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ECONOMÍA CIRCULAR Y SERIES TEMPORALES: UN ANÁLISIS PARA EUROPA Y ESPAÑA

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"Economics gives no signs of acknowledging the role of natural resources in the economic process. Economists still do not seem to realize that, since the product of the economic process is waste, waste is an inevitable result of that process and ceteris paribus increases in greater proportion than the intensity of economic activity."

— *Georgescu-Roegen, Nicholas (1971). The Entropy Law and the Economic Process*

Index of contents

Resumen (Spanish)

Uno de los retos más relevantes del siglo presente es dar una respuesta global al modelo de crecimiento económico imperante desde la revolución industrial que se asentó sobre el consumo de masas en el siglo XX. En concreto, se trata de realizar una transformación profunda tanto desde el aparato teórico a la hora de modelizar conjuntamente economía con sociedad y ecología, así como una revisión práctica extendiendo el diseño y análisis de políticas públicas a una medición más profunda de los efectos ambientales generados por los distintos modelos de crecimiento y desarrollo económico.

Esta transición se ha convertido en uno de los ejes que guían la política económica de la Unión Europea con las denominadas Estrategias Europa 2020 y Europa 2030. Asimismo, los Objetivos de Desarrollo Sostenible (ODS) introducen transversalmente una actualización de las agendas de los gobiernos nacionales, autonómicos y locales.

Más particularmente en el caso europeo (Parlamento Europeo, 2019), la Comisión Europea presentó un plan de acción sobre la economía circular en 2015, acompañado de normativa sobre residuos y vertidos, reutilización de las baterías de los vehículos o la reducción de los plásticos. Más concretamente, resultan aplicables las cuatro Directivas [(UE) 2018/849, (UE) 2018/850, (UE) 2018/851 y (UE) 2018/852], que incorporan objetivos con importantes implicaciones en las economías de los países europeos:

- Un objetivo común de la Unión de reciclar el 65 % de los residuos urbanos antes de 2035 (55 % en 2025 y 60 % en 2030).
- Un objetivo común de la Unión de reciclar el 70 % de los residuos de envases a más tardar en 2030.
- Un objetivo vinculante de reducir el depósito en vertederos a un máximo del 10 % de los residuos municipales de aquí a 2035.

- La prohibición del depósito en vertederos de los residuos recogidos por separado, exigiendo la recogida selectiva de biorresiduos en 2023 y de tejidos y residuos peligrosos de los hogares antes de 2025.
- Otros objetivos para promover la reutilización y la responsabilidad extendida del productor.

Ante la existencia de un creciente número de objetivos supranacionales, y también nacionales y regionales, resulta vital desde la investigación académica aportar información, indicadores y conclusiones para poder evaluar los efectos de las políticas públicas introducidas hasta la fecha, así como poder sugerir algunos principios que sirvan de guía para un mejor diseño de las políticas públicas futuras.

En este debate juega un papel fundamental la desmaterialización del crecimiento económico, desde un punto de vista de eficiencia, ya que se busca una reducción por unidad de producción del uso de materiales desacoplando el crecimiento económico de ciertos efectos nocivos como la contaminación, el consumo de agua, energía o la propia generación de residuos. Esta visión persigue la investigación e implementación de aquellos procesos y dinámicas con menor impacto ambiental o más eficientes en términos ambientales, de forma que el crecimiento económico se vaya desligando de todos estos efectos no deseados. Esta vertiente se acerca también a la promoción de una economía más ligada a los servicios y al conocimiento, al ser menos intensivos en materiales que otras alternativas de generar valor añadido. Por otro lado, existe un enfoque basado en la equidad y la sostenibilidad intergeneracional, que se enfocaría en mantener por una parte los bienes, servicios y procesos ecológicos a lo largo del tiempo, disponibles para toda la población mundial, ahora y para las próximas generaciones. Esto implicaría una revisión del modelo económico más profunda, imponiendo restricciones ambientales y cambiando los objetivos socialmente deseables, pasando del crecimiento económico a otros indicadores alternativos ligados a la satisfacción de necesidades, la calidad de vida y la equidad social, también intergeneracional.

Ante este panorama, la Economía Circular surge como un marco híbrido que conjuga ciertos aspectos teóricos ligados a la economía ambiental (Pearce y otros, 1990) con aspectos prácticos aplicados a la empresa, integrándolas en las decisiones de producción y distribución, a las ONG y al ámbito social, así como en las principales organizaciones internacionales como las NNUU, la OCDE o la Unión Europea. Uno de los puntos de referencia es la consideración de que la eficiencia económica se consigue con un sistema lo más cercano posible a las 3Rs (Reducir, Reutilizar y Reciclar), que algunos autores han extendido con idéntico propósito: desmaterializar la economía generando menos residuos por unidad de producción, así como tender al reaprovechamiento de los materiales que ya se han producido. Este marco teórico resulta de interés, de cara a la medición de los efectos medioambientales de las políticas públicas en términos de eficiencia económica o de sostenibilidad tratando además de integrar a la tradición económica las nuevas ideas de circularidad y sostenibilidad. Entre algunos de los trabajos teóricos que han prestado atención a definir y analizar la economía circular se encuentran Ghisellini y otros (2016), Kirchherr y otros (2017) o. Winans y otros (2017). Sin embargo, no quedan resueltas algunas cuestiones como la existencia de efectos rebote indeseados sobre la generación de residuos que puedan contrarrestar las ganancias de eficiencia (Zink y Geyer, 2017). Es por ello que algunos autores como Pérez-Lagüela y otros (2019) han propuesto la Economía Espiral como una actualización de la anterior, asumiendo la existencia de ciertas limitaciones físicas y sociales que hacen que sea imposible hablar de una economía plenamente circular. También es apreciable el debate abierto por Parrique y otros (2019), al apuntar a las limitaciones, en términos de efectividad, de las políticas de crecimiento verde que se han aplicado en Europa durante las últimas décadas al propugnar un desacople que no se ha llegado a materializar de forma suficiente. Esta visión resulta más crítica, al propugnar una respuesta multidisciplinar que integre el cambio del modelo socioeconómico, a la par que se implementan nuevos procesos para promover el reciclaje, el ecodiseño o la prevención, lo que conjugaría de alguna manera la perspectiva de eficiencia con la de equidad al abordar la sostenibilidad de forma integral.

Otro de los puntos que nutren el cuerpo de esta tesis es la consideración de que la realidad es dinámica y cambiante, por lo que las series temporales se presentan como la forma esencial de analizar los fenómenos económico-ambientales para estudiar las relaciones y sus cambios en un horizonte lo más largo posible. En esta línea, entendemos que un análisis de las políticas públicas y sus efectos sobre la economía han de tener una medición a lo largo del tiempo y comparada con la situación previa y posterior, no sólo en los periodos inminentemente próximos, sino atendiendo incluso a posibles cambios estructurales que puedan imponer a las relaciones económico-ambientales.

Más particularmente, esta tesis busca acercarnos a la evolución de la relación entre crecimiento y desarrollo económico con respecto a la generación de residuos urbanos. Además, será de importancia el grado de recuperación y reintegración en el ciclo productivo de estos residuos como una forma de medir los resultados del sistema económico en términos de impacto ambiental con motivo del consumo de la sociedad. Por todo ello, en el proyecto de esta tesis se definían los siguientes objetivos:

1) Analizar los determinantes socioeconómicos de la generación de residuos y del reciclaje y su influencia sobre las dinámicas de las variables de economía circular.

2) Estudiar si las fases del ciclo económico, y en especial la recesión de 2008, generan efectos sobre las variables de economía circular.

3) Medir la posible convergencia regional en la generación de residuos y en el reciclaje a nivel nacional y europeo.

4) Encontrar las actuaciones públicas que han resultado más relevantes para modificar los patrones de generación de residuos o aumento del reciclaje.

5) Analizar las dinámicas de los residuos urbanos atendiendo a la capacidad para ser recuperados por el sistema productivo (circularidad).

6) Estudiar los distintos flujos de residuos urbanos: orgánico, papel y cartón, vidrio metales y plásticos para conocer la evolución de su composición, separación y reciclaje, prestando una especial atención al plástico.

Para estudiar los objetivos anteriores se plantea una estructura de cuatro capítulos que abordará los siguientes temas de una forma autocontenida:

En el primer capítulo, entraremos a estudiar la convergencia o divergencia regional a la hora de la generación de residuos urbanos por persona en el primer capítulo, lo que nos mostrará si estamos ante un fenómeno homogéneo a nivel nacional, regional o local. En este punto, adelantaremos algunas de las variables que parecen estar más ligadas a que existan diferencias regionales. En un segundo capítulo, entraremos a analizar la relación desde finales del siglo XX hasta la actualidad, pasando por la Gran Recesión, entre el ciclo económico y la generación de residuos, reflexionando sobre aspectos como el desacople entre crecimiento económico y residuos urbanos, aunque también lo extenderemos a un concepto más amplio de desarrollo al introducir el Índice de Desarrollo Humano. Posteriormente, en el tercer capítulo, se analizará este desacople entre economía y residuos urbanos, pero entrando en el grado de sostenibilidad del tratamiento de residuos, incorporando la evolución de la fracción de residuos que son recuperados (la suma de los residuos urbanos reciclados y sometidos a compostaje) y pueden, por tanto, ser reutilizados posteriormente. En este capítulo, nos reencontraremos con la importancia de la heterogeneidad, así como con diversas limitaciones que parecen aplicarse desde diversos campos para que la evolución hacia una economía plenamente circular haya sido tan intensa.

Finalmente, en el cuarto capítulo se ha estudiado el grado de reciclaje de los envases, al ser de las fracciones inorgánicas más importantes, ya que son también las recogidas de forma separada, con todo lo que ello conlleva en términos de logística, infraestructuras, diseño de políticas a nivel regional o local, entre otras cuestiones. Se presta una especial atención a los efectos de la Gran Recesión y del COVID-19 en la evolución de los envases, así como se estudia separadamente el caso de los envases plásticos, al ser este uno de los materiales más controvertidos en los últimos años en cuestiones de envases y productos de un único uso, por sus consecuencias ambientales y sobre la salud de las personas y de los ecosistemas.

Aunque cada capítulo contiene una sección dedicada a discutir los resultados y tratar de analizar la efectividad de las políticas públicas y extraer las oportunas conclusiones, a continuación del cuarto capítulo se encuentra una sección de conclusiones que tratará de recoger, de forma unificada, los principales resultados, debates y conclusiones que han podido extraerse de la realización de esta tesis doctoral.

Cabe señalar, en este último punto, que la estructura de los capítulos está pensada con el fin último de ser publicados para dar a conocer los resultados de esta investigación, por lo que la estructura del presente texto será cuasi-equivalente a la de un compendio de publicaciones, con la salvedad de que los dos últimos capítulos todavía no han sido publicados a fecha de finales de octubre de 2023, si bien uno de ellos se encuentra en revisión y el otro se enviará próximamente. Los capítulos I y II se encuentran publicados con las referencias Alcay y otros (2020) y Alcay y otros (2021), respectivamente.

Abstract

One of the most important challenges of the present century is to provide a global response to the model of economic growth that has prevailed since the industrial revolution, which was based on massive consumption in the twentieth century. Specifically, it is a matter of undertaking a deep transformation both from the theoretical side in terms of modeling the economy jointly with the society and the ecology, as well as a practical revision, extending the design and analysis of government policies to a more in-depth measurement of the environmental effects generated by the different economic growth and development models.

This transition has become one of the pillars guiding the European Union's economic policy with the so-called Europe 2020 and Europe 2030 Strategies. Likewise, the Sustainable Development Goals (SDGs) introduce a transversal updating of national, regional, and local government agendas.

More particularly in the European case (Parlamento Europeo, 2019), the European Commission presented an action plan for a circular economy in 2015, supported by regulations on waste and landfills, reuse of vehicle batteries or the reduction of plastics. In particular, the four Directives [(EU) 2018/849, (EU) 2018/850, (EU) 2018/851 and (EU) 2018/852], which incorporate targets of major implications for European national economies:

- A common Union target to recycle 65% of municipal waste by 2035 (55% by 2025 and 60% by 2030).

- A common EU target to recycle 70% of packaging waste by 2030.

- A binding target to reduce landfilling to a maximum of 10% of municipal waste by 2035.

- A landfills ban on separately collected waste, requiring separate collection of bio-waste by 2023 at the latest and of textiles and hazardous waste from households by 2025 at the latest.

- Other targets to promote reuse and extended producer responsibility.

Given the existence of a growing number of supranational, national, and regional targets, it is crucial for academic research to provide information, indicators and conclusions to evaluate the effects of the public policies implemented until now. Thereafter, it will be crucial to suggest possible principles that could serve as a guide for a better design of future government policies.

Dematerialization of economic growth plays a fundamental role in this debate, from an efficiency point of view, as it seeks a reduction of material use per unit of output, decoupling economic growth from certain undesirable effects such as pollution, water and energy consumption or the generation of waste. This vision pursues the research and implementation of those processes and dynamics with less environmental impact or more efficient in environmental terms, so that economic growth is decoupled from all these unwanted effects. This approach is closer to the promotion of a services and knowledge-based economy, as they are less material-intensive than other alternatives for generating added value. On the other hand, there is an approach based on equity and intergenerational sustainability, which focuses on maintaining ecological assets, services and processes over time, available to the entire world population, now and for future generations. This would imply a more extensive revision of the economic model, imposing environmental restrictions and changing the socially desirable objectives, shifting from economic growth to alternative indicators linked to the fulfillment of needs, quality of life and social justice, including intergenerational equity.

In this context, the Circular Economy emerges as a hybrid framework that combines certain theoretical aspects linked to environmental economics (Pearce et al., 1990) with practical applications to business, integrated into production and distribution decisions, NGOs and the social sphere, as well as the main international organizations such as the UN, the OECD and the European Union. One of the points of reference is the consideration that economic efficiency is achieved with a system as close as possible to the 3Rs (Reduce, Reuse and Recycle), which some authors have extended for the same purpose: to dematerialize the economy by generating less waste per unit of production, and to tend to reuse materials that have already been produced. This theoretical framework is of interest for measuring the environmental effects of public policies in terms of economic efficiency or sustainability, as well as trying to integrate the new ideas of circularity and sustainability into the economic tradition. Among some of the theoretical works that have paid attention to define and analyze the circular economy are Ghisellini *et al.* (2016), Kirchherr *et al.* (2017) or. Winans *et al.* (2017).

However, some issues remain unresolved, such as the existence of undesired rebound effects on waste generation that may counteract efficiency gains (Zink and Geyer, 2017). This is why some authors such as Pérez-Lagüela *et al.* (2019) have proposed the Spiral Economy as an update of the previous one, assuming the existence of certain physical and social limitations that make it impossible to speak of a fully circular economy. The debate opened by Parrique *et al.* (2019) is also appreciable as they point out the limitations, in terms of effectiveness, of green growth policies implemented in Europe during the last decades. These policies have advocated a decoupling that has not adequately been achieved. This approach is more critical, proposing a multidisciplinary response that integrates the change of the socioeconomic paradigm, while implementing new processes to promote recycling, eco-design or prevention, which would somehow combine the perspective of efficiency with that of equity by addressing sustainability in a comprehensive manner.

Another of the points that feed the body of this doctoral thesis is the consideration that reality is dynamic and changing, and therefore time series are considered to be the essential way of analyzing economic and environmental phenomena in order to study the relationships and their changes over the longest possible horizon. In this line, we understand that an analysis of public policies and their effects on the economy must be measured over time and compared with the previous and subsequent situation, not only in the forthcoming periods, as well as attending to possible structural changes that may be imposed on economic-environmental relations.

More particularly, this thesis aims to approach the evolution of the relationship between growth and economic development with respect to the generation of municipal waste. In particular, the degree of recovery and reintegration into the productive cycle of these waste products will be of importance as a way of measuring the economic systems performance in terms of the environmental impact of societal consumption. For all the above reasons, we have defined the following objectives for this thesis project:

1) To analyze the socioeconomic determinants of waste generation and recycling and their influence on the dynamics of circular economy variables.

2) To study whether the phases of the economic cycle, especially the recession of 2008, generate effects on the circular economy variables.

3) To measure the possible regional convergence in waste generation and recycling at national and European level.

4) To find the most relevant public actions to modify waste generation patterns or increase recycling.

5) To analyze the dynamics of urban waste in terms of its capacity to be recovered by the productive system (circularity).

6) To study the different urban waste flows: organic, paper and cardboard, glass, metals and plastics in order to understand the evolution of their composition, sorting and recycling, paying special attention to plastics.

We propose a structure of four chapters that will address the following objectives in a self-contained manner:

We will study regional convergence or divergence in the generation of municipal waste per capita in the first chapter, which will show us whether we are dealing with a homogeneous phenomenon at the national, regional or local level. At this point, we will advance some of the variables that seem to be more linked to the existence of regional differences.

In the second chapter, we will analyze the relationship between the economic cycle and waste generation from the end of the 20th century to the present, including the Great Recession, reflecting on aspects such as the decoupling of economic growth and urban waste, although we will also extend this to a broader concept of development by introducing the Human Development Index.

Subsequently, the third chapter will analyze this decoupling between the economy and urban waste but will also examine the degree of sustainability of waste treatment, incorporating the evolution of the fraction of waste that is recovered (the amount of urban waste that is recycled and composted) and can therefore be reused later. In this chapter, we will again discuss the role of heterogeneity, as well as the various barriers that seem to apply from different fields for the evolution towards a fully circular economy to have been so intense.

