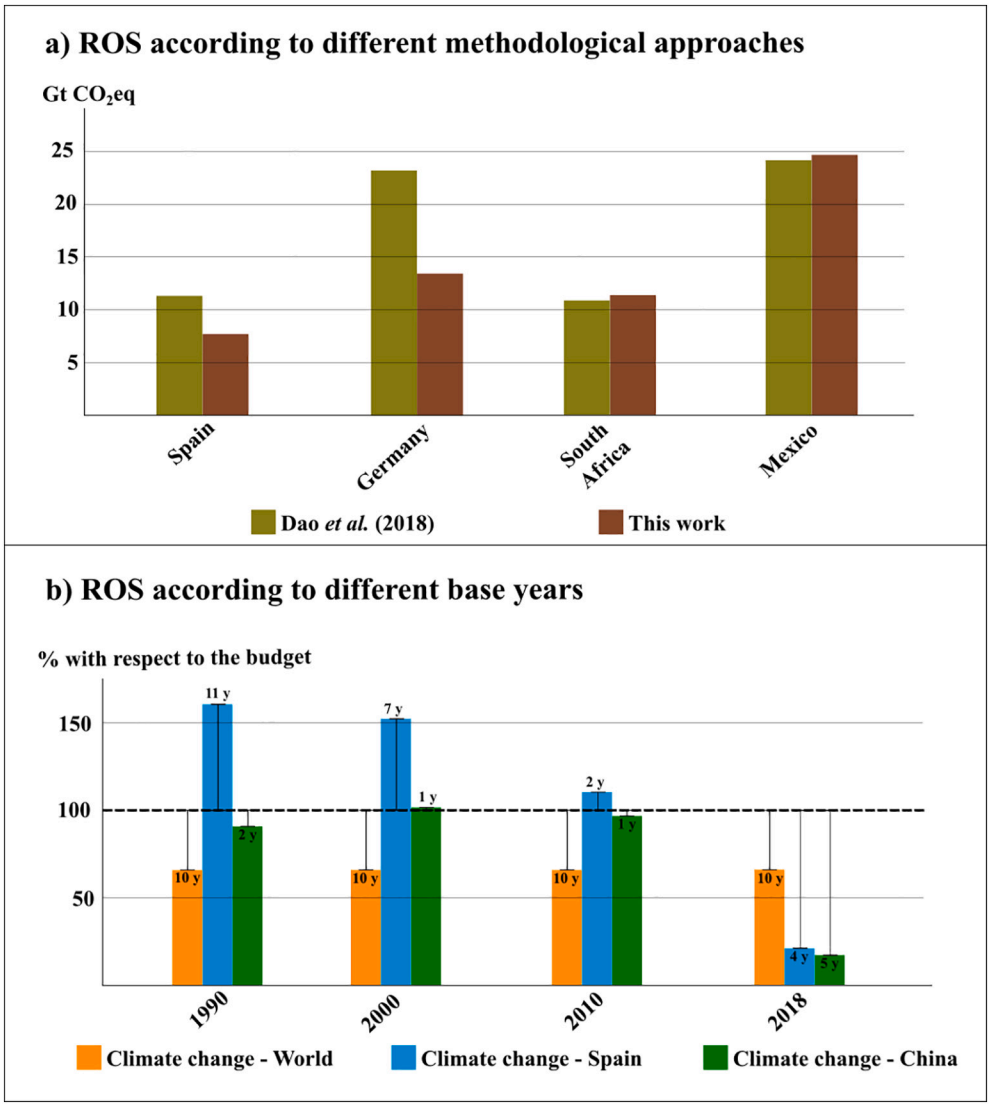


Fig. 3. Different methodological approaches to define budget limits. a) Comparison between the Dao et al. (2018) and this article methodologies for the computation of the initial budget for the climate change category in four countries and 1990 as the base year. b) Comparison of the use of different base years to establish the initial budget in two countries and at the global scale for 1990, 2000, 2010, and 2018 (b). In b, vertical bars represent the percentage of the consumed budget. When bars are below 100%, the estimated years to reach the 100% level are shown. When bars are over 100%, the number of years since the budget has been consumed is shown.

China differs with respect to the Spanish case for the period 1990 and 2000: the more recent the base year, the higher its responsibility. This is the general behavior observed in countries with low levels of development in 1990, which is usually associated with a low magnitude of environmental impacts. From 2000 to 2018, especially between 2010 and 2018, China acted as a highly industrialized country, so its tendency during this last period was rather coincident with that of Spain. In addition, and even though the temporal responsibility of a social actor is not reflected when a more recent base year is selected, it could be advisable to adopt a more recent base year to facilitate international negotiations and undertake global joint actions.

4.2. ROS of Spanish regions and municipalities

The results of the regional ROS for the 17 Spanish regions (Autonomous Communities) and those for the 313 municipalities of Galicia are shown in Figs. 4 and 5, respectively. In both cases, the necessary reduction in the anthropization land category and the year when carbon neutrality is expected based on a linear reduction of emissions are shown.

Spain, the 17 Spanish regions, and the 313 municipalities of Galicia are initially required to reduce 14.6% of their impact in the anthropized land environmental category, and the required year to be carbon neutral is initially estimated in 2029. This is a direct consequence of applying the grandfathering principle, which means that the reduction effort will be, in relative terms, the same for all the actors. Nevertheless, when the correction factor is applied according to different criteria, different outcomes are obtained. Thus, regions and municipalities experiencing a higher population increase by 2030 are benefited from the initial distribution, as shown in Figs. 4A1, B1 and 5A1, B1, which were calculated according to scenario 1. Figs. 4A2, B2 and 5A2, B2, calculated according to scenario 2, show that regions and municipalities with a lower population density are benefited from the initial distribution, whereas in the case of scenario 3 the benefited regions and municipalities are those with a higher gross domestic product (Figs. 4A3, B3 and 5A3, B3).

These results indicate that reduction requirements are driven by the variables selected to readjust the initial distribution. For instance, Asturias and Castilla y León,[5] the regions with the highest population decrease, showed the highest reduction requirements. Several regions, such as Galicia or Cataluña, present different reduction requirements in the three scenarios because the correction factor applied differs in the three cases. The results corresponding to municipalities show the same pattern. Those municipalities showing the highest population increase are allocated with lower reduction objectives in scenario 1, such as

[5] The location of the Autonomous Communities can be consulted in Appendix A.