Finally, in the fourth chapter we have studied the degree of recycling of packaging, being the most important inorganic fractions, since they are also the ones collected separately, with all that this implies in terms of logistics, infrastructure, policy design at regional or local level, among other issues. Special attention is paid to the effects of the Great Recession and COVID-19 on the evolution of packaging, and the case of plastic packaging is studied separately, as this is one of the most contested materials in recent years in terms of packaging and single-use products, due to its environmental consequences and its impact on the health of people and ecosystems.

Although each chapter contains a section dedicated to discussing the results and trying to analyze the effectiveness of public policies and extract the appropriate conclusions, after the fourth chapter there is a section of conclusions that will try to summarize, in a cohesive way, the main results, debates and conclusions that have been drawn from the execution of this PhD thesis.

It should be noted, on this last point, that the structure of the chapters has been conceived for the purpose of being published to spread the results of this research, so that the structure of the following text will be virtually equivalent to that of a compendium of publications, with the exception that the last two chapters have not yet been published at the end of October 2023. The third chapter is currently under revision and the final one will be submitted in the following weeks. The first and the second chapter have been published as Alcay *et al.* (2020) and Alcay *et al.* (2021).

Chapter I. A study of convergence in two indicators of waste generation efficiency for Spanish regions

1.1. Introduction

The transition to the Circular Economy, from a European perspective, is based on the resource efficiency agenda set out in the framework of the "Europe 2020 strategy for smart, sustainable and inclusive growth", supported by the "Roadmap to a Resource-Efficient Europe" whose priority objective is to turn the European Union into a low-carbon, resource-efficient, green and competitive economy. Since the early 2000s, the European Commission has been promoting the creation of a circular economy associated with a 'zero waste' policy by combining policy actions and initiatives with a network of organizations. However, zero waste is challenging to achieve. Usually, Circular Economy concept can be closely related to the 3R Principles: Reduce, Reuse, and Recycle, as described in Tisserant *et al.* (2017), Ghisellini *et al.* (2016) and Lieder and Rashid (2016). One of the pillars of the circular economy is "closing the loop" to generate an integrated waste management system by recovering materials, but also by using the energy of the waste. Thus, organic waste can be reused for various purposes, such as reducing energy or raw material dependency. Tomic and Schneider (2018) summarize the alternatives for converting such waste into energy. Sharma *et al.* (2020) propose hydrogen obtained through certain innovative biochemical processes as an energy source while D' Adamo et al (2019) present the case of bio-methane as an important energy source in the transport sector in Italy.

The importance of waste management analysis can be understood if we consider the large amount of literature that has recently been generated in this area. Without being exhaustive, we can cite the works of Kashwan (2017) where the relationship between waste production and inequality is analyzed; and Corsini *et al.* (2018), which focuses on awareness of the environmental impact and the willingness to assume personal actions is decisive for reducing waste generation. Lieder and Rashid (2016) conclude that joint support of all stakeholders is necessary in order to successfully implement the CE concept at large scale and Tamayo et al (2017) find that economic incentives are useful for reducing waste and promoting recycling. Other works such as Lavee and Khatib (2010) and Lavee and Nardiya (2013) create a model that estimates the expected costs of making a transition to recycling in Israel so that the government has a decision-making tool to award grants efficiently. All of them coincide on the need to take measures by policy makers so that member countries reduce the volume of solid waste and also the differences in waste management efficiency between them.

The works of Castillo *et al.* (2019a, 2019b) indicate how much progress has been made in this respect and how the approval of the European guidelines on waste generation has achieved, first, greater efficiency in the generation and treatment of waste and, above all, a reduction in the differences between European Union countries, especially between some Central and Northern European countries such as Denmark, Austria and Germany, and Eastern European countries that joined the European Union in the 2000s, whose performance is poor. This is the case in Croatia, as can be seen in Luttenberger (2020), which faces several problems in the implementation of the waste treatment and recycling system. Di Maria *et al.* (2020) point to the targets set by the EU as an effective way of reducing waste and increasing recycling in Europe. They also consider the opportunity for Europe to set up a new line of economic activity that will reduce emissions of polluting gases, improve health and create jobs. In spite of the fact that the enactment of the current Waste Framework Directive in 2008 has clearly favored convergence among the EU-27, the differences are still significant.

The results of these papers are to some extent expected given the heterogeneity that exists between the considered countries. The question that remains is what can happen when an environment is analyzed that in principle should present greater similarities, as is the case of regions within the same country. The case of Spain is of great interest in this regard, given the economic, fiscal and cultural differences between the Spanish regions, as well as the high degree of disaggregation of political decision making. In this regard, we should take into account that the Spanish territorial implementation of waste treatment is diverse owing to the autonomous nature of regions which enjoy key competences in multiple areas. Following the Spanish legislation on waste, Law 22/2011, which transposes the European Directive 2008/98/EC, Spain has the power to set targets for waste reduction, based on European criteria, and to develop an annual strategy to achieve them. Regional governments can establish their own waste prevention plans, develop their own waste legislation and have the powers to monitor, inspect and sanction production and waste generation activities. Municipalities regulate the management of waste collection and treatment services, as it is their obligation to guarantee such services. Therefore, if the objective is to reduce the generation of waste at a national level, there needs to be a common approach in all Spanish regions, something that does not seem to be occurring at this time.

This lack of homogeneity in taking environmental decisions makes it very difficult to implement a single pattern of behavior for waste generation in the Spanish regions. The results of Expósito and Velasco (2019), who employ data for 2013, reveal important differences in the Spanish regional recycling market. Similar results are obtained in Pérez-López *et al.* (2018) and Bel and Fageda (2010), who study the effects of economies of scale, intermunicipal cooperation and management issues on the costs of recycling service; and in Díaz-Villavicencio *et al.* (2017), who analyze the implication of education and workers' training on recycling. This heterogeneity is also found in other regional analyses. For instance, de Jaeger *et al.* (2011) analyze the differences in waste collection using data from 299 municipalities in Flanders, Belgium, for the year 2003. Similarly, Agovino *et al.* (2019) find differences between waste collection policies in Italian municipalities for 2012. All of these studies coincide in revealing the presence of very heterogeneous regional behaviors regarding the generation and treatment of waste. Another vector that can generate heterogeneity between regions is the presence of tourist activities. Falcone (2019) points out that tourism represents an important determinant of waste generation, which in turn can diminish the tourist appeal of the area if the problem is not properly managed. However, we should note that the methods employed in these studies only take into account one year or, at least, the time dimension is scarcely considered. Therefore, it seems appropriate to perform an analysis with the help offered by the temporal dimension, so that this dynamic component can be taken into account and the results can be interpreted from a long-run perspective.

Against this background, the aim of the chapter is to determine whether there is a similar pattern of behavior in the recent development of waste generation across the Spanish regions or whether, by contrast, several patterns of behavior can be found. To that end, and following Castillo *et al.* (2019a, 2019b), we can apply the statistics proposed in Phillips and Sul (2007, 2009) to test the null hypothesis of convergence for a pool of data. If we are unable to reject the hypothesis, we can conclude in favor of the existence of a common behavior between all the Spanish regions in terms of waste generation. However, if we are able to reject the hypothesis, then we will be able to identify multiple patterns of behavior and, consequently, determine the regions associated to them and the forces that may drive them.

The rest of the chapter is organized as follows. Section 2 describes the data and the methods. Section 3 discusses the results obtained. Section 4 concludes.

1.2. Data and methods

1.2.1. Data

The variable under analysis is municipal solid waste (MSW) collection. These data have been obtained from the Spanish Institute of Statistics (INE). The data covers the period 1998-2016 and we have considered the 17 Spanish regions. The data for the two autonomous cities (Ceuta and Melilla) are not available and, subsequently, we have excluded them. The data are measured in physical units of mass of all the urban waste collected by authorized managers throughout the national territory. Gross domestic product (GDP) and population series have also been obtained from Spanish Institute of Statistics (INE) database.

Using these three variables, we can elaborate two different indicators. On the one hand, we use an indicator to see the evolution from the point of view of the total production of the economy. This is defined as the ratio between MSW and GDP and reflects the productive efficiency of the regions with respect to waste generation, in the sense that the lower the waste generation per unit of GDP, the more environmentally efficient the region. On the other hand, we can analyze waste prevention from the perspective of household consumption. To that end, we will take into account the per capita generated waste (MSW/population), this indicator providing us with information on the consumption habits of the regions and their environmental impact.

The explanatory variables employed in Section 3 have also been obtained from INE. More details of these variables are provided in the Appendix 1.

1.2.2. Convergence and Phillips-Sul methodology

Convergence has been defined in the economic literature as a process where the dispersion of a variable, usually per capita GDP, reduces for a group of countries or regions. At the limit, when the variance is 0, all the components of this group show the same value of the variable and, therefore, exhibit a similar per capita

GDP. The interest in this type of analysis grew due to the seminal paper by Barro and Sala-i-Martí (1992) which opened the door to a very large number of papers devoted to the analysis of convergence. In this regard, we should cite the papers of Carlino and Mills (1993, 1996) and Bernard and Durlauf (1995), where the concept of stochastic convergence is developed, and those of Payne *et al.* (2017) and Solarin (2019), where this concept of convergence is analyzed.

However, none of these papers develop or use a statistic that focuses on testing the null hypothesis of convergence. This problem is considered in Phillips and Sul (2007, 2009), PS hereafter, who designed a very popular statistic that has been extensively employed to test for convergence. We can cite the papers of Camarero *et al.* (2013a, 2013b), Kounetas (2018) and Apergis and Payne (2019) in this regard. Finally, Castillo *et al*. (2019a, 2019b) also employ this methodology to analyze the evolution of waste efficiency in EU countries.

Following PS, let us consider that X_{it} represents either of the two measures of waste generation, with i=1, 2, …, 17 (the 17 Spanish regions) and t= 1998, …, 2016. This variable can be decomposed as $X_{it} = \delta_{it}\ \mu_t$, where μ_t is the single common component and δ_{it} is the time-varying factor loading coefficient that measures the idiosyncratic distance between the common trend components μ_t and X_{it}. PS suggest testing for convergence by analyzing whether δ_{it} converges towards δ. To do so, they first define the relative transition parameter, as follows:

$$h_{it} = \frac{X_{it}}{N^{-1}\sum_{i=1}^{N} X_{it}} = \frac{\delta_{it}}{N^{-1}\sum_{i=1}^{N} \delta_{it}} \qquad (1)$$

This parameter describes the transition path for the i-th region relative to the panel average. In the presence of convergence, δ_{it} converges towards δ and, therefore, h_{it} should converge towards 1, while its cross-sectional variation, H_{it}, which is defined as follows:

$$H_{it} = N^{-1}\sum_{i=1}^{N}(h_{it}-1)^2 \overset{As}{\to} 0, as\ T \overset{as}{\to} \infty \qquad (2)$$

should go to 0 when T goes towards infinity. Then, PS test for convergence by estimating the following equation:

$$log\frac{H_1}{H_t} - 2log[\log(t)] = \alpha + \beta\log(t) + u_t, t = T_o, \dots, T \quad (3)$$

with $T_o = [rT]$, and r=0.3. Equation (3) is commonly known as the log-t regression. The null hypothesis of convergence is rejected whenever parameter β is lower than 0. PS suggest estimating model (3) by methods which correct for the presence of autocorrelation and heteroskedasticity and, later, employ the t-statistic to test the null hypothesis β=0. The use of these robust methods ensures that this t-ratio converges towards a standard N (0,1) distribution and, therefore, we will reject the null hypothesis of convergence whenever this t-statistic takes values lower than -1.65.

If we reject convergence, PS propose the following robust clustering algorithm for identifying clubs in a panel:

i. Order the N states according to their final values

ii. Starting from the highest-order state, add adjacent states from our ordered list and estimate model (3). Then, select the core group by maximizing the value of the convergence t-statistic, subject to the restriction that it is greater than -1.65.

iii. Continue adding one state at a time of the remaining states to the core group, and re-estimate model (3) for each formation. Use the sign criterion (t-statistic >0) to decide whether a state should join the core group.

iv. For the remaining states, repeat steps (ii)–(iii) iteratively and stop when clubs can no longer be formed. If the last group does not have a convergence pattern, conclude that its members diverge.

PS recommend performing club merging tests after running the algorithm using equation (3) in order to avoid an over-estimation of the number of clubs.

Finally, we have followed the suggestion of PS and extracted the trend components of the series by filtering them using the Hodrick and Prescott (1997) filter, applying the standard value λ=400.

1.3. Results

1.3.1. Waste efficiency convergence

MSW/GDP ratio

The results of the application of the PS methodology are reflected in Table 1. The null hypothesis of convergence is clearly rejected when we analyze the MSW/GDP ratio. Then, there is no common pattern of behavior, and several clubs may exist, as the subsequent use of the PS cluster algorithm proves. We can observe the existence of 4 different clubs, whilst two regions (Madrid and Galicia) diverge. Club 1 is the group formed by Andalucía, Islas Baleares, Islas Canarias, Cantabria, Castilla-La Mancha and Extremadura. Asturias and Murcia belong to club 2. Cataluña, Castilla y León and the Comunidad Valenciana form Club 3. Finally, Aragón, Navarra, the País Vasco and La Rioja are included in club 4. Galicia and Madrid exhibit a different behavior and cannot be included in any of these clubs. Thus, they diverge. Figure 1 presents these results in a map.

In order to better understand the results obtained, the average values of the MSW/GDP ratio have been obtained for each one of the estimated clubs and are presented in Figure 2 jointly with the values of Galicia and Madrid. Club 1 exhibits the greatest values of the ratio at the end of the sample. Therefore, we can consider this club to include the least efficient regions. By contrast, the ratio of Madrid is the lowest, followed by that of club 4. Finally, we should comment on the case of Galicia. This was the region with the highest values of the MSW/GDP ratio at the beginning of the sample. However, its evolution has allowed it to remarkably reduce the waste generation and it is placed at the average at the end of the sample.

It is worth noting that there is a general decline for all the paths of the MSW/GDP ratio, showing the great effort made by all the regions to reduce their waste generation. Additionally, if we compare the range of variation of the MSW/GDP ratios, we can observe that the distance between the highest and the lowest value is greater at the end of the sample than at the beginning. This can be understood as additional evidence of the heterogeneity of the Spanish regions so far as waste generation is concerned.

Per capita waste

When we consider the per capita waste generation ratio, we can also reject the convergence null hypothesis, as can also be seen in Table 1. Then, we should again consider the presence of several convergence clubs. The use of the PS algorithm provides somewhat different club estimations. We can observe that the Islas Baleares diverge. Club 1 is composed of the Islas Canarias and Murcia. Club 2 includes Andalucía, Cantabria, Cataluña, Castilla-La Mancha and the Comunidad Valenciana. Aragón, Asturias, Castilla y León, Extremadura, Galicia, Madrid, Navarra, the País Vasco and La Rioja make up Club 3. Figure 3 reflects the club composition, whilst Figure 4 presents the average values of the per capita MSW generation for each club.

If we focus on Figure 4, we observe that per capita waste does not exhibit a very clear downturn trend. Rather, it remains at around the initial levels, and a somewhat negative trend is only observed after the Great Recession. Additionally, we can appreciate that the values of the Islas Baleares are clearly greater than the rest, followed by those of club 1, which includes the Islas Canarias. Then, given that the economies of these two regions largely depend on tourism, this fact may play an important role in the per capita waste generation, as we will discuss below. Finally, we can also observe that there is no reduction in the differences between the average values of the clubs and, therefore, the degree of heterogeneity is greater than that of the MSW/GDP indicator. Even worse, there is no sign that these differences will disappear in the near future.

1.3.2. Forces that may drive the club creation.

The results reported in the previous section have proved the heterogeneity of the evolution of waste generation management across the Spanish regions, reflected in the existence of several patterns of behavior. This section is devoted to an analysis of the forces that may drive the creation of these clubs. To that end, we have estimated the model:

$$y_i = x_i' \beta + u_i \ (i=1, 2,\ldots, 17) \tag{4}$$

where the dependent variable y_i may have various possible outcomes, each of them related to the number of clubs that the PS methodology has estimated. These different values imply a preference or an ordination of the clubs, which should be taken into account in the estimation. Therefore, ordered probit methods should be employed. The explanatory variables (x_i) have been selected from a set of general socioeconomic variables, such as per capita GDP, education level or public expenditure; other environmental variables, such as the number of recycling plants, landfills or homes that have reported environmental problems and, finally, variables that reflect the institutional context such as the transparency of administrations or the crime rate. These variables are defined in the Appendix. The final specification has been selected by following a general-to-particular strategy, where the non-significant variables have been iteratively removed. Finally, we should note that the quality of the estimations is limited by the scant length of the sample, given that we have only 17 possible observations. This sample availability would be even shorter if we excluded the divergent regions. Then, in order to maximize the degrees of freedom of the estimation, we have preferred to retain the divergent regions in the estimation. They have been incorporated into the probit as a separate group. This means that we have 6 groups for the MSW/GDP ratio and 4 groups for the per capita waste. In any event, we should note that the results presented here are robust to other allocations of the divergent regions.

MSW/GDP ratio

The results are presented in panel A Iof Table 2. The explanatory variables included in the final specification are the per capita GDP (pcGDP), the percentage of people between the ages of 25 and 64 who participate in training activities (EDU), the percentage of the service sector over the total DP (SERV) and the expenditure of local authorities in the Environment Spending Chapter (ENVIRON). Note that we can relate positive coefficients with higher efficiency and lower waste generation intensity, given that that belonging to a higher group indicates greater efficiency in waste generation (less waste per unit of GDP).

Then, we can observe that the higher the per capita GDP, the lower the MSW/GDP ratio of the region. This relationship is also valid for the level of continuing education and the level of environmental spending. By contrast, the dimension of the service sector has an opposite effect. The greater the service sector, the greater the MSW/GDP ratio.

The analysis of the results leads us to observe the importance of the economic structure and the general level of economic activity in relation to waste generation, as has been analyzed in Arbulú *et al.* (2015) or in Namlis and Komilis (2019). In the Spanish case, this factor is very relevant for those regions with great importance of tourist activities. Those regions having the largest service sector exhibit the highest degrees of MSW/GDP ratio. This result can be easily understood if we take into account that the environmental regulation of the service sector is more relaxed. Therefore, the greater the percentage of the service sector, the lower the efficiency.

Education and qualifications are also key in these industrial sectors that use highly trained workers. Then, industrial regions with a higher level of education will, in turn, end up being more reduced levels of waste. These characteristics are shared by the regions that make up Club 4 (País Vasco, Navarra, Aragón and Rioja), added to which we could incorporate Madrid as the most efficient region. They stand out for their high per capita income and for an economic structure with a greater weight of industry, agri-food or highly specialized services such as logistics, the financial sector or research. Higher and continuous education and training is also a feature of these regions.

Per capita MSW

The results obtained from the estimation of model (4) are reflected in Panel II of Table 2. The explanatory variables included in the final specification are the average household expenditure in the region (HOUSE_EXP), the number of university graduates (GRADUATES), the years of government of left-wing parties in the region (LEFT_WING), the proportion of people living in settlements of less than 10,000 inhabitants (DISPERSION) and the expenditure of local authorities on environmental items (ENVIRON).

The analysis of the estimated model leads us to very interesting insights. Household income is directly related with the per capita MSW ratio. This result should be interpreted with some caution. It might show that an increase in household income would result in a greater environmental impact, which would contradict the results obtained for the MSW/GDP ratio. However, we should take into account the fact that the Islas Baleares have the greatest per capita waste and, at the same time, one of the largest per capita GDP and household incomes. By contrast, the regions included in the estimated club 1 have the lowest values of these economic indicators, whilst those in estimated club 3 have the highest. Therefore, the relationship between income and per capita waste collection is clearly altered by the behavior of the Islas Baleares.

The effect of the educational level and provision of higher education is again of considerable importance in determining the level of waste per capita. The higher the education level, the in the lower the per capita MSW.

Left-wing parties have traditionally been associated with the extension of civil rights, social protection, or environmental sustainability as their priorities of government. The fact that the variable years of left-wing regional governance is significant would confirm that the ideology of the government matters so far as waste collection is concerned. In this regard, our results are consistent with the fact that left-wing governments are more aware of environmental concerns, as suggested by Harring *et al*. (2018).

The fact that territorial dispersion amounts to per capita waste generation implies that the territorial organization and spatial distribution of economic activity is not neutral with respect to waste generation. If the regions with the highest proportion of their population in nuclei with less than 10,000 inhabitants have a lower environmental impact measured in per capita waste generation, this implies that there are differences in rural and urban societies. It follows that the insular and urban grouping structure around the coast, in which various regions belonging to groups 1, 2 and 3 are grouped, is one of the explanations of the higher intensity of per capita waste generation.

In this regard, we should take into account the results of Kennedy *et al.* (2007). These authors observe a trend in cities over the last few decades to a greater use of materials, especially for the construction of new buildings. Given that, building materials are difficult to recover and recycle, it is to be expected that they will end up being dumped. Thus, the amount of waste generated may be increasing even in cities that have implemented an efficient recycling system for urban waste.

Finally, the results also suggest a clearly differentiated behavior between the Northwestern and the Mediterranean regions of Spain, with the latter showing a lower per capita waste generation than the former.

1.4. Discussion

Prevention in the generation of waste produces considerable environmental benefits and plays a key role in the roadmap to advance towards an Efficient Europe in the use of the resources within the 2020 Strategy of the European Union. Hence, all EU countries seek to design policies to reduce the generation of waste. In Spain, the political initiatives relating to the Circular Economy at national, regional, and local level have also been notable in recent years.

It should be noted that the effort made in Spain to contain waste generation has been truly remarkable, going from a total of 0.70 tons of waste per inhabitant in 1995 to 0.58 in 2016, a reduction of almost 20%. However, this effort has not been homogeneous. Our results demonstrate, on the one hand, the existence of clearly different regional patterns of behavior and, on the other hand, they identify variables that can help explain these differences. This will facilitate the design of policies aimed at reducing waste generation even further.

Our results indicate that the regions with a greater dependence on the tourism sector show a significantly worse performance both in the MSW/GDP ratio and, especially, in the per capita MSW. If we focus on this last measure, it can be seen that Islas Canarias, Islas Baleares and Murcia, regions where the tourism sector is very important, have double the average generation of waste than the regions with the best performance.

So, it seems appropriate to think about reducing the generation of waste in those regions with the worst performance in order to improve the global data for Spain. In this context, we should note that the works of Weber *et al*. (2019), Diaz-Farina *et al*. (2020) study the effect of the implementation of unit-pricing schemes in waste management in Spanish tourist areas. These authors conclude that this type of economic stimulus makes it possible to decrease the generation of waste and, at the same time, to increase recycling levels. Similar conclusions are drawn from other studies not focused on the Spanish case, such as Sakai *et al.* (2008) for Japan or Bueno and Valente (2019) who analyze the experience of Trento (Italy). The results of these works indicate that the reductions in waste generation are around 30% after applying these unit-pricing schemes. If we accept this figure as an achievable goal, the levels of waste generated from Islas Canarias and Murcia would decrease to around the average value of Spain. The case of Islas Baleares would improve, but the values would still be very high, and the effort would have to be greater and more persistent over time. All in all, this strategy would help to reduce the distances between the Spanish regions and, as a result, to improve the levels of sustainability of the Spanish economy.

However, this should not be the only strategy to follow, but should be accompanied by others. The results discussed in the preceding sections indicate various key factors for improving the ratios of waste generation. These include public awareness, through the improvement of educational levels, a more rigorous regulation of waste generation in the services sector, and a clear commitment by regional and local administrations to an efficient consumption system.

1.5. Conclusions

Following the recent literature on the economics of waste, we have studied the evolution of municipal solid waste generation in the Spanish regions. To that end, we have focused on two different indicators: the MSW/GDP ratio and the per capita MSW generation. The use of the methodology proposed in Phillips and Sul (2007) leads us to conclude that there has not been a convergence process between the Spanish regions. Rather, we can observe the existence of several patterns of behavior, which implies the existence of a very heterogenous behavior so far as waste prevention is concerned. In this regard, we can see that Madrid, the País Vasco and the regions of the Ebro Valley present the lowest MSW/GDP ratios and that these regions plus Asturias, Castilla-León, Extremadura and Galicia have the lowest per capita MSW. By contrast, the regions situated along the Mediterranean coast and the Islas Canarias exhibit the greatest MSW ratios.

We have employed several socioeconomic variables to explain these different patterns of behavior. The estimation of two ordered probit models leads us to observe that the level of economic activity, the education level of the population, public environmental expenditure, the ideology of the government of the region, the degree of dispersion of the population and, especially, the economic structure are factors that can help us to explain the regional differences in waste prevention. In this regard, we should note that there is a clear relationship between the dependence of the regional economy on the tourism industry and waste generation. This fact must be considered to design strategies aimed at shifting the Spanish economy towards zero-waste, the adoption of unit-pricing schemes being an interesting option for achieving this goal, as analyzed in previous literature.

Finally, we should recognize that a more in-depth investigation into the temporal and regional evolution of MSW is required. Here we have focused on the generation and prevention of waste. However, we should be aware that recycling and energy recovery are two of the priorities of the circular economy in terms of waste management. Consequently, it is also very important to examine how technological progress and the evolution of social demands can help to adapt waste treatment capacity to the needs set by European Union objectives. The study of the long-term relationships between GDP (as a measure of the evolution of a society) and waste generation/recycling would be of great interest in this regard. Knowing the relationship between the production of goods and services and the generation of waste is not only useful for better prediction, but also for evaluating to what degree the production and consumption system is more dependent on the incorporation of materials and the generation of waste. This dependence, studied in the literature as decoupling, has not been evaluated for waste generation and recycling in the EU. This remains for future research.

1.6. Figures and tables

Table 1. Testing for convergence

	MSW/GDP	MSW/population
	Panel I. Testing for convergence	
$\hat{\beta}$	-1.97	-1.04
Log t-ratio	-32.43	-31.44
	Panel II. Estimated clubs	
	Regions	Regions
Club 1	AND, BAL, CAN, CAB, CLM, EXT	CAN, MUR
Club 2	AST, MUR	AND, CAB, CAT, CLM, CVA
Club 3	CAT, CYL, CVA	ARA, AST, CYL, EXT, GAL, MAD, NAV, PAV, LAR
Club 4	ARA, NAV, PAV, LAR	
Divergent	GAL, MAD	BAL

This table reflects the results of the use of the methodology proposed in Phillips and Sul (2007). Panel I presents the estimation of equation (3), whilst Panel II shows the results of the application of the cluster algorithm.

Table 2. Ordered probit estimation.

Table 2. Ordered probit estimation

Variables	estimations	Variables	Estimations
Panel I. MSW/GDP		Panel II. per capita MSW	
pcGDP	0.000853 (2.64)	HOUSEHOLD	-0.00278 (-3.26)
EDU	1.48 (2.11)	GRADUATES	1.16 (3.58)
SERV	- 0.23 (-2.64)	LEFT-WING	0.11 (2.97)
ENVIRON	0.000015 (2.05)	ENVIRON	0.000012 (2.17)
		DISPERSION	15.24 (2.97)

This table reflects the results of the ordered probit estimation of equation (4). Panel I considers the MSW/GDP ratio, whilst Panel B analyzes the per capita waste generation. The dependent variable takes the value i when the region is included in club i, with i=1, 2, …, M and M being 6 for Panel I and 5 for Panel II. The values in parenthesis are the robust t-statistics for testing the null hypothesis whose associated coefficient is 0.

Figure 1. Estimated clubs. MSW/GDP ratio

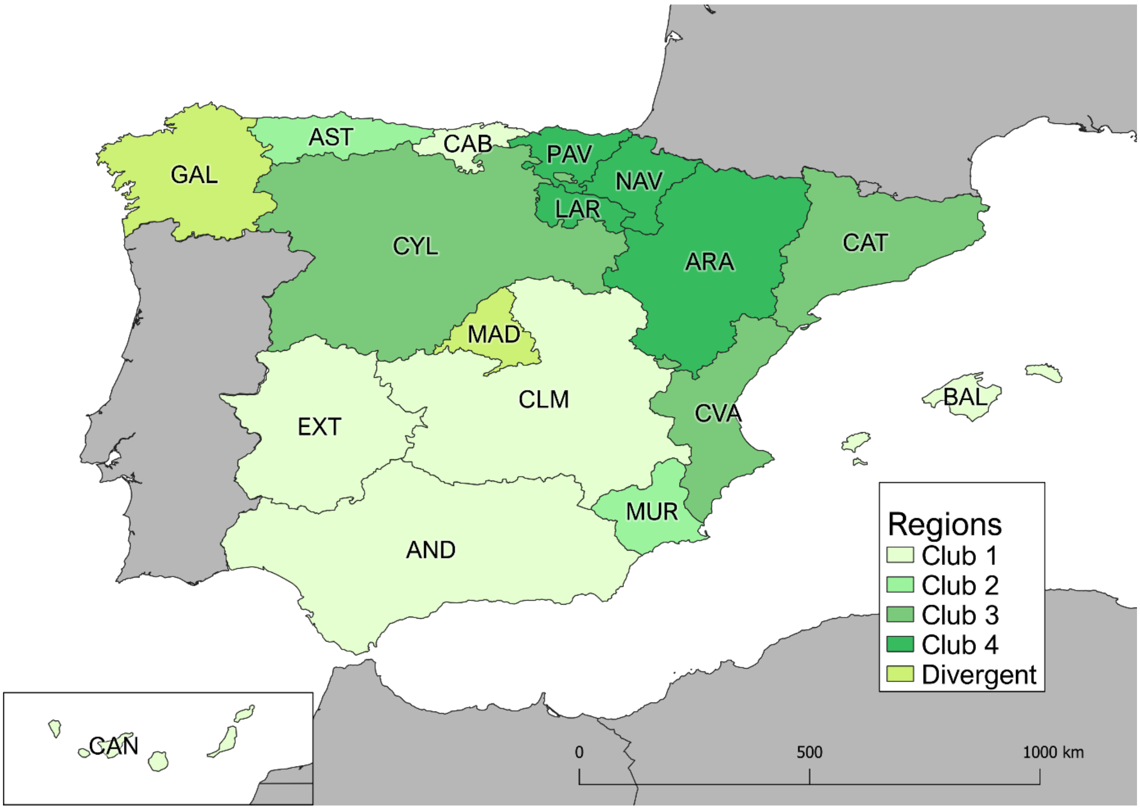


Figure 2. Waste evolution per unit of GDP per club

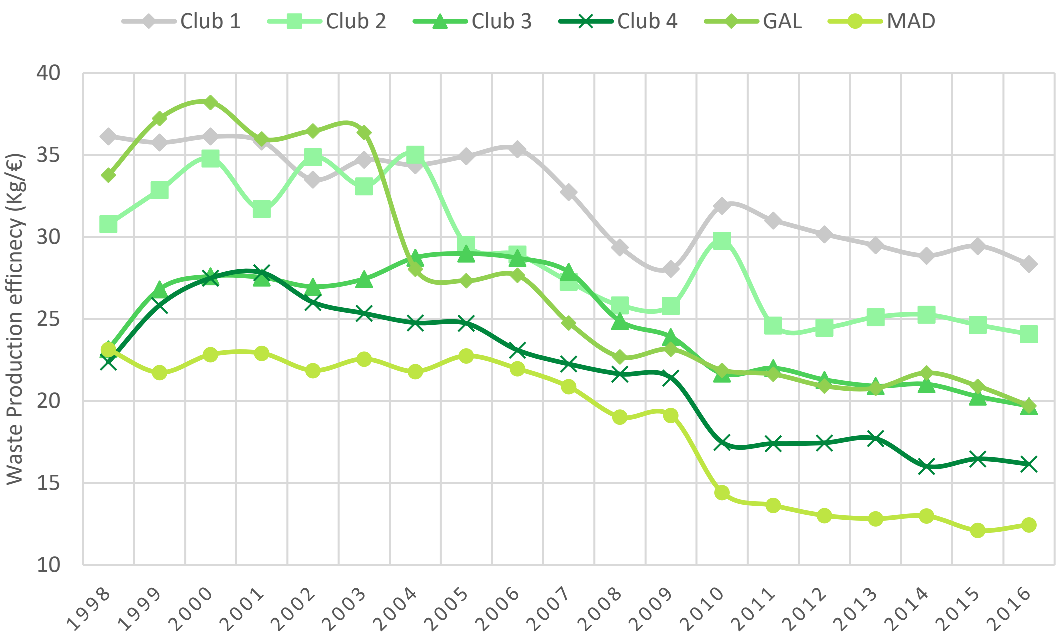


Figure 3. Division by clubs. Waste per capita

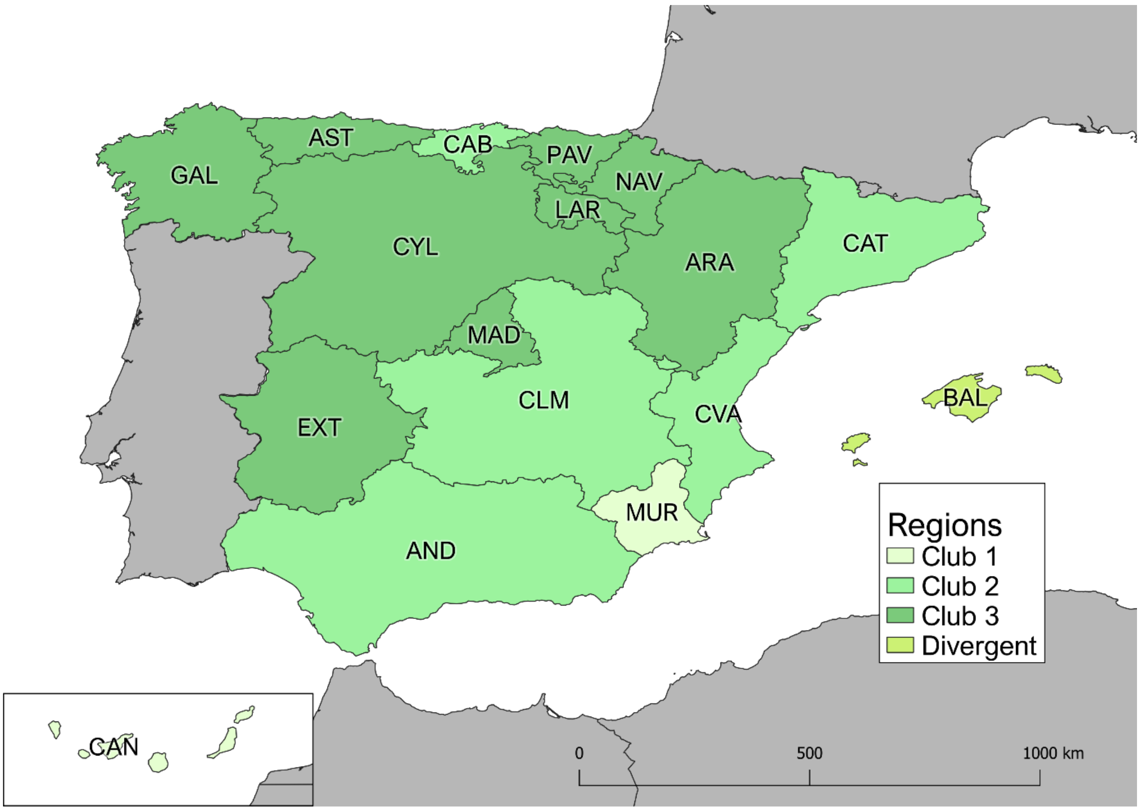
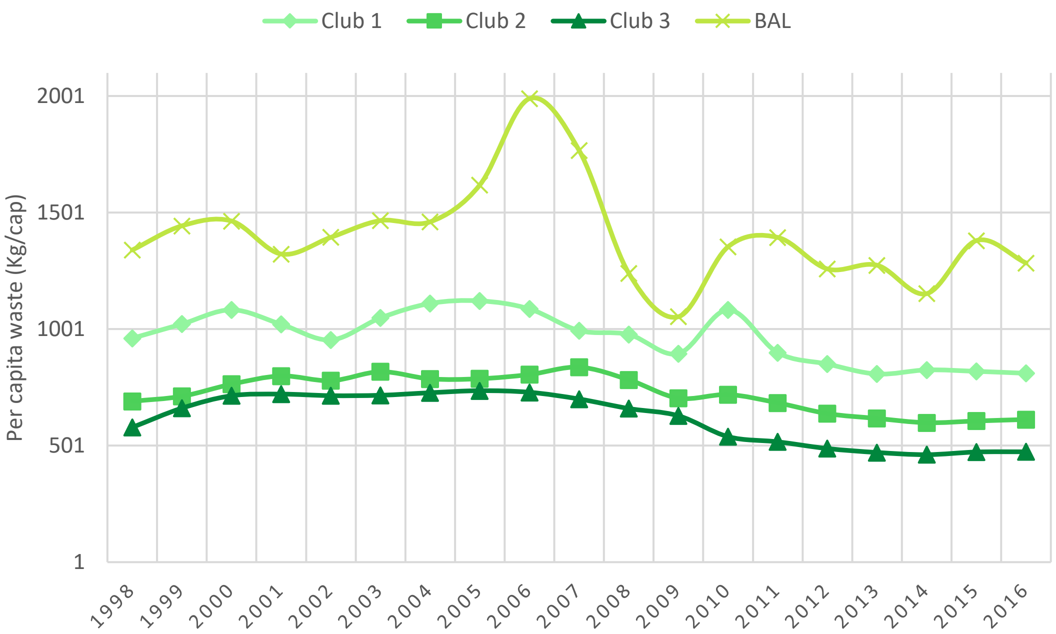