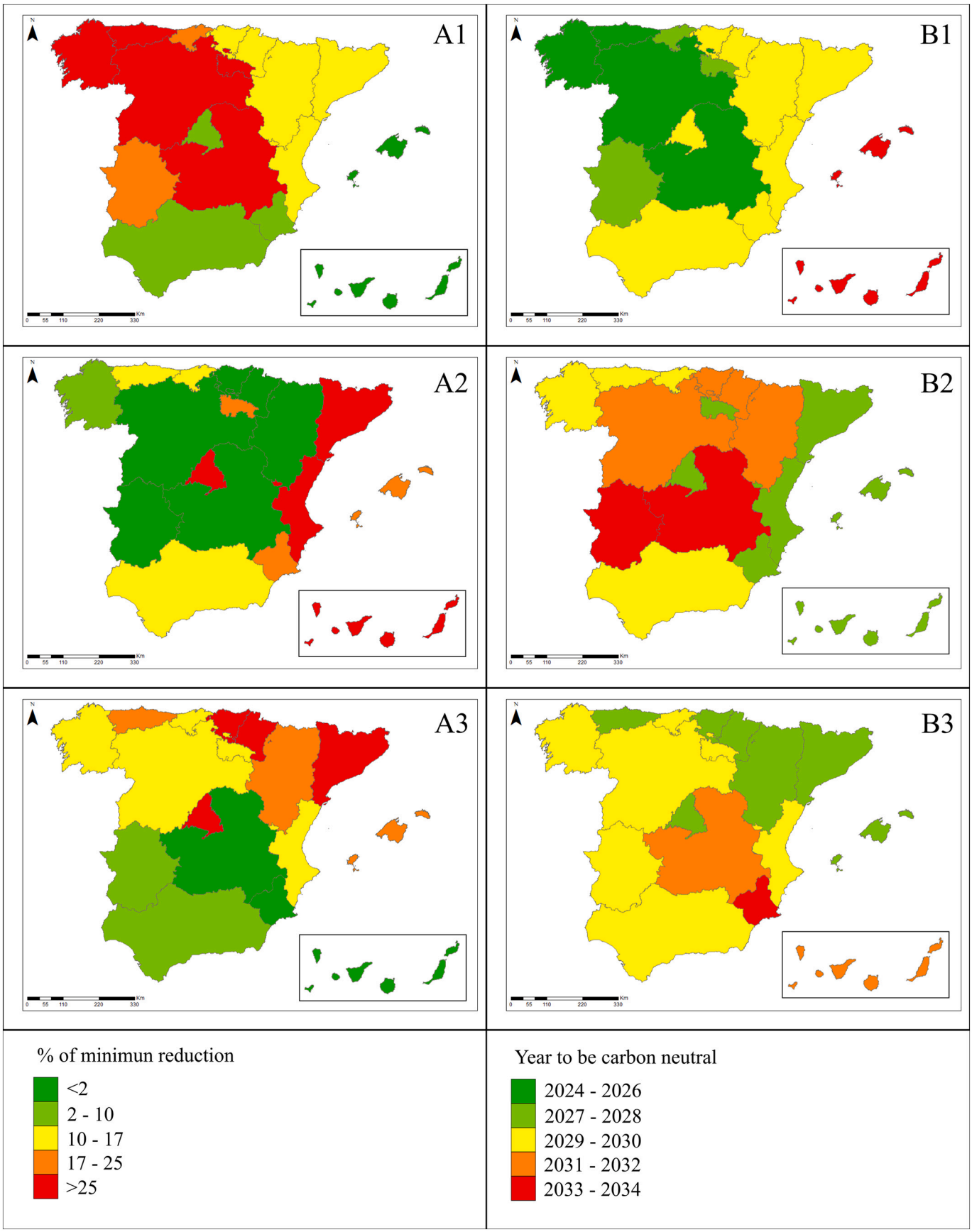


Fig. 4. Regional results. Figures in column A represent the necessary reduction of impacts in the anthropized land category. Figures in column B represent the year each region should become carbon neutral. A1 and B1 correspond to scenario 1, A2 and B2 correspond to scenario 2, and A3 and B3 correspond to scenario 3.