Figure 4. Waste evolution per capita by clubs



Chapter II. A study of the decoupling between waste generation and two economic development indicators for European countries with attention to the Great Recession

2.1. Introduction

The search for the delicate balance between economic growth and sustainability has been one of the core issues on the agendas of politicians and decision-makers in recent years. The underlying idea is that economies should be able to maintain sustained growth over time, but consuming fewer natural resources and avoiding the degradation of the environment. This goal is not always easy to achieve and therefore supranational entities such as the United Nations or the European Union have spared no efforts in this respect, drawing up various recommendations for their member states with the common objective of preserving the environment. Examples are the MDGS (Millennium Development Goals), replaced by the later SDGs (Sustainable Development Goals) of the UN. In particular, the aim of Target 12.5 of the SDGs is to substantially reduce waste generation through prevention, reduction, recycling, and reuse.

The 7th Environmental Action Plan (EAP) introduced by the EU in 2014 follows similar lines. Once again, one of the objectives set within this plan is to reduce the adverse effects of municipal waste on the environment via the promotion of a circular economy, with a special focus on turning waste into a resource, with more prevention, re-use and recycling. This EAP program was followed by the launch in 2020 of the new Circular Economy Action Plan for a Cleaner and More Competitive Europe (CEAP). Section 4.1 is entitled "Enhanced waste policy in support of waste prevention and circularity".

A simple reading of these programs reveals that one of the principal environmental preservation measures is the reduction of waste generated, seeking to decouple environmental degradation and economic growth. This idea of decoupling applies not only to the generation of waste. In general, the programs seek to encourage sustainable economic growth, which is none other than growth which does not generate environmental degradation. Given that this is a crucial factor in all green policy, it is not surprising that the literature analyzing the relationship between economic growth and environmental degradation has grown significantly, mostly focusing on carbon dioxide emissions. For instance, we can cite the papers by Wang and Wang (2019) on the USA case, Zhao *et al.* (2017) on the Chinese case, Shuai *et al.* (2019) on a sample of 133 countries, and Chen *et al.* (2018) on the OECD countries. A summary of these results can be found in Haberl *et al.* (2020).

The growing importance of the waste generation on the environment degradation has subsequently attracted the attention of some researchers, who have analyzed the relationship between waste generation and economic growth. There are many examples of it, since the seminal paper of Johnstone and Labonne (2004) up to the most recent of Gardiner and Hajek (2020b), Mazzarano *et al.* (2021), and Magazzino *et al.* (2021). The conclusions reached by this pleyade of works are far from being robust in the sense that, although the nexus between waste generation and economic growth seems to exist, the debate about the intensity and the direction of this relationship is far from being closed.

A possible explanation of this variety of result may be related to the fact that the relationship between these variables have been estimated under the assumption of parameter stability. However, we should note that this hypothesis may not hold, given that some events can alter it. A very recent example is the so-called the Great Recession, but there exist some other previous examples as the Great Crash (1929), the oil shock (1973), or the dot.com crisis (2004), amongst many others. Whilst the impact of these events on the evolution of the socioeconomic indicators has been frequently analyzed, there are little evidence (if exists) of their effect on the waste generation and on the possible translation to the waste generation/economic growth link. As a consequence, as Namlis and Komilis (2019) suggest, it seems appropriate to reevaluate the waste/economic growth relationship under the prism of the possible presence of changes in their parameters. We should note that estimations may be biased if these breaks are not included in the model specification and, consequently, the conclusions drawn from them could be misguide.

Against this background, the aim of the paper is to study the relationship between waste generation and economic growth in EU countries by considering that the hypothesis of the parameter stability may not hold. The relaxation of this hypothesis can help us to capture the effect of the Great Recession, as well as the one caused by different crisis occurred during the sample considered, on the waste generation. Then, the standard specifications are no longer adequate. Rather, we should employ econometric methods that allow for the presence of structural breaks in the parameters of the model. In our view, the procedure defined in Bai and Perron (1998, 2003a, 2003b) is a very appropriate one, given that it has the advantage of endogenously determining both the number of structural breaks and the period during which these structural breaks appear. Furthermore, we employ two different variables to capture the economic evolution of the countries. On the one hand, we employ the per capita Gross Domestic Product (GDP), which is the most standard measure. On the other hand, we also use the Human Development Index (HDI), also employed in Namlis and Komilis (2019), given that this variable is also correlated with waste generation and provides a different view on the evolution of society, more related to the idea of wellbeing.

The article is organized as follows. Section 2 compiles the main publications that have addressed the analysis of the relationship between waste generation and economic evolution. Section 3 presents the methodology and the data. The main results are reported in Section 4 and discussed and analyzed in Section 5. The main conclusions are given in Section 6.

2.2. Literature Review

This Section presents a brief review of the recent contributions of the literature that try to explain the evolution of the waste generation as a function of the economic growth. In this regard, we should first note that the relationship between environmental degradation and economic development has become an important part of the objectives and policies proposed by international institutions such as the United Nations, the OECD, and the European Union. As a consequence, the interest of analyzing the environmental effects of economic growth and evaluating possible public policies has generated a large literature. The particular case of the waste generation has not escaped to this tendency and we can recently observe a growing interest on the analysis of the relationship between waste generation and economic growth.

The most employed variable to measure the waste generation has been the municipal solid waste (MSW), whose relationship with respect the economic growth (mostly measured by GDP) has been largely addressed in the literature since the seminal paper of Johnstone and Labonne (2004), who study the case of 30 OECD countries. This work was subsequently followed by a good number of similar works, most of them focused on the European countries. This is the case of Namlis and Komilis (2019), Vujić *et al.* (2015), Mazzarano *et al.* (2021), and those of Gardiner and Hajek (2017, 2020a), amongst many others. Similar studies focusing on countries outside Europe include Tao *et al.* (2008) and Gui *et al.* (2019) on the case of China, Jebli and Youssef (2015) on the case of North African countries, and Yılmaz (2020) on the OECD countries. We should alto note the existence of excellent reviews of this literature, as the ones of Gardiner and Hajek (2020b), Boubellouta and Kusch-Brandt (2020), and Magazzino *et al.* (2021).

All these papers use a linear relationship between MSW and different socioeconomic indicators, with the GDP being the most commonly used. However, another group of papers base their study on the use of a non-linear specification, clearly related to the very general proposal of Kuznets (1955), which was particularized to the environmental case in Grossman and Krueger (1995). For example, Mazzanti and Zoboli (2008, 2009), Montevecchi (2016), Ercolano *et al.* (2018), Madden *et al.* (2019), Cheng et al. (2020) examine the so-called Waste Kuznets Curve (WKC). However, the evidence in favor of the WKC is far from being robust. Moreover, we should note that Baalbaki and Marrouch (2020) cannot find evidence if favor of this curve when using a much more general approach, based on the use of the flexible polynomial specification of Wang (2013), which nests the WKC specification. This result seriously queries the existence of this WKC, as Aslanidis and Iranzo (2009) also note for the general environmental Kuznets Curve.

Then, the analysis of this literature leads us to conclude that the link between waste generation and economic growth still is an open debate matter. A possible reason for this lack of unanimity in the conclusions may lie in the fact that the relationships between waste generation and economic growth have been estimated under the assumption of stability of the parameters. However, this assumption is somewhat dubious, given that some events (such as the already mentioned Great Recession) may have affected to the waste generation/economic growth relationship. Then, a possible way to reconcile this amalgam of very different results is the relaxation of the hypothesis of parameter stability, allowing for the parameters of the model to change. In this regard, we should note that the presence of structural breaks has not been considered in this type of literature, ignoring the fact that the flexibility provided by the inclusion of these breaks may help us to better understand the nature of the waste generation-economic growth nexus.

Having said that, next Sections are devoted to the analysis of the relationship between waste generation and economic growth under the scope offered by the presence of structural breaks.

2.3. Data and methods

2.3.1. Data Source

As mentioned, the aim of the paper is to analyze the relationship between the generation of waste and the evolution of the EU economies. The data employed to that end are data of per capita municipal solid waste (MSW) as a measure of waste generation. This variable also provides us with a useful approximation to environmental degradation, in that it reflects not only the consumption patterns of a country's population, but also the environmental awareness, and the adoption of industrial environmental practices by companies when designing products and packaging.

In order to measure the evolution of the economies, we employ the per capita GDP, which is the most standard variable employed to that end. However, we also use the HDI. The inclusion of this variable, in line with previous works (Namlis and Komilis, 2019; Sanyé-Mengual *et al.*, 2019; Kalimeris *et al.*, 2020), is to strengthen the welfare dimension provided by GDP with a broader vision of development covering elements such as education or the health of societies.

The MSW and GDP data have been obtained from the Eurostat database (Eurostat, 2019), whilst the data of the HDI have been collected from the World Development Indicators database. European Union countries were selected for which we had substantial information during the period 1995-2018. The final sample is composed of the following countries: Austria, Belgium, Bulgaria, Cyprus, Czechia, Denmark, Estonia, Finland, France, Germany, Greece, Hungary, Ireland, Italy, Latvia, Lithuania, Luxembourg, Netherlands, Poland, Portugal, Romania, Slovakia, Slovenia, Spain, Sweden, and United Kingdom. We have additionally considered the data of the total EU27 when available (MSW and GDP).

Table A1 of the Appendix 2 reflects some descriptive statistics of the variables. As we can observe, the per capita MSW of the EU27 was 492 kilograms in 2018, somewhat greater than the value at the beginning of the sample (465 kilograms). Then, we can observe a slight growth. This increment is not homogeneous, given that we can observe that Bulgaria (-2.1%), Germany (-0.1%), Hungary (-0.8%), Netherlands (-0.2%) Romania (-1.0%), Slovenia (-0.9%) and Spain (-0.3%) show negative growth rates, whilst Austria (1.2%), Denmark (1.7%), Finland (1.3%), Greece (2.3%), Latvia (1.9%), Portugal (1.6%) and Slovakia (1.5%) show growth above 1%. This heterogeneous behavior is maintained if we split the sample into two subperiods, 1995-2007 and 2007-2018, in order to analyze the possible effect of the GR. We can now observe that the generation of per capita MSW in the EU27 increased by 0.9% in the period before the GR, while it decreased at a rate of -0.5% after it. But even this result is not common to all countries, since there are countries that decreased their waste generation before the GR, mainly Bulgaria (-1.9%), Slovenia (-1%) and Germany (-0.6%), while Austria (2.6%), Denmark (3.5%), Latvia (3.3%), Portugal (2.5%) and Sweden (2%) increased their waste by above 2%. After the GR, most countries decreased their waste, although some increased it substantially, as is the case for Czechia (1.6%) and Slovakia (3.2%). The increase in the generation of waste generally occurs in the recovery period 2014-2018. Therefore, the results are far from being homogeneous, even despite the great effort that has been made by the EU Commission for harmonization and coordination in the fight against waste generation.

This heterogeneity also appears when considering the economic growth of the countries. The per capita GDP of the EU27 as a whole grew at an average of 1.5% per year during the period 1995-2018. However, there are countries that grew much faster, such as Ireland (4.1%), Poland (4.1%), Estonia (4.6%), Latvia (5.1%)

and Lithuania (5.4%), while other countries like Italy (0.3%) or Greece (0.7%) did not reach 1%. Likewise, growth before the GR was 2.2% in the EU27 as a whole, while this figure was much more moderate (0.7%) after the GR and was even negative for countries such as Greece (-2.6%), Italy (-0.7%), Cyprus (-0.6%), Finland (-0.1%) and Luxembourg (-0.1%). Moreover, we can appreciate notable differences between the 2007-2014 and the 2014-2018 periods. We also observe that all countries experienced positive growth above 1% in the period 2014-2018.

The HDI data are much more homogeneous, partially due to the bounded construction of this indicator, which takes values in the (0,1) interval. However, despite this, they do show slight variations both by country and by period. The average HDI of the EU27 countries included in the sample grew at an average rate of 0.6% throughout the sample period, although the growth was 0.8% before the GR and only 0.4% afterwards. Over the entire period, the growth in some countries was very meagre. Belgium, France and the Netherlands had an average growth rate of 0.3%, while Estonia, Latvia and Lithuania were around 1.0%. Likewise, the growth rate after GR was clearly lower, especially in countries such as Belgium, Finland, France, Germany, Italy and Luxembourg with increases of 0.2%.

The previous descriptive analysis sustains our initial idea of the possible presence of structural breaks related to GR in the relationship between per capita MSW, HDI and the evolution of the economy. The following section analyzes this issue more deeply using more powerful econometric methods.

2.3.2. Testing for structural breaks

Following the seminal works of Grossman and Kreuger (1995) and Holtz-Eakin and Selden (1995), our starting model is the linear relationship between MSW and GDP:

$$\ln(MSW_{it}) = \alpha_i + \beta_i \ln(GDP_{it}) + e_{it}, \tag{1}$$

With: i=1, …, 31, t= 1995, …, 2018

As Grossman and Krueger (1995) note, this model is a reduced form that has the advantage of summarizing the net effect between these two variables. It has also some limitations, such as the absence of information of why this relationship exists. In any event, this is a quite standard specification, which is commonly employed in the literature.

This model considers that the parameters of the model cannot change over time. This is a quite questionable restriction, especially if we take into account some events, such as the GR, that could have affected this relationship by modifying the value of the GDP elasticity. Therefore, it seems necessary to adapt the previous model to the presence of structural breaks.

We can employ several econometric tools to that end, but we consider that the methodology developed by Bai and Perron (1998, 2003a, 2003b) is the most suitable given its flexibility and good performance even with samples like the one we have in this paper. This methodology has the advantage of endogenously determining the number of breaks, as well as the period when these breaks occur. This is based on the estimation of the following model:

$$\ln (MSW_{it}) = \alpha_{ij} + \beta_{ij} \ln (GDP_{it}) + v_{it}, \qquad (2)$$

With: i=1, 2, …, 31, t= TB_{j-1}, …, TB_j, j=1, …, m+1

where TBj means the period where the breaks appear, with TBo = 1995 and TBm+1 = 2018, m being the number of breaks, and v an innovation that can follow a wide range of stationary models, including the general ARMA model. We should note that the variance of this innovation need not be constant and, therefore, breaks in variance are considered provided they occur at the same dates as the breaks in the parameters of the regression.

The Bai–Perron (BP) procedure involves the estimation of the above equation, considering that the break may appear at any point in the sample. A Chow-type test is then defined in order to determine the existence of the first break. The estimation of the period where this first break occurs coincides with the period where the Chow-type statistic attains its maximum value. The presence of multiple breaks can be analyzed by using the UDmax and WDmax statistics which test the null hypothesis of no structural breaks versus the presence of an unknown number of breaks. The number of breaks has been estimated by considering a maximum value of 3 breaks and subsequently applying the sequential procedure defined in Bai and Perron (1998), combined with the repartition method described in Bai (1997). In those cases where the UDmax and WDmax reject the non-structural break null hypothesis but the sequential method cannot find any break, we have determined the number of breaks by using the statistics proposed by Schwarz (1978). Finally, we have used the quadratic spectral kernel to take into account the presence of possible autocorrelation and heterogeneity in the perturbations, combined with the Andrews (1991) automatic bandwidth selection with an AR (1) approximation.

Given that the Bai–Perron procedure only works correctly once regime-wise stationarity is proved, we are limited to applying it to those cases where the unit root null hypothesis has been previously rejected. Thus, an appropriate strategy should be based, first, on the application of the unit root tests and, once stationarity is shown, we should then apply the BP procedure for estimating the number of breaks, the periods where the breaks appear and, finally, the mean of the variable of each of the regimes.

We should note that some papers have previously analyzed the relationship considered in (1) by employing the so-called Waste Kuznets Curve, as we mentioned earlier. Then, we have also adapted the WKC model to admit the presence of structural breaks and we have additionally estimated the following equation:

$$MSW_{it} = \alpha_i + \beta_i \, GDP_{it} + \gamma_i \, GDP^2_{it} + e_{it}, \quad (3)$$

With: i=1, …, 31, t= 1995, …, 2018

A comparison shows that the results obtained from the estimation of (2) clearly outperform those of model (3) when, for instance, we compare both estimations by using the information criterion proposed in Schwarz (1978), even if we admit the presence of structural breaks in (3). As a consequence, the results of the estimation of the WKC models will be omitted and we will focus exclusively on the estimation of model (2).

Finally, as previously mentioned, we will also consider the relationship between the waste generation and the HDI, by simply substituting HDI for GDP in equation (2).

2.4. Results

2.4.1. Unit root inference

As a previous step to using the BP methodology, we have mentioned that we should first analyze the time properties of the variables included in equation (2). If we can reject the unit root null hypothesis, then we will be able to apply this methodology. The unit root inference has been based on a specification that includes an intercept and a trend. Additionally, we have used the quasi–generalized least squares detrending method proposed by Elliot *et al.* (1996), instead of using the standard statistics proposed in Dickey and Fuller (1979) which are based on the ordinary least squares estimation. Furthermore, we have considered the possible presence of several breaks in the trend function, in order to avoid the bias caused by ignoring them; see Perron (1989) in this regard. Then, we have employed the statistics proposed by Carrion-i-Silvestre *et al.* (2009), considering a maximum value of 3 breaks. Examples of the use of these statistics for environmental variables can be found in Cai *et al.* (2018), Yilanci *et al.* (2019) and Churchill *et al.* (2020). The results are presented in Tables A2, A3 and A4 of the Appendix 2.

We first observe that the evidence against the unit root null hypothesis is scarce when the presence of broken trends is not considered. However, the inclusion of breaks in the trend function changes this picture and the evidence against this hypothesis is robust for the three variables under consideration. Nevertheless, there are some exceptions. We cannot reject the presence of a unit root for the per capita MSW in the case of Italy. However, if we exclude the last observation, we find robust evidence against it and, therefore, we will maintain this country in the analysis. The absence of evidence against the unit root hypothesis for the GDP of Hungary and Romania is more problematic, even if we consider a liberal 10% significance level. Similarly, we have not been able to reject the unit root null hypothesis for the HDI of Finland. Although we should omit the results of these countries, we will maintain them to facilitate the comparison of their results with those of the rest of the countries.

We can also see that the breaks in the trend function can be grouped around three periods of time. The first appears around the year 2000 and is related to the burst of the dot-com bubble, also reflecting the introduction of new environmental policies aimed at waste prevention (European Parliament and Council Directive 1994/62/EC on packaging and packaging waste and the Council Directive 1999/31/EC on the landfill of waste). The other two breaks are connected (2008 and 2013) to the effects of the Great Recession, the fall in GDP worldwide and its later recovery. These results confirm our suspicion about the importance of the Great Recession in the evolution of the per capita MSW and, therefore, it seems advisable to analyze whether these breaks also affect the determinants of the MSW, which is the goal of the next section.

Once we have proved that the variables are not integrated, we can then apply the BP procedure to test for the presence of structural breaks in the relationship between waste generation and the two measures of the evolution of the economies that we have selected.

2.4.2. MSW and GDP relationship: Is it stable?

Tables 2 and 3 present the results of the application of the BP methodology to the GDP-MSW and HDI-MSW relationships, respectively. This Table includes information on some statistics that analyze the null hypothesis of parameter stability, the estimations of the parameters and their corresponding robust standard deviations, the periods where the breaks occur and some statistics for analyzing the goodness of the estimation. We should first note that both UDmax and WDmax statistics always reject the null hypothesis of non-structural breaks. Therefore, none of these relationships is stable across the sample. The number and the periods when the structural breaks appear vary across the countries. However, we can observe that they are again concentrated in three periods of time, coinciding with those obtained from the unit root inference. Parameter βj corresponds to the elasticity in each estimated sub-period (j=1,2,3,4). Figure 1 presents maps with the estimated elasticities in 1995, 2007 and 2018. We classify these elasticities into three groups: Absolute decoupling ($\beta_j \leq 0$), Relative decoupling ($0 < \beta_j \leq 1$) and coupling ($\beta_j > 1$).

If we analyze the estimated elasticities in Table 2 at the beginning of the sample, we can observe the range goes from -0.94 (Slovakia) to 2.28 (Greece), whilst the value for the total EU27 is 0.56. We can also appreciate that Slovakia, Slovenia, Bulgaria, and Germany show absolute decoupling, whilst Spain, Czechia, Sweden, Portugal, Denmark, Austria and Greece present elasticities greater than 1. The estimated elasticities at the end of the sample are somewhat different. The range now goes from -0.89 (Belgium) to 2.67 (Slovakia). We can observe that Belgium, Luxembourg, United Kingdom, Netherlands, Bulgaria, Sweden, and

Denmark present absolute decoupling of GDP elasticity, whilst Portugal, Poland, Slovenia, Greece, and Slovakia show a coupling relationship in the final estimated segment of the sample.

This initial analysis of the estimated elasticities denotes a clear heterogeneity in the results. In spite of this, it is true that the elasticity for the total EU27 does not show substantial changes, always taking positive values and pivoting at around 0.5.

If we compare the estimations of elasticities before and after the Great Recession, we can observe that some countries have clearly increased them. This is the case of Germany (-1.01, 0.46), Poland (-0.12, 1.13), Slovakia (0.37, 2.67), Spain (-0.50 , 0.55) and the United Kingdom (-3.49, -0.58). By contrast, Denmark (1.47, -0.07), the Netherlands (0.24, -0.56) and Sweden (0.66, -0.12) show a significant improvement during the recession.

Focusing on the decoupling process between GDP and MSW for European countries, the results shown in Table 2 reflect an improvement during the whole period in most countries. Bulgaria is the only country in which absolute decoupling is maintained, while Portugal is the only one that maintains a near coupling relationship. The decoupling process intensified during the 2000s for several countries, although it was brought to a halt by the economic crisis which, except for some countries that maintained the downward trend, resulted in a reduction of waste generation with respect to GDP. Sweden, Denmark, the United Kingdom and Luxembourg exhibit a more pronounced improvement. By contrast, Estonia, Slovakia and Slovenia experienced more intensive waste generation in terms of GDP. We can also observe that most countries exhibit relative decoupling. The EU27 average slightly improved over this period, showing a situation of relative decoupling.

Table 3 shows the results when considering the HDI as an indicator of wellbeing in EU countries. The values of the estimated elasticities are larger (in absolute terms) than those obtained for the MSW-GDP relationship. This can be easily understood if we take into account the fact that the HDI varied slightly, mainly due to its construction. Then, it comes as no surprise that the estimated elasticities may take somewhat large values (in absolute terms). The range of HDI elasticities at the initial estimated segment of the sample goes from -9.32 (Slovakia) to 8.82 (Austria). Likewise, Slovakia, Slovenia, Bulgaria, Belgium, and Germany exhibit absolute decoupling, the estimated values of Latvia and Estonia are small, and the rest of the countries exhibit estimated HDI elasticities greater than 1. The results are somewhat different if we consider the elasticities of the final estimated segment. The range goes from -6.56 (Belgium) to 24.44 (Slovakia). We can also see that Austria, Belgium, Bulgaria, Denmark, Germany, Greece, Hungary, Ireland, Luxembourg, Netherlands, Romania, Sweden, and United Kingdom exhibit negative estimated elasticities, whilst Czechia, Estonia, France, Lithuania, Poland, Portugal, Slovakia, Slovenia, and Spain show a strongly coupled relationship.

The GR also represents a clear disruption in the relationship between MSW and HDI, if we compare the estimated HDI elasticities before and after the GR. The results in Table 3 and Figure 1 allow us to see that this elasticity has clearly decreased in some countries, with Belgium, Denmark, Germany, Greece, Ireland, the Netherlands and Sweden showing the greatest progress towards decoupling after the GR. By contrast, Estonia, Italy, Poland, Portugal and Spain worsened the most.

There was also an overall reduction in the HDI-MSW estimated elasticities, with Austria, Belgium, Greece, the Netherlands, and the United Kingdom showing the greatest improvement. The only countries that deteriorated are Czechia, Estonia, France, Latvia, Lithuania, Poland, Portugal, Slovakia and Slovenia, increasing their estimated elasticities. The remaining maintained a stable or slightly improved relationship. Therefore, we can observe that not only production but, more broadly, the wellbeing of countries may be becoming decoupled from the generation of waste.

This indicates that while countries that have experienced an increase in decoupling in production have improved or maintained decoupling in their development, countries that have undergone a deterioration in decoupling in production have also maintained or worsened decoupling in their development.

Finally, our results allow us to note that the decoupling between MSW and HDI is less evident than that observed between MSW and GDP, which has been the general trend in European countries during this period.

2.5. Discussion

Our results offer three very interesting insights. First, we observe that the link between waste generation and economic development is heterogeneous across EU countries. Then, the noticeable differences between the GDP elasticities questions the use of the homogeneous panel data approach, as well as reveal the inexistence of an effective shared policy to achieve convergence. Secondly, we can also see that this relationship is quite sensitive to the economic cycle, presenting several breaks. This supports the argument that environmental policy is not a central issue in periods of recession. Additionally, the presence of these breaks confirms the need of relaxing the parameter stability hypothesis in the waste generation/economic growth relationship. Finally, we also see that there is a movement towards relative decoupling, although it is still incipient progress that only applies to some EU members.

We also observe that our results offer new evidence about the heterogeneous behavior of EU countries in their transition towards greener economies. In spite of the significant efforts made in order to reduce the differences and facilitate convergence, as Castillo-Giménez *et al.* (2019a) show, such differences are still significant, as can also be seen in Castillo-Giménez *et al.* (2019b) and in Minelgaitė and Liobikienė (2019). Even worse, we can also observe that the GR has intensified these behavioral differences between European countries, increasing the existing heterogeneity in waste generation.

In this regard, we should note that the recurrence in these environmental policy disparities has led some researchers to divide the EU countries into "leaders, midfielders and laggards" according to the level of implementation of these policies. If we analyze the MSW-GDP elasticities at the end of the sample, we can observe that the countries with negative elasticities are those commonly considered leaders in the literature (Belgium, Denmark, Luxembourg, Netherlands, Sweden and United Kingdom), as Knill et al. (2012) and Melidis and Russel (2020) point out, with the noticeable inclusion of Bulgaria. These countries are distinguished by their ambitious waste prevention plans (European Environment Agency, 2020) with detailed proposals for each sector of the economy and a group of indicators and quantitative objectives that are periodically monitored for compliance. In addition, they have achieved establishing selective collection and recycling as a core element of waste management, severely restricting other alternatives such as dumping or incineration (European Environment Information and Observation Network, 2020). Also significant are the initiatives to raise awareness of environmental issues and the efforts to coordinate the recycled goods market by creating business synergies. By contrast, Greece, Poland, Portugal, Slovakia and Slovenia exhibit elasticities greater than 1, with these countries commonly being classified in the laggard group. This group of countries is defined by a waste treatment sector that is incipient, with limited recycling or selective collection, leaving the largest fraction of urban waste to be landfilled. Furthermore, waste prevention plans may be conditioned in some cases by the need to maintain economic growth in order to converge with the rest of Europe, at the cost of assuming waste increase. The plans of these countries also tend to be less ambitious and lack evaluation or measurable quantitative targets. The remaining countries, the midfielders, are in a mixed state, with relatively ambitious plans and with a waste treatment system shifting from landfill to recycling or incineration, aiming to achieve the targets imposed by the EU. A similar classification of countries is obtained by Ríos and Picazo-Tadeo (2021) who rank a group of countries according to their waste treatment desirability.

If we consider the HDI as a driver of the MSW, the qualitative conclusions can be maintained by providing a similar view of the evolution of the countries. The fact that the results for the HDI are higher in absolute value and the decoupling may be more challenging to achieve could be due to two factors. On the one hand, as noted above, the bounded construction of the index. On the other hand, following the reasoning of Kalimeris *et al.* (2020), the HDI increase may require a larger material base than GDP because it has to sustain the growth of life expectancy and education.

The picture that emerges from this analysis is quite clear and we can see that the economies with the lowest per capita GDP and HDI levels do not exhibit decoupling. A possible explanation for this result lies in the fact that these countries adopted policies focused on favoring convergence with respect to the rest of the EU countries. As a consequence, environmental policies were considered secondary at this time and therefore postponed, as is shown in Burns *et al.* (2020). In this regard, we should note that these convergence policies favored the creation of employment and the inherent increase in consumption levels, this being a key factor for understanding the evolution of waste generation, as Khajevand and Tehrani (2019) and Yilmaz (2020) note. By contrast, the most developed countries have had more possibilities to introduce environmental policies, such as recycling or environmental awareness programs, that have proven very effective in reducing urban waste, as can be deduced from Cecere *et al.* (2014), Gilli *et al.* (2018) and Cole *et al.* (2014).

The effect of the GR offers a very clear example of this dual situation. We can see that this economic crisis has significantly altered the relationship between waste and GDP/HDI, as discussed in the results section. Its impact has not been homogenous, being more pronounced in the so-called midfielders and laggards' countries. This can be easily understood if we consider that these countries gave priority to policies of convergence and budgetary stability over environmental policies after the GR. These austerity policies noticeably slowed down the development of necessary environmental regulations and, even worse, led to a loss of ambition in meeting environmental objectives, as Burns *et al.* (2020) and Burns and Tobin (2020) point out.

Another challenge for public policy is to achieve absolute decoupling considering the heterogeneous nature of Europe. In this regard, our estimated elasticities at the end of the sample mostly show a relative decoupling between waste and GDP and a more modest decoupling between waste and HDI. On this basis, it cannot be firmly concluded that the path towards absolute decoupling, if it is possible, will occur with the traditional development policy mix. According to our results, the convergence policies implemented to date have been unsuccessful in combining economic and environmental development. In this regard, we should note that Gardiner and Hayek (2020b) also find evidence of the insufficiency of growth policies to reduce waste generation, especially for lower income countries within the EU. Under these circumstances, the European Union should promote the introduction of new and more powerful environmental policies to reduce waste generation.

Moreover, we consider that waste policies should not only cover the management side, such as promoting recycling, but also amplify their scope by considering other essential factors, mainly consumption patterns, product design and waste/environmental education, as is mentioned in Abbott *et al.* (2013), Cecere *et al.* (2014), D'Amato *et al.* (2016) and Gilli *et al*. (2018). This policy mix perspective would integrate not only regulations on recycling ratios or material use bans, but a more proactive contribution by generating synergies between companies to move towards industrial ecology, raising awareness among people less inclined to pursue environmental policies and strengthening the role of economic measures such as environmental taxes. The environmental policy design would require a more holistic approach in which involving personal motivations may play a crucial role, given the positive effect generated by the desire for social approval which leads people to make visible the fact that they are complying with environmental policies, as Bucciol *et al*. (2019) note. As a corollary, the modest progress made towards absolute decoupling suggests the necessity of introducing further European policies to establish a genuine green economy.

2.6. Conclusions

This work has analyzed the relationship between environmental degradation and economic developments by studying the link between MSW and two socioeconomic indicators: the standard per capita GDP and the HDI. Our results confirm the existence of a clear connection between them, but we have also proved the presence of structural breaks in this relationship. This result demonstrates that waste generation has been quite sensitive to economic shocks such as those resulting from the dot.com crisis (around 2000) and, especially, the GR. Therefore, these inclusion of these events in the model specification has revealed very helpful to improve the quality of the model estimations and, as a consequence, better understand them.

The presence of these breaks helps us to appropriately estimate the effect of economic developments on waste generation. Once these structural breaks are accounted for, we can observe that there is relative decoupling between both

MSW-GDP and, to a lesser extent, MSW-HDI. We can also see that the Great Recession constituted a severe setback that slowed down much of the progress made until 2007 so far as waste prevention is concerned. In particular, the recovery from the GR (2014-2018) involved an increase in waste generation, especially in those countries with the lowest per capita GDP values.

Our results also offer evidence of the heterogeneity of the environmental behavior of EU countries. The GR even increased the polarization between countries that already had a decoupling relationship before the crisis and maintained it (Denmark, Sweden, and the United Kingdom in the top positions) and those that were in a more modest situation which became worse (Slovakia and Slovenia in the bottom positions).

Consequently, our results show that some countries have achieved the goal of decoupling waste generation and economic growth, but this process is still at a very incipient stage if we analyze the EU as a whole. This suggests that there is still a need to introduce policies at the European level to homogenize results and set more ambitious goals to prevent and reduce waste generation in accordance with international treaties and commitments made in both the SDGs and the CEAP.

Finally, as we have mentioned previously, the results are conditioned by the length of the sample. The availability of larger time series of the MSW variables would be recommendable. Then, it seems sensible to carry out new studies once new data are available. In particular, it could be of great interest to relax the restriction that the breaks in the variance are located at the same time as the breaks in the parameter regression, a question that is left for future research.

2.7. Notes

[1] We should also cite the seminal paper of Cole *et al*. (1997) who study the Kuznets curve for several environmental degradation measures, including waste generation.

[2] We should note that this statement not only concerns to waste generation but can also be extended to most measures of environment degradation, as Vadén *et al.* (2020) and Haberl *et al.* (2020) note.

[3] Given that the MSW data for 2018 are not available for Cyprus, Greece, and Ireland, the sample covers 1995-2017 for these countries.