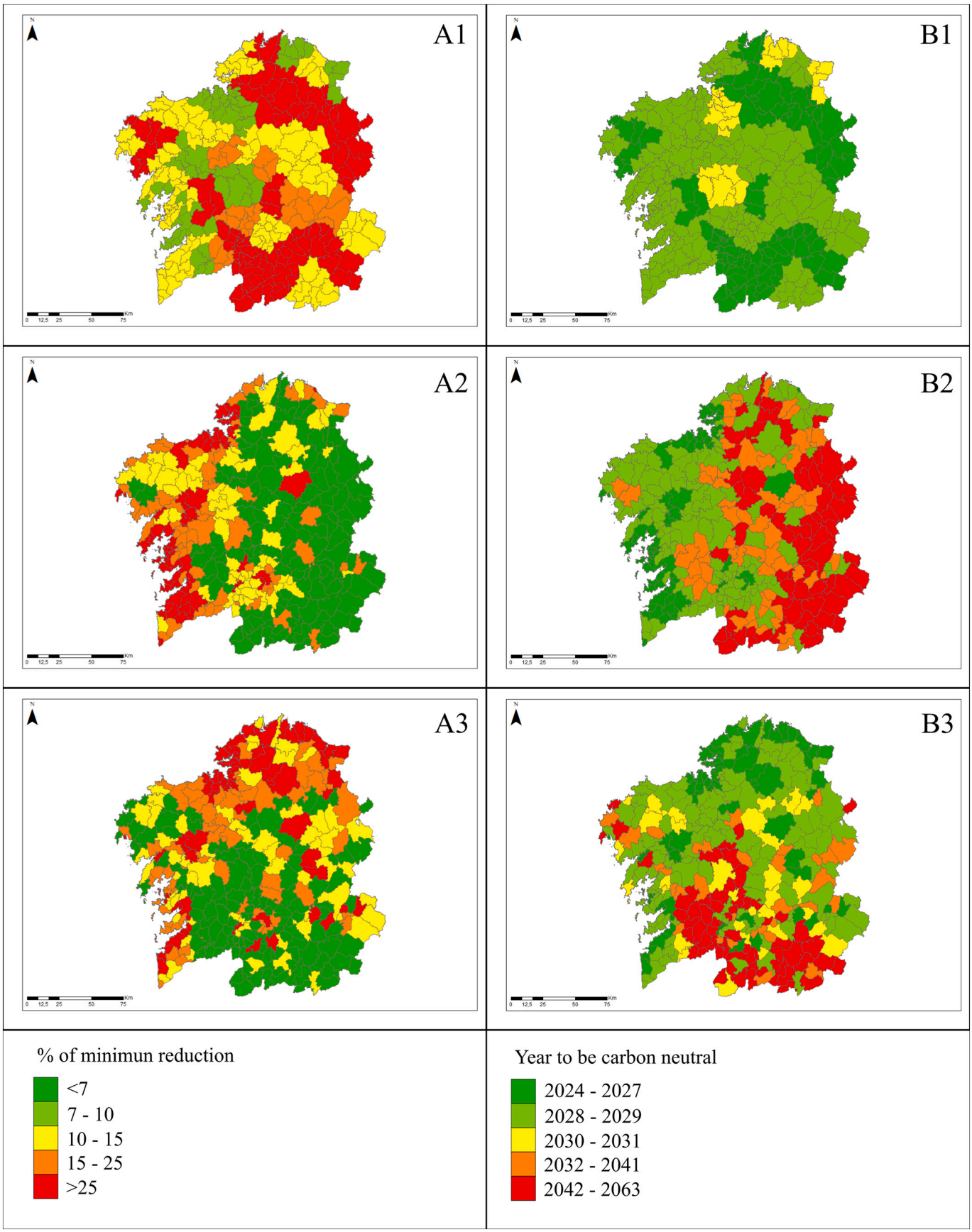


Fig. 5. Municipal results. Figures in column A represent the necessary reduction of impacts in the Anthropized land category. Figures in column B represent the year each municipality should become carbon neutral. A1 and B1 correspond to scenario 1, A2 and B2 correspond to scenario 2, and A3 and B3 correspond to scenario 3.

Table 4
Results for three sectors (S3, S4, and S5), an industrial park composed of four companies from four sectors (C1_IP, C2_IP, C3_IP, C4_IP), and three companies from the automotive sector (C1, C2, and C3).

ROS of	Code	CC (kg CO_2eq)	NF (kg N)	FU (Mm^3)	LSC (Km^2)	BL (PDF)
Sectors	S3	6.6E+10	5.7E+06	5.4E+00	5.9E+00	1.4E-06
	S4	4.1E+11	1.0E+07	3.8E+03	0.0E+00	0.0E+00
	S5	5.8E+10	6.3E+06	2.3E-01	7.7E-03	1.8E-09
Companies from S2	C1	2.9E+10	1.6E+06	3.2E-01	0.0E+00	0.0E+00
	C2	2.9E+09	1.6E+05	3.2E-02	0.0E+00	0.0E+00
	C3	2.9E+08	1.6E+04	3.2E-03	0.0E+00	0.0E+00
Companies from S2 (LCA)	C1	2.92E+11	1.62E+07	3.17E+00	0.00E+00	0.00E+00
	C2	1.46E+10	8.08E+05	1.58E-01	0.00E+00	0.00E+00
	C3	4.37E+09	2.42E+05	4.75E-02	0.00E+00	0.00E+00
IP (LCA)	IP	2.72E+10	1.13E+06	1.18E+02	6.21E-06	1.42E-12

Note: CC: Climate Change; NF: Nitrogen Footprint; FU: Freshwater use; LSC: Land-System Change; BL: Biodiversity Loss.

Ponteareas or Ames,[6] whereas A Coruña and Vigo, the most populated municipalities in Galicia, are assigned with higher reduction requirements than sparsely populated municipalities in scenario 2. Oleiros, the richest municipality with a per capita gross domestic product (> 20,000 euros), is the municipality with the largest reduction requirement. A significant fraction of the municipalities shows different ranges of reduction targets depending on the scenario (Fig. 5).