[4] MSW data for Ireland are missing for 2013 and 2015 and, therefore, we have linearly interpolated them.

[5] Casini and Perron (2019) provide an excellent review of the recent advances in structural breaks in time series.

[6] We should note that we could relax this restriction and consider the presence of breaks in the variance at different periods than those of the parameter regression. Perron *et al.* (2020) propose a statistic to analyze this point, based on the procedure defined in Qu and Perron (2008). However, the scarce data availability warns against the use of these statistics. Consequently, we prefer to focus on the analysis of changes in the parameters of the regression and leave the case of changes in the variance for future research, once more observations have been added to the sample.

[7] Additionally, we should take into account that the lack of evidence against the unit root null hypothesis may be related to the relatively short length of the sample.

[8] The values in parentheses represent the elasticities for the estimated segments before and afterthe GR, respectively.

2.8. Figures and tables

Table 1. Relevant recent studies on the analysis of the relationship between economic development and waste generation

Study	*Methodology*	*Region*	*Period*	*Socioeconomic Indicator*	*Impact Indicator*	*Conclusions*
Gardiner and Hajek (2020b)	Panel vector error correction model	284 European regions (NUTS-2)	2000-2018	GDP, R&D, GFC, EMP	MSW, HE	GDP and MSW are mutually reinforcing. GDP causes R&D, which slows down MSW.
Madden et al. (2019)	Geographically and temporally weighted regression. WKC	128 Municipalities in Australia	2011-2015	INC, DENS	MSW	Mixed evidence on WKC. Presence of relative decoupling. Dependence on regional socioeconomic factors.
Namlis and Komilis (2019)	Statistical analysis. Principal component analysis	10 EU countries	2008-2015	GDP, HDI, UR	Different MSW streams, CE	MSW streams are strongly correlated with GDP and HDI

Notes: DENS -Population density, DMC – Domestic Material Consumption, EMP – Employment, GDP – Gross Domestic Product, GFC – Gross fixed capital formation, GPI – Genuine Progress Indicator, HDI- Human Development Index, HC – Final household consumption, HE – Heating energy, INC – Mean Income, INCI -Incinerated waste, ISEW – Index of Sustainable Economic Welfare, LAND – Landfilled waste, LCA – Life Cycle Analysis, MSW – Municipal solid waste, REC – Recycled waste, R&D – Research and development, UR – Unemployment rate, WKC – Waste Kuznets Curve.

Table 2. Testing for breaks and estimation of the equation (GDP and MSW)

	UD_{Max}	WD_{max}	α_1	β_1	TB_1	α_2	β_2	TB_2	α_3	β_3	TB_3	α_4	β_4
EU27	334	515	-6.3	0.56	2002	-4.68	0.40	2009	-6.78	0.60	2013	-4.55	0.38
			0.78	0.08		0.25	0.02		5.46	0.54		0.34	0.03
Austria	52	64	-16	1.49	2003	-6.01	0.52	2009	-4.38	0.36			
			3.3	0.32		0.68	0.06		0.5	0.05			
Belgium	432	432	-4.32	0.35	2008	15.54	-1.57	2012	8.47	-0.89			
			0.34	0.03		4.32	0.41		2.25	0.21			
Bulgaria	106	129	0.71	-0.15	2011	1.66	-0.29						
			0.6	0.07		1.42	0.16						
Cyprus(b)	59	84	-3.2	0.28	2000	-5.3	0.49	2007	-9.46	0.9	2013	-3.52	0.31
			0.62	0.06		0.38	0.04		0.6	0.06		1.27	0.13
Czechia	172	257	-12.58	1.23	2000	-4.32	0.32	2008	-7.6	0.67			
			2.16	0.1		0.31	0.03		1.1	0.11			
Denmark	110	143	-16.05	1.47	2006	0.57	-0.07						
			1.8	0.17		2.69	0.25						
Estonia	333	407	-1.63	0.08	2008	7.25	-0.9	2013	-9.56	0.9			
			0.41	0.04		2.5	0.27		0.33	0.03			
Finland	113	125	-10.09	0.91	2000	-8.84	0.78						
			0.61	0.06		1.35	0.13						
France	94	144	-6.21	0.54	2011	-3.00	0.23						
			0.43	0.04		1.46	0.14						
Germany	77	119	-0.21	-0.02	2002	9.87	-1.01	2006	-5.25	0.46			
			1.73	0.17		3.83	0.37		1.02	0.1			
Greece(b)	515	629	-23.11	2.28	1998	-5.25	0.45	2009	-5.53	0.49	2014	-14.85	1.45
			3.67	0.38		0.46	0.05		0.39	0.04		2.64	0.27
Ireland(b)	443	680	-4.66	0.4	2000	-6.59	0.6	2007	-19.6	1.82	2013	-0.4	0.01
			0.13	0.01		1.15	0.11		2.65	0.25		0.37	0.03
Italy	56	86	-9.8	0.89	1998	-9.26	0.84	2003	-5.23	0.45	2011	-5.47	0.47
			0.8	0.08		1.14	0.11		2.03	0.2		2.22	0.22
Latvia	314	384	-4.35	0.36	2014	-1.5	0.06						
			0.38	0.04		0.68	0.07						
Lithuania	106	129	-3.07	0.26	1998	-7.3	0.73	2002	-3.85	0.32			
			1.6	0.19		0.35	0.04		0.33	0.04			
Luxembourg	365	365	-7.52	0.64	1999	-4.7	0.38	2012	7.03	-0.66			
			0.37	0.03		0.77	0.07		2.46	0.22			
Netherlands	337	519	-3	0.24	2011	5.27	-0.56						
			0.87	0.08		1.84	0.17						
Poland(a)	64	98	-4.65	0.4	2000	8.82	-1.15	2004	-0.08	-0.12	2013	-11.79	1.13
			0.72	0.08		2.39	0.27		0.75	0.08		1.10	0.12
Portugal	102	125	-14.7	1.44	2000	-10.93	1.04	2007	-9.35	0.89	2011	-10.78	1.03
			0.43	0.05		1.21	0.12		9.85	1.01		0.61	0.06
Slovakia	421	515	7.02	-0.94	2001	-4.71	0.37	2014	-26.64	2.7			
			0.99	0.11		0.62	0.06		0.73	0.08			
Slovenia	240	369	7.69	-0.87	2001	-11.16	1.07	2010	-13.98	1.34			
			1.83	0.19		1.62	0.17		3.22	0.33			
Spain	627	966	-11.02	1.06	2003	-1.00	0.04	2007	-19.26	1.85	2013	-6.35	0.55
			0.98	0.1		4.25	0.42		2.61	0.26		0.23	0.02
Sweden	129	139	-15.62	1.43	1998	-7.66	0.66	2009	0.49	-0.12			
			1.02	0.1		0.87	0.08		1.86	0.17			
United Kingdom	597	730	-10.15	0.94	2002	8.21	-0.85	2008	35.26	-3.49	2012	5.27	-0.58
			0.3	0.03		3.64	0.35		3.56	0.34		2.57	0.25

This table presents the results of the estimation of model (2), with TBj (j=1,2,3) being the estimated periods when the break appears. The number of breaks has been selected by using the sequential procedure described in Bai and Perron (1998). UDmax and WDmax test the no structural break null hypothesis, which is rejected in all the reported cases when using the appropriate critical values. Robust standard deviations are presented below the estimated parameters.

(a)Structural breaks selected by BIC

(b)Last observation 2017

Table 3. Testing for breaks and estimation of the equation (HDI and MSW)

	UD_{Max}	WD_{max}	α_1	β_1	TB_1	α_2	β_2	TB_2	α_3	β_3	TB_3	α_4	β_4
Austria	25	39	-59.91	8.82	2000	22.67	-3.44	2005	6.56	-1.04			
			17.39	2.59		5.21	0.77		1.76	0.26			
Belgium	347	535	-0.81	-0.16	1998	-0.55	1.52	2014	-1.46	-6.56			
			0.06	0.38		0.03	0.22		0.06	0.57			
Bulgaria	66	102	4.63	-0.78	2010	22.32	-3.46						
			2.61	0.40		7.86	1.18						
Cyprus(b)	307	375	0.07	2.34	2007	0.76	6.89	2011	-0.35	0.72			
			0.02	0.08		0.70	4.28		0.21	1.53			
Czechia	116	120	-13.71	1.89	2000	-13.49	1.82	2008	11.66	-1.90	2013	-68.22	9.89
			2.98	0.45		1.36	0.20		2.65	0.39		5.82	0.86
Denmark	44	68	-21.85	3.16	2002	-57.84	8.45	2006	-21.14	3.07	2011	9.45	-1.42
			5.32	0.79		3.75	0.55		7.19	1.05		11.34	1.66
Estonia	144	176	-4.00	0.47	2008	32.91	-5.06	2013	-46.61	6.74			
			1.80	0.27		9.86	1.46		8.49	1.25			
Finland	103	103	-29.94	4.33	2000	-12.03	1.66						
			2.26	0.34		2.25	0.33						
France	213	328	-18.47	2.64	2005	10.60	-1.66	2013	-26.10	3.75			
			2.29	0.49		2.83	0.42		6.23	0.92			
Germany(a)	171	264	-0.45	-0.02	2002	-0.85	-2.88	2006	-0.13	4.56	2014	-0.75	-4.31
			0.05	0.34		0.05	0.46		0.03	0.34		0.10	1.47
Greece(b)	24	37	1.08	8.44	1999	-0.62	1.15	2005	-0.39	2.48	2009	-0.85	-1.00
			0.38	1.56		0.01	0.03		0.14	0.88		0.21	1.42
Hungary(a)	155	238	-0.28	1.60	1999	-0.60	0.80	2006	-2.50	-8.39	2010	-1.14	-0.98
			0.24	0.85		0.02	0.10		0.35	1.77		0.15	0.82
Ireland(b)	38	59	-0.25	1.78	2000	-0.04	2.17	2008	-0.96	-4.35	2012	-0.56	-0.27
			0.06	0.27		0.11	0.82		0.61	5.34		0.02	0.31
Italy	264	323	-12.75	1.79	1998	-16.28	2.32	2005	24.89	-3.77	2011	-1.16	0.07
			0.95	0.14		0.78	0.12		4.95	0.73		8.59	1.26
Latvia	113	174	-1.47	0.02	2000	-9.03	1.19	2014	-5.01	0.61			
			3.12	0.48		2.68	0.40		4.49	0.67			
Lithuania	73	89	-9.22	1.27	1998	-21.70	3.12	2002	-15.69	2.20			
			6.25	0.95		2.02	0.30		1.61	0.24			
Luxembourg	276	337	-13.55	1.94	2011	12.93	-1.97						
			1.07	0.16		6.30	0.92						
Netherlands	191	295	-52.84	7.73	1999	-2.43	0.28	2008	18.66	-2.82	2012	19.18	-2.90
			8.73	1.29		1.18	0.17		8.19	1.20		1.91	0.28
Poland(a)	617	950	-0.73	1.68	2000	-3.03	-7.67	2004	-1.35	-1.04	2013	0.07	8.65
			0.10	0.38		0.22	0.95		0.11	0.58		0.03	0.18
Portugal	107	164	-43.48	6.40	1998	-10.88	1.51	2007	44.96	-6.80	2013	-64.93	9.52
			2.08	0.31		3.44	0.51		6.50	0.97		10.24	1.51
Romania	136	182	-11.57	1.60	2009	-0.73	-0.09						
			2.17	0.33		21.39	3.19						
Slovakia	3915	4758	60.45	-9.32	2001	-10.74	1.42	2007	-0.08	-0.16	2014	-165.91	24.44
			8.16	1.23		3.84	0.57		4.32	0.64		1.43	0.21
Slovenia	113	175	20.86	-3.20	2001	-39.54	5.74	2010	-107.23	15.68	2014	-30.92	4.44
			3.95	0.59		8.17	1.21		3.15	0.47		4.40	0.65
Spain	573	883	-49.49	7.30	1999	32.58	-4.91	2005	58.76	-8.79	2013	-30.59	4.39
			4.66	0.70		3.74	0.56		2.49	0.37		1.88	0.28
Sweden(a)	198	242	-0.42	3.47	1998	-0.14	6.38	2003	-0.07	6.33	2009	-0.81	-0.20
			0.03	0.24		0.10	0.95		0.17	1.63		0.03	0.38
United Kingdom	450	693	-28.56	4.14	2003	56.45	-8.39	2010	5.15	-0.86			
			2.18	0.32		4.81	0.71		4.52	0.66			

This table presents the results of the estimation of model (2), using HDI instead of per capita GDP, with TBj (i=1,2,3) being the estimated periods when the break appears. The number of breaks has been selected by using the sequential procedure described in Bai and Perron (1998). UDmax and WDmax test the no structural break null hypothesis, which is rejected in all the reported cases when using the appropriate critical values. Robust standard deviations are presented below the estimated parameters.

(a)Structural breaks selected by BIC

(b)Last observation 2017

Figure 1. Elasticities between MSW and GDP/HDI

Figure 1.A MSW/GDP estimated elasticities in 1995

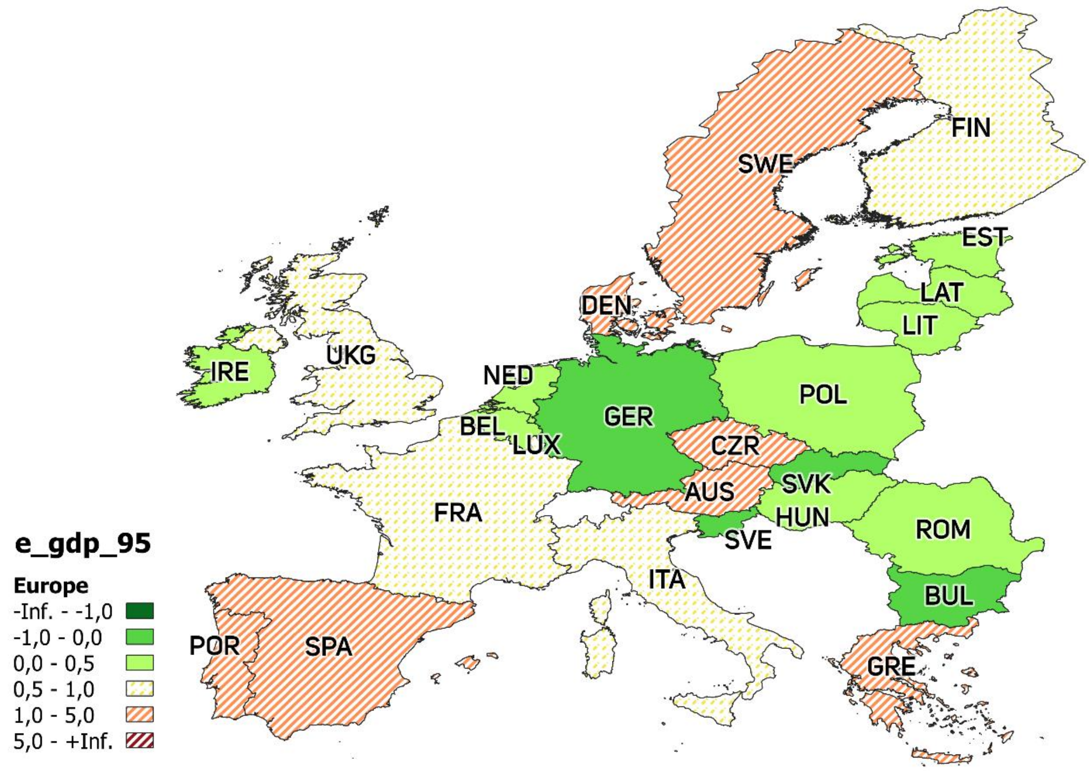


Figure 1.B HDI/GDP estimated elasticities in 1995

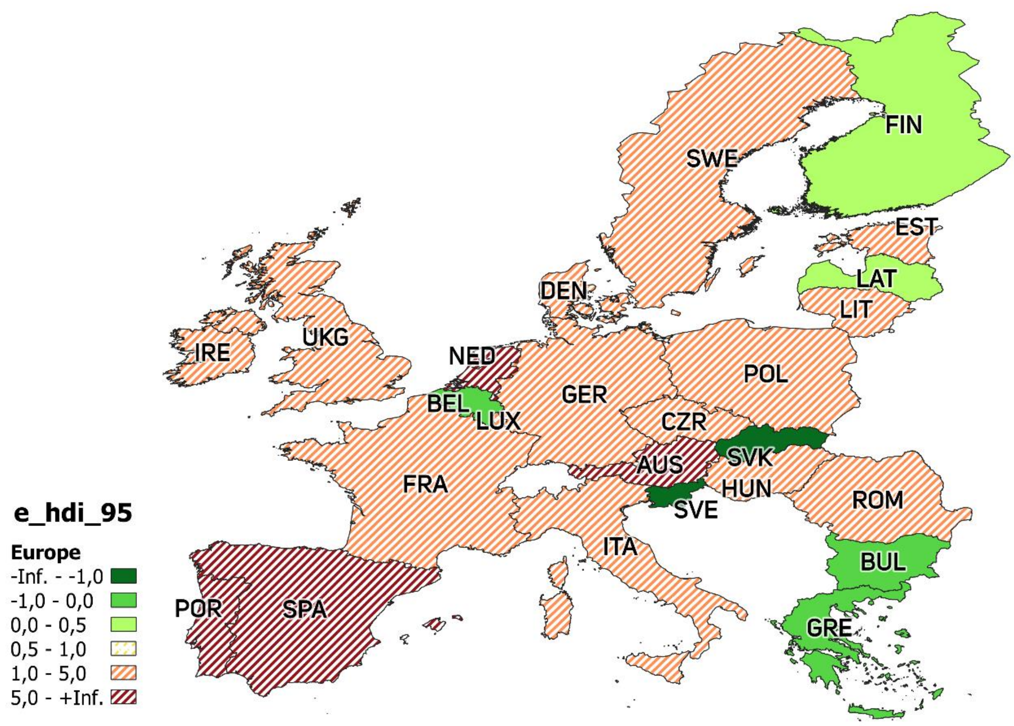


Figure 1.C MSW/GDP estimated elasticities in 2007

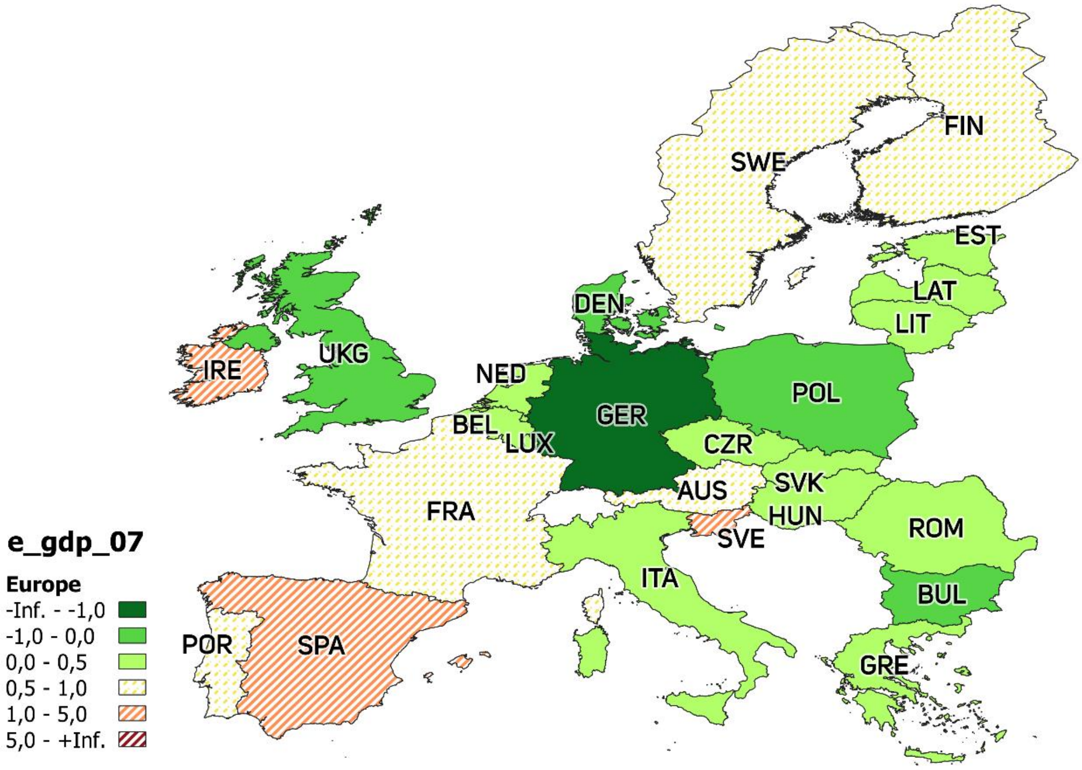


Figure 1.D HDI/GDP estimated elasticities in 2007

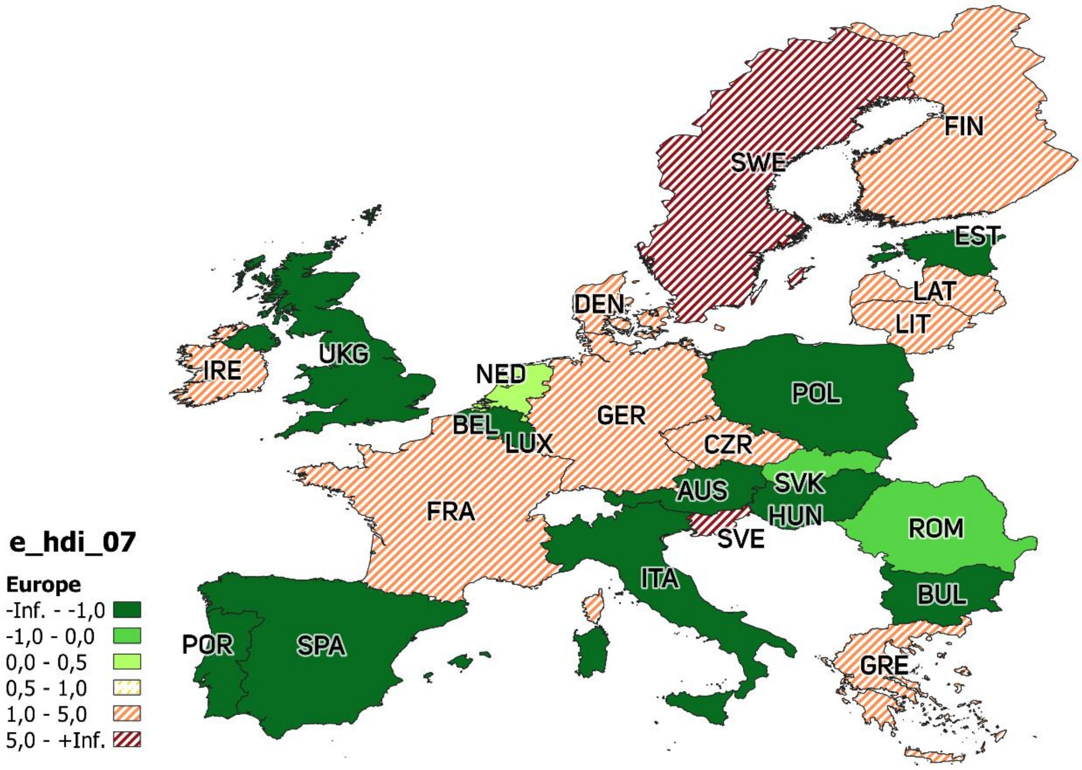


Figure 1.E MSW/GDP estimated elasticities in 2018

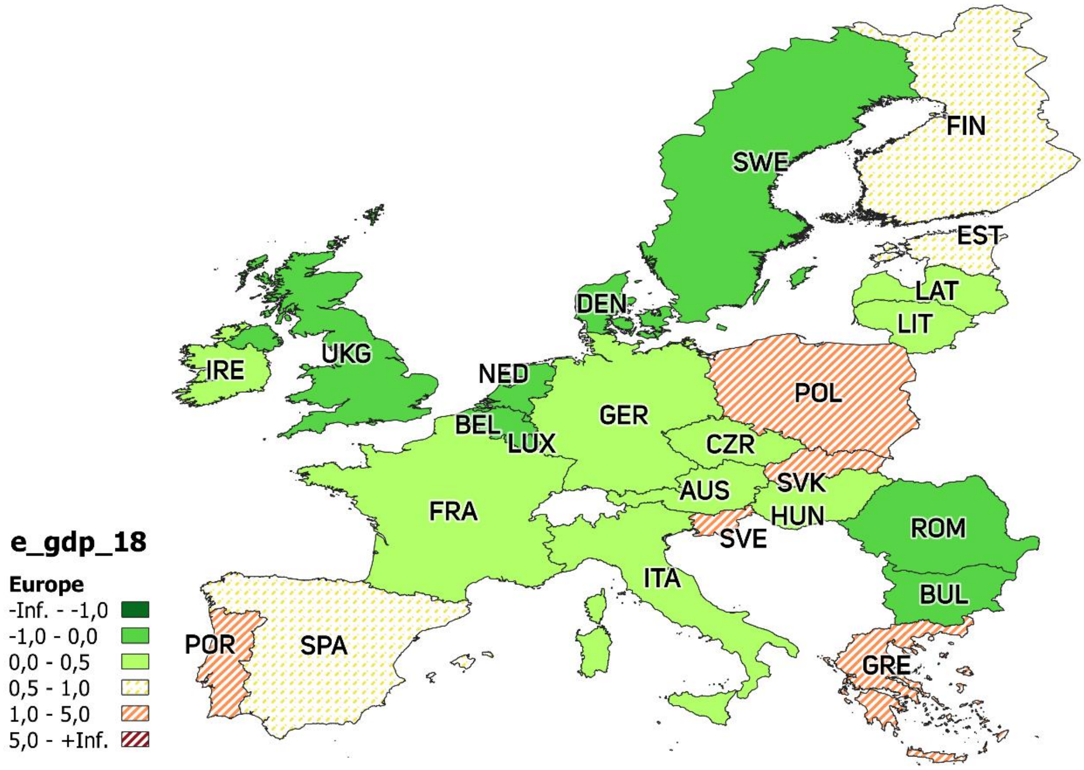
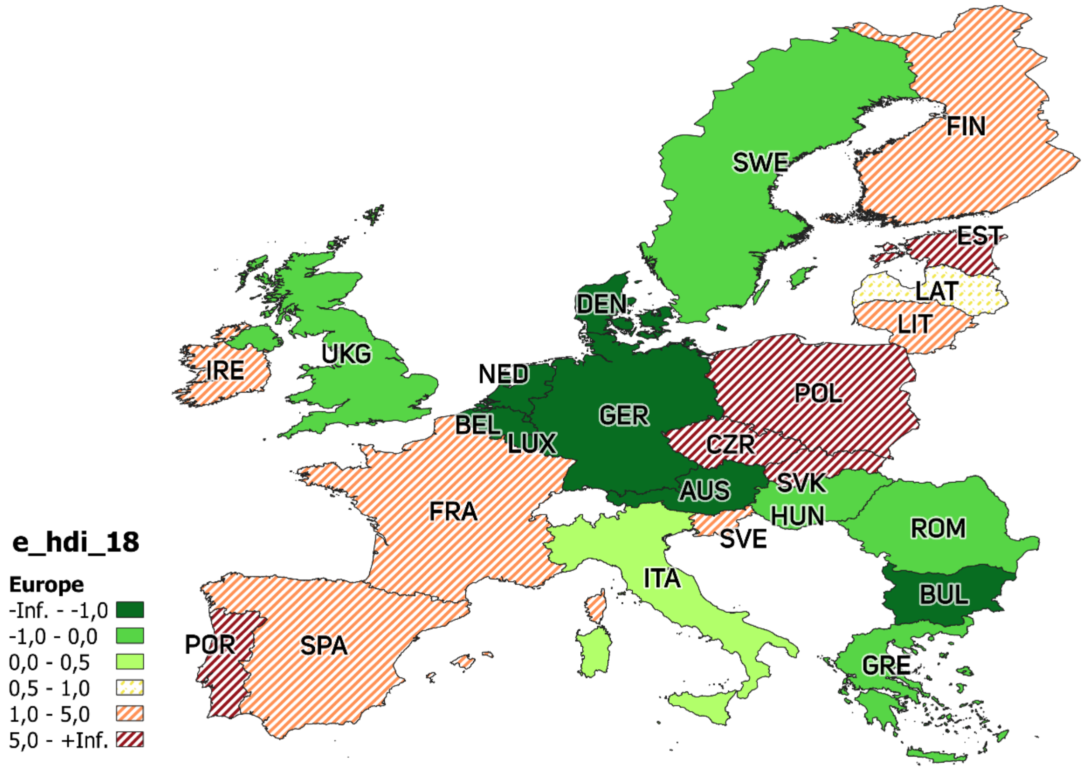


Figure 1.F HDI/GDP estimated elasticities in 2018



Chapter III. A study of the evolution of recovered and unrecovered waste relative to GDP for OECD countries as a way of assessing the effectiveness of circular economy policies

3.1. Introduction

The traditional linear model of production and consumption has recently been challenged by some alternative models built on the foundations of the search for more sustainable economies. The linear model follows the idea that goods are manufactured from raw materials, are later used and, finally, become waste, in the sense that there is no return of these manufactured goods to the economy. In contrast to this view, new trends in waste management advocate the transition from these models to others based on the circularity of the economy. The origin of the concept of circular economy (CE) is not clear, as is noted in Winans *et al.* (2017). It combines the advances in theoretical models of environmental economics (Pearce et al., 1990) with the practices of companies and institutions carried out in the field of sustainability (Ellen MacArthur Foundation, 2013). Most authors coincide on the fact that the circular economy involves the redefinition of the linear system of production, consumption, and waste treatment in order to avoid undesirable rebound effects (Zink and Geyer, 2017) and to move towards a closed-loop system that generates a smaller environmental footprint. Taking this into account, it is clear that waste generation is one of the key aspects of the CE, with waste management being governed by the 3Rs principle: Reduce, Reuse, and Recycle. More recently, other authors have considered it necessary to add a fourth R, Recovery, as suggested by Kirchherr *et al.* (2017).

The novelty of the CE concept has attracted the interest of many environmental researchers, which has generated a vast literature where CE has been studied both from the theoretical and, especially, the empirical point of view, with applications to different countries, sectors, or regulations. Articles that summarize these contributions include George *et al.* (2015), Ghisellini *et al.* (2016), Geissdoerfer *et al.* (2017) and Donaghy (2022), amongst many others. These papers analyze the transition of societies to a more circular economy, placing special emphasis on the consequence of this transition and on the main global challenges to be addressed to reconcile economic development with environmental sustainability.

The results and the conclusions reflected in these papers have led many international institutions, including the European Union, to recognize the importance of the CE in the route to more sustainable economies and, therefore, they have led to the design of strategies devoted to achieving a CE. Some representative reports are those issued by the European Commission (2020a; 2020b) and the United Nations Environment Program (2015; 2017). These efforts were crystallized in the definition of the Sustainable Development Goals (SDGs). In this regard, we should note that Goals 11 and 12 establish several targets directly related to municipal solid waste management: target 11.6, "Member States decided to, by 2030, reduce the adverse per capita environmental impact of cities, including by paying special attention to air quality and municipal and other waste management" and target 12.5, "Member States decided to, by 2030, substantially reduce waste generation through prevention, reduction, recycling and reuse".

This body of legislation has implied notable advances in the sustainability of the economies and its consequences have been studied in the literature. Most papers have focused on the very interesting analysis of the decoupling between waste generation and the evolution of the economy. Some representative papers of this growing literature are those by Mazzanti and Zoboli (2008), Degli Antoni and Marzetti (2019), Sanyé-Mengual *et al.* (2019), and Gardiner and Hajek (2020b), amongst many others. This literature is devoted to the analysis of the reduction of waste generation in the OECD countries, trying to verify whether these countries have decoupled waste generation and economic growth. The results reflected in these papers are mostly positive in terms of relative decoupling and we can consider that the environmental policies have been somewhat successful in reducing waste, although the Great Recession has partially interrupted this process, as Alcay *et al.* (2021) show.

The reduction in municipal solid waste (MSW) generation is undoubtedly encouraging news so far as the sustainability of the economies is concerned. However, this should be treated with some caution given that the literature mostly focuses on the estimation of total solid waste but does not analyze its composition. This is a very important issue because the return of waste to the consumption circle is crucial for an economy to become truly circular. If this is not the case, the absence of recycling and reuse prevents economies from fulfilling the aforementioned 3Rs principle. Therefore, it seems sensible to analyze the degree of circularity of the economies by disaggregating the total waste and examining its evolution.

Against this background, the aim of this paper is to analyze the evolution between per capita municipal solid waste and the evolution of economies, when waste is disaggregated into two components: recovered and unrecovered. This analysis can provide useful results in order to analyze the real degree of circularity of the OECD economies, given that it can help us to better appreciate the capacity of these economies to use goods in a closed-loop manner.

The rest of the paper is organized as follows. Section 2 presents the description of the database, a brief descriptive analysis of the variables employed in the study, and the econometric methods used. Section 3 presents the results, whilst the policy implications are discussed in Section 4. The paper ends with a summary of the most important conclusions.

3.2. Data and methods

3.2.1. Data

Data has been collected from the OECD database. In particular, we use per capita gross domestic product at constant 2015 prices (GDP hereafter) and per capita municipal solid waste (MSW). This latter variable has been disaggregated among its main components: recycling, composting, incineration, and landfill (OECD, 2022). Then, we have constructed two waste measures: per capita recovered waste (RW) and per capita unrecovered waste (URW). RW is the addition of the per capita recycled waste and the per capita composted waste. By contrast, URW is obtained by adding the rest of the per capita municipal solid waste components (incinerated, landfilled, or other operations).

The available sample varies for the different countries included in the study. The starting year is 1990, although some countries only have information since 1995. The last observation is from 2019, but some countries only provide information up to 2018. The OECD countries included in the study are the following: Austria, Belgium, France, Germany, Hungary, Italy, Japan, Korea, the Netherlands, Poland, Spain, Switzerland, the United Kingdom, and the United States. Some missing values have been linearly interpolated, when necessary.

The study of the relationships between RW/URW and GDP can provide a very valuable analysis of waste prevention policies, changes in product design, as well as consumption habits. Before proceeding to their estimation, it seems sensible to carry out a descriptive analysis of the variables in order to better understand their evolution across the sample.

3.2.2. Descriptive analysis

Tables A1 and A2 in the Appendix 3 present the values of the GDP, RW and URW variables for the sample from 1995 (the common initial value) to 2018 (the common final value), as well as their corresponding growth rates. We have also split the total sample into two subperiods (1995-2007) and (2008-2018), with the latter also being divided into the GR recession period (2008-2013) and the subsequent recovery period (2014-2018). This partition can help us to observe the possible influence of the GR on the evolution of the variables.