4.3. ROS of Spanish sectors, industrial park, and companies

The results presented in Table 3 were distributed among 163 sectors and several companies. A summary of the results for the ROS of these sectors is presented in Table 4, as well as the results for the industrial park and the companies. All the results for the sectors are presented in Appendix C. In Table 4, the results corresponding to climate change are expressed as a budget limit with 1990 as the base year. The rest of the environmental categories are expressed through the corresponding annual limits.

The calculated ROS depend on their contribution to the total environmental impact of national production. Land system change and biodiversity loss show a ROS of zero in the energy sector. This is a limitation of the methodologies employed to calculate both environmental impacts as they use a similar mathematical basis, where the environmental impact depends on the coverage of several types of ecosystems: cropland, pastures, urban land, and forests. The territory used by this sector is negligible in comparison with other sectors. Companies from the fishing sector (S1) and the automotive sector (S2) present the same situation.

Nevertheless, the zero impact of these environmental categories is not expectable. Furthermore, the biodiversity loss only includes territorial biodiversity loss considering mammals, reptiles, amphibians, and birds. These limitations should be borne in mind for the implementation of the ROS framework and illustrates the need to further improve these methodological approaches. If companies act as a part of an industrial park, the values presented in Table 4 can be used as the common goal for all of them. In other words, if the companies took part in an industrial park, the companies involved could impact beyond their limits provided that the industrial park as a whole is under the defined limits.

5. Discussion

5.1. Main contributions and limitations of this study

Before the main contributions and implications of this study are discussed, several methodological limitations should be recognized. Firstly, the ROS framework was designed from a management perspective, and is supported by advances in scientific fields such as ecology or environmental sciences. Both the review of indicators performed in Appendix A and the outcomes of this study when calculating the ROS of territories and companies reveal that there is still space for improvement in the methodologies used to define boundaries for GECs, and hence the usefulness of the framework is constrained by the quality of the environmental indicators initially selected.

An important limitation of the defined framework is the lack of economic and environmental data to perform EEIOA and LCA of territories, sectors and companies. In addition, the downscaling method, based on EEIOA, does not escape from critiques (see e.g. Moran and Wood, 2014), some of which come from the use of too many estimations and data aggregation for the development of databases. This calls for an increased effort of governments and the scientific community to provide economic and environmental data and points out to the need for further research on these methodologies, such as for instance the works of Tukker et al. (2018) and Cabernard and Pfister (2021).

The diversity of social organizations operating in the different world countries implies that the allocation of responsibilities related to GECs is likely to be distributed by a complex multi-scale governance process that considers this social diversity (Downing et al., 2019). In addition, the applicability of the framework is currently limited to the voluntariness of organizations. The inexistence of a strong international organization hosting and monitoring the agreements results in the lack of a governance structure to ensure international cooperation and achieve the global environmental goals.

Despite the limitations described above, this study fosters advance towards both the common governance of the GECs and the comprehensive management of environmental sustainability. This work demonstrates that, even with the methodological limitations and scarcity of data, it is already possible to decentralize the management of GECs even at municipal and company levels, and to integrate the reduction targets of the different social actors within a territory defined here as a key territorial unit. Decentralization of GECs management has been previously detected as a key element to successfully address environmental sustainability (Ranängen et al., 2018), but, as far as we are aware, no previous studies have shown that calculating coherent environmental objectives with a high resolution is possible. Moreover, to the best of our knowledge, the integration of the environmental objectives of consumers and producers has not been previously performed in such an integrative manner. Since the production and the consumption process can be connected within the key territorial unit, the accomplishment of the goals by companies (producers) will result in quantifiable benefits for territories (consumers) and vice versa.

5.2. Implications for policymakers

The results of this study open several avenues for policymakers. The responsible operating space presented a comprehensive view of the environmental concerns which need attention to foster the strong sustainability perspective in economics, i.e. social or economic capital

[6] The location of the municipalities can be consulted in Appendix A.

cannot replace some benefits that nature gives to humanity (Barbier and Burgess, 2017). Thus, policy agendas and business strategies are encouraged to address all the goals related to environmental sustainability, i.e. all global and local environmental concerns using different indicators to monitor the same environmental category if necessary. Although there is not an international organism able to guarantee the accomplishment of these global environmental goals, this study points out to existing mechanisms to monitor the responsibility of the different social actors on the GECs, and hence it encourages national governments to assume their own responsibility within the economic developing plans, an essential step to promote a "biophysically grounded and socio-economically responsible" development (Häyhä et al., 2016). The application of policies based on direct impacts without enough international control, can result in the redistribution of impacts to other regions of the world, and not to reduce them (Meng et al., 2018; Beylot et al., 2020). In this context, a footprint perspective in the design of economic plans becomes essential.

We have already mentioned that data availability and methodological shortcomings limit the rigorous computation of environmental budgets when applying the ROS framework. Therefore, to develop further the capabilities of the ROS framework and thus its usefulness for policymaking, improving both the quantity and quality of environmental and economic data provided by regional and national governments would be required. As discussed above, some planetary boundaries, such as those related to biodiversity loss or the atmospheric aerosol loading should be refined to improve the allocation procedure among the different social actors.

Nevertheless, the importance of data and methodological shortcomings can be minimized by revisiting the concept of environmental limits. We are aware that focusing on limits, boundaries or environmental ceilings can be counterproductive for the sustainability discourse, a shared criticism to both the planetary boundaries and the carrying capacity concepts (Downing et al., 2019). However, as initially demanded by Rockström et al. (2009) and Steffen et al. (2015), the objective is not allocating quotas, but reducing environmental impacts as much as possible. This implies that policymakers should use the ROS framework for economic plans design taking into account that the reduction targets for both the GECs and the TECs are estimates to guide the transition towards a more drastic transformation. This statement reduces the importance of using highly accurate accountabilities, since the environmental objectives of all social actors coincide on minimizing their impacts, promoting resilience and designing regenerative systems (Raworth, 2017; Downing et al., 2019).

Our results also show that 77% of the climate change related to consumption is generated by goods and services produced within Spain, as well as, for instance, 66% of the nitrogen footprint and 70% of freshwater consumption. This connection between consumption and production motivates the use of a key territorial unit to determine environmental objectives for both the production and consumption processes, what implies that territories (nations) depend on companies to achieve their objectives, and companies depend on the consumption of citizens from all territorial levels (regions and municipalities). Therefore, the joint performance of all the social actors will be determinant for the success of the economic plan within the key territorial unit. This conclusion implies that all social actors (governments, consumer alliances, sectors and companies) need to cooperate altogether to guarantee the success of an economic plan designed to preserve nature and the derived ecosystem services.

5.3. Implications for circular economy implementation

Despite the controversy around the CE concept, it is currently a key point of policy agendas to address environmental sustainability (Korhonen et al., 2018b). CE is supposed to adjust the extraction of resources and generation of wastes and emissions to suitable values for environmental sustainability (Suárez-Eiroa et al., 2019). To this aim, a wide range of strategies can be applied: recovering, remanufacturing, repurposing, repairing, recycling, rethinking, etc. (Reike et al., 2018). The ROS framework claims for prudence to affirm sustainable practices when implementing CE systems. From a strong sustainability perspective, improving efficiency or recycling rates does not necessarily mean acting sustainably. Indeed, circularity can coexist with an unsustainable production-consumption system (Pauliuk, 2018). Therefore, for CE to promote sustainable economic development, strategies and actions plans need to be constantly linked to an absolute environmental framework that determines the sustainability boundaries of the ecological system. The proposed ROS framework can be directly used for establishing such boundaries, and hence to define the reduction targets and the environmental impact limits that should be included into CE strategies.

A direct consequence of applying the ROS framework is that the different implementation levels of CE become connected. The definition of integrated environmental goals for the different social actors results in a panoramic view of the distribution of responsibilities to achieve environmental sustainability within the key territorial unit. Whatever the mix of policies and actions implemented for the different social actors within the KTU matches the environmental goals, the CE system could be considered to be successful. This fact does not end the discussion about how to integrate the action plans at different levels of implementation in practice, a key concern to advance towards CE (McDowall et al., 2017), but it may foster the maintenance of the final (environmental) objectives in perspective during the procedure.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolecon.2021.107258.

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