As we can see, the GDP at the beginning of the sample shows substantial variability. It varies from a minimum value of \$12,141 (Poland) to a maximum value of \$52,047 (Switzerland). At the end of the sample, the range of values ranges from \$30,259 (Poland) to \$68,580 (Switzerland). Korea exhibits the fastest growth rate (4.7%,) during the period before the GR, whilst Japan shows the slowest (1%). If we now consider the subperiod (2008-2018), we can see that Poland shows the highest growth rate (3.4%), whilst Italy presents the lowest growth rate (- 0.5%). If we focus on the recovery period (2014-2018), Poland once again shows the highest growth rate (4.4%). We should also mention that there is an intense GDP growth across the selected sample, although the GR clearly reduced this growth. Furthermore, we can see some catching-up convergence, in the sense that the countries with the lowest GDP at the beginning of the sample exhibit the highest growth rates.

RW varies from 5.3 per capita kgs (Poland) to 246.7 per capita kgs (Germany) at the beginning of the sample, whilst it goes from 67.5 per capita kgs (Japan) to 406.8 per capita kgs (Germany) at the end of the sample. We should also note that the percentage of recovered waste over the total waste varies from 50.5% (Austria) to 1.6% (Hungary) in 1995, whilst it goes from 67.1% (Germany) to 20% (Japan) at the end of the sample. As can be seen, although the lowest bound has substantially increased, the largest one has risen only moderately, with many countries with a level of waste recovery that barely exceeds 50% of the total waste. If we now consider the period 1995-2007, Hungary shows the highest increase in recovered waste (18.2%), whilst the USA presents the smallest growth rate (2.5%). It is also noticeable that all the countries exhibit positive growth rates during this period. If we now consider the post GR period (2008-2018), we can see that Poland shows the highest growth rate (12.7%), with Spain exhibiting the lowest (-2.7%). The effect of the GR can be seen better if we consider the period 2008-2013. Thus, Spain shows the lowest growth rate (-7.5%), whilst Italy maintains a growth rate of 8%. By contrast, during the recovery period (2014-2018), Poland presents the highest growth rate (11.9%), with Japan showing the lowest (-1.5%). Then, we can see that many countries show a decline in recovered waste after the GR, a situation that is partially corrected during the recovery period. Additionally, we can also see substantial heterogeneity.

URW goes from 216.4 per capita kgs (Austria) to 549.7 per capita kgs (USA) at the beginning of the sample. Likewise, it goes from 150.6 per capita kgs (Korea) to 550.6 kgs (USA) at the end. The evolution over time has also been quite heterogeneous, as the growth rates of the different periods reflect. The URW growth rates range from 2.1% (Korea) to -4.5% (Germany) during the pre-GR period. Moreover, 11 of the 14 countries present negative growth rates. However, this satisfactory performance seems to be maintained after the GR, given that we can see that the growth rates go from 1.1% (Austria) to -5% (Italy) during the 2008-2018 period and 12 of the 14 countries exhibit negative growth rates during that period. Moreover, the magnitudes of the growth rates after the GR are (on aggregate) lower than those of the pre-GR period. However, this result can be qualified if we consider the 2008-2013 and the 2014-2018 periods. We can see a truly clear reduction of the growth rates during the 2008-2013 period. For instance, these growth rates go from -6.7% (Italy) to 2.1% (Austria), with Austria and Germany (0.5%) being the only ones that exhibit positive growth rates. The picture slightly changes for the next period (2014-2018). We can see that the growth rates vary from -3.4% (The Netherlands) to 3.4% (USA). Additionally, the magnitudes increase (on average) with respect to the previous period, although it is true that 10 of the 14 countries continue to exhibit negative growth rates.

However, only 4 countries show a growth rate lower than -1.5% (8 for the 2008-2013 period). Then, the effect of the GR is clear, implying a noticeable reduction of the URW during the worst years of the crisis.

This preliminary analysis shows, on the one hand, that the behavior of the 14 countries is far from homogeneous. On the other hand, the existence of a close relationship between RW/URW and the evolution of the economies is clear, as is the influence of the GR on this relationship. The methodology employed for the estimation of this relationship is presented in the following section.

3.2.3. Methodology

As previously mentioned, the aim of the paper is to estimate the relationships between the evolution of recovered waste and unrecovered waste for each one of the selected OECD economies. The model specification can be stated as follows:

$$\ln(\mathrm{RW})_{\mathrm{it}} = \alpha_{1\mathrm{i}} + \beta_{1\mathrm{i}} \ln(\mathrm{GDP}_{\mathrm{it}}) + u_{\mathrm{it}} \quad (1)$$

$$\ln(\mathrm{URW})_{\mathrm{it}} = \alpha_{2\mathrm{i}} + \beta_{2\mathrm{i}} \ln(\mathrm{GDP}_{\mathrm{it}}) + \mathrm{v}_{\mathrm{it}} \quad (2)$$

With i = 1, 2, …,14 and t =1990/1995, 1991, …, 2018/2019.

However, the descriptive analysis has alerted us to the possible existence of structural breaks, especially due to the Great Recession. Consequently, we consider that the stability of this relationship may be questionable, as Alcay *et al.* (2020) show for total municipal solid waste generation. Therefore, it seems sensible to estimate the previous system allowing for the presence of some breaks. Thus, the model can be specified as follows:

$$\ln(\mathrm{RW}_{\mathrm{it}}) = \mathrm{a}_{1\mathrm{ij}} + \beta_{1\mathrm{ij}} \ln(\mathrm{GDP}_{\mathrm{it}}) + u_{1\mathrm{it}} \quad (3)$$

$$\ln(\mathrm{URW}_{\mathrm{it}}) = \mathrm{a}_{2\mathrm{ij}} + \beta_{2\mathrm{ij}} \ln(\mathrm{GDP}_{\mathrm{it}}) + \mathrm{u}_{2\mathrm{it}} \quad (4)$$

with j=1, …, m being the number of breaks that occur at periods TB1, …, TBm.

We have several possibilities to estimate the system of equations composed of (3)-(4). We have opted to use the methodology proposed by Qu and Perron (2007) (QP hereafter)[1] . This methodology allows for the presence of multiple structural changes that may occur at unknown periods, whilst these breaks can affect the regression coefficients and/or the covariance matrix of the errors. Moreover, the distribution of the regressors does not have to remain constant across regimes and the method of estimation is quasi-maximum likelihood based on normal errors.

The general approach of the methodology of Qu and Perron (2007) is the following. Let us consider that we have N cross-sections and the sample size is of dimension T. Then, let the vector y_t be the one that includes the endogenous variables of the system, in such a way that y_t= (y_{1t}, … y_{nt}). Similarly, let z_t be the (qx1) vector that contains the regressors z_t= (z_{1t},…, z_{qt})′. We should assume that the variables included in y_t an z_t do not exhibit unit roots. The selection matrix "S" is of dimension nq x p with full column rank, where p is the total number of parameters. It involves elements that take the values 0 and 1 indicating which regressors appear in each equation. The total number of structural changes in the system is m and the break dates are denoted by the m vector M = (TB1; …; TBm). The subscript j indexes a regime (j =1, …, m + 1), the subscript t indexes the temporal observation (t =1, … T), and the subscript e indexes the equation (e =1, …, f) to which a scalar dependent variable y_{it} is related.

The general model proposed is of the form:

$$y_t = z_t' S\beta_j + u_t \qquad (5)$$

[1] We have also followed the alternative methodology designed in Bai and Perron (1998, 2003a, 2003b) and have estimated equations (3) and (4) in an independent manner for each country. The results are quantitatively similar to those presented here, which provides robustness to the analysis. They are available in the Appendix 3, table A6.

with u_t having mean 0 and covariance matrix Σ_j for $T_{j-1}+1 \leq t \leq T_j$. If we compare equation (5) with the system (3)-(4), it is clear that $y_t = \{\ln(RW_{it}), \ln(URW_{it})\}$ and $z_t = \{\ln(GDP_{it})\}$.

To determine the number of breaks in the system, we have used the UDmaxLRT and WDmax LRT statistics to test whether at least one break is present. When the tests reject it, the test SEQt ($\ell+1 \mid \ell$) is sequentially applied for ℓ=1, 2 until the test fails to reject the null hypothesis of no additional structural breaks. Following critical values derived from response surface regressions, the tests offer evidence of the presence of two breaks in the system of equations for each country.

The next Section present the results of the application of the QP methodology for the 14 countries included in our sample. Previous to the analysis of the results, we should note that, as previously mentioned, the use of this methodology requires the variables included in the system to be non-integrated. Then, we should verify in a first step that the variables are stationary. For this purpose, we have employed both the Dickey Fuller-GLS statistic, proposed by Elliott *et al.* (1996), and the statistics developed in Carrion-i-Silvestre *et al.* (2009), which allows the trend function of the variables to present structural breaks at unspecified periods. The number of lags has been included by considering the statistic MIC proposed in Ng and Perron (2001). The results of the tests are reported in Tables A3-A5 in the Appendix 3. As we can observe, the series do not present unit roots once structural changes are considered and the QP methodology can be used in our framework.

3.3. Results

The results of the application of the QP methodology are presented in Table 1 and Figure 1, where we can see the estimations of the RW/GDP and URW/GDP elasticities. The first insight that emerges from the analysis of Table 1 is the importance of considering the presence of changes in the elasticities. We can see that the null hypothesis of no structural breaks is clearly rejected for all the cases, in such a way that the QP methodology estimates two structural breaks for the different countries. Although the periods when these breaks occur are not totally coincident, we can consider the existence of three main estimated segments. The first estimated segment goes from the beginning of the sample to 2001, clearly related to the period where the implementation of waste treatment and recovery policies and the harmonization of national statistics was particularly important. This is followed by the period before the GR (2002-2007) and, finally, the one that reflects the evolution after the GR (2008- 2018/2019).

If we consider the initial segment, we can observe that all the RW/GDP estimated elasticities are positive and, moreover, greater than one, with the exception of Switzerland (0.82). The rest of the estimated elasticities range from 1.65 (Poland) to 11.65 (Belgium). The presence of some remarkably high estimated elasticities is better understood if we take into account that the waste recovery industry was still incipient during this initial period and, consequently, the expansion of the degree of recovered waste was remarkable at that time. By contrast, we can see that the URW/GDP estimated elasticities are mostly negative, except for Italy (0.19) and the United Kingdom (0.50). The variation of these estimated elasticities goes from -0.06 (Spain) to -3.40 (Austria).

If we now consider the period previous to the GR (2002-2007), we can see that the RW/GDP estimated elasticities are mostly greater than 1, whilst the range of variation is much shorter than that observed for the preceding period. By contrast, the RW/GDP estimated elasticities of Austria (-0.19) and Italy (-6.01) are now negative. If we compare the results with those of the previous period, we can see that the absolute values of the estimated elasticities decline, reflecting a less intense growth in waste recovery, as Figure 1 also reveals.

We can additionally observe that the URW/GDP estimated elasticities take negative values for 9 of the 14 countries, with Korea (-0.02) and Japan (-2.05) being the extreme cases. Austria (0.67), Hungary (0.10), Italy (2.77), Poland (0.20), and Spain (0.29) exhibit estimated elasticities greater than zero, reflecting a direct relationship between GDP and unrecovered waste generation. The combination of the two relationships allows us to observe a very interesting substituting process between recovered waste and unrecovered waste. For Hungary, Poland, and Spain, although waste recovery continues to increase, GDP no longer favors the reduction of unrecovered waste. By contrast, we should note that Austria and Italy report an increase in unrecovered waste while recovered waste decreases in relation to GDP during this period.

Finally, we should now focus on the post GR period. The analysis of the RW/GDP estimated elasticities show that only five countries show estimated elasticities greater than 1: France (3.46), Hungary (1.31), Italy (4.20), Poland (4.28), and Spain (1.35). Additionally, seven countries show a positive elasticity, but they are lower than one. The range of this second group of elasticities goes from 0.03 (UK) to 0.85 (Switzerland). Finally, Belgium (-1.32) and Japan (-0.71) show negative elasticities.

The case of the URW/GDP estimated elasticities is somewhat different. We can see that only 3 countries show a positive estimated elasticity: Austria (0.04), Korea (0.23) and the United States (0.54). The rest of the countries report a negative elasticity, with these estimated elasticities going from -0.18 (Spain) to -2.84 (Italy).

The joint analysis of the estimated elasticities offers some additional interesting insights. We can see that RW/GDP and URW/GDP elasticities are negative for Belgium and Japan, suggesting the existence of absolute decoupling. We can also observe that France, Germany, Hungary, Italy, Poland, Spain, Switzerland, the Netherlands, and the United Kingdom show positive RW/GDP and negative URW/GDP elasticities. Therefore, we could consider that the recovery policies were quite effective in these countries. Austria and Korea show a situation of relative decoupling in the two relationships, although with a higher elasticity for RW. Finally, the United States has a relative decoupling relationship but turning to a situation in which unrecovered waste grows more promptly.

The analysis of Figure 1 also provides some useful insights. We can first see that the RW/GDP elasticities clearly reduced across the sample, although they are mostly greater than 0 at the end. By contrast, the URW/GDP elasticities remained generally stable throughout the three sub-periods, although there are differentiated behaviors per country, which are detailed as follows.

Belgium and Japan are two cases where a substitution effect towards waste recovery initially takes place and, after the GR, reaches a situation of absolute decoupling in total waste in which both recovered and unrecovered waste decrease with economic growth. France, Germany, Switzerland, and the Netherlands present consistent positive elasticity for recovered waste and negative elasticity for unrecovered waste, showing the continuity of substitution between types of waste treatment. Hungary, Poland, and Spain also show a general behavior of recovering countries, with the nuance that unrecovered waste maintained a positive elasticity during the period of growth prior to the GR. Notably, the GR significantly decreases the elasticity for recovered waste in Germany and Switzerland. The Italian behavior is erratic before and after the GR in the ratio of recovered waste, as well as unrecovered waste. Italy is the only country to show such a substantive shift towards sustainability and increased waste recovery with the arrival of the GR. Finally, Austria, Korea and the United States show ratios that before the GR involved a substitution of unrecovered waste for recovered waste. However, the advent of the GR led to a decrease in the elasticities of the ratio of recovered waste to GDP, as well as a change from negative to positive values for the elasticities of unrecovered waste to GDP. Finally, although Austria (0.85 vs. 0.04) and Korea (0.67 vs. 0.23) have higher elasticities for waste recovery, which still marks a trend of higher growth of recovered waste than unrecovered waste, in the United States (0.45 vs. 0.54) the value for unrecovered waste is higher after GR.

3.4. Discussion

The analysis of the previous results allows us to observe that waste management has evolved quite effectively, and most countries have achieved a relative decoupling between waste generation and economic growth. This is an exceptionally good step in the right direction, in that the countries have clearly reduced waste generation. However, the final target of these policies was the transition to a CE and we should note that circularity cannot be captured by the total MSW. Rather, the analysis of the evolution of both recovered and unrecovered waste is required for this. Our analysis has brought to the surface an unexpected composition effect that raises doubts about the real effectiveness of environmental policies around MSW. Here, we are thinking of the relatively deficient performance in terms of recovered waste.

In this regard, we should note that an economy can only be considered circular if three conditions are met. First, decoupling, or at least relative decoupling, between MSW and GDP is necessary in order to guarantee that the waste reduction principle holds. However, circularity also requires that reuse and recycling principles hold. Then, this implies that the RW/GDP elasticity of RW should be positive, indicating that waste recovery is being promoted with economic growth and, finally, URW elasticity should be negative, in order to guarantee the existence of a substitution effect from linear waste treatment processes to circular ones.

Following these conditions, our results show that the countries have followed waste policies devoted to waste prevention. The consequence has been the reduction of the unrecovered waste, but the evolution of the recovered waste has not been so positive. However, we should recognize that waste reduction policies are easier to implement than waste recovery ones. This latter process has some limitations which could make it impossible to achieve circularity. Perhaps it would be more appropriate to think in terms of the transition towards spiral economies.

Some of these restrictions respond to physical and technological limitations. We should cite Valero and Valero (2019) in this regard. These authors point out that there are losses of energy and materials in each phase of recovery, following the thesis sustained by Georgescu-Roegen (1971), which make it impossible to achieve 100% effective recycling. Moreover, the increasing use of complex alloys, components, and materials, involves a clear increment in the cost of recycling and, in some cases, the loss of part of the elements produced, making the recycling of waste almost impossible. Then, we can consider that circularity is almost impossible to achieve, and it is better to posit the move towards spiral economies as a real attainable goal.

We should also take into account that some additional problems may arise during the process of waste management and separation itself, as is the case of incorrect sorting or the organizational aspects of the sector (Van den Bergh, 2020). In particular, it is quite noticeable that the proportion of plastic waste is increasing in municipal solid waste. Di *et al.* (2021) and Jang *et al.* (2020) highlight that plastics present a paradigmatic case of the difficulty of achieving high recycling or even energy recovery rates through incineration due to inefficient recovery processes, high technological requirements, as well as significative material losses. Therefore, more efforts are required to develop the appropriate technologies that can help to recover higher proportions of MSW.

Another restriction to waste recovery is related to the social-political aspects of waste management, including the lack of effectiveness of European and international waste policies. Despite a reasonably good performance of the initiatives of these institutions, we agree with Burns *et al.* (2020) that these policies have shown a lack of ambition and concreteness. If we combine this with the budgetary constraints that the fight against the GR imposed on OECD economies (Bartl, 2014), the consequence is that more aggressive environmental policies aimed to promote recovery have been postponed or, even worse, discarded.

Personal motivations and education may also affect recycling, as noted by Minelgaitė and Liobikienė (2019) who link the lack of environmental awareness and poor waste sorting behavior at home with socioeconomic levels. Similarly, Ranta *et al.* (2018) and Kirchherr *et al.* (2018) offer evidence that institutional and cultural barriers determine the effectiveness of public environmental policies. Such policies aimed at waste management may have encountered major difficulties in penetrating social norms and thus failed to achieve their objectives. The conjunction of changing the priorities of environmental policy and regulation together with the relaxation of recovery objectives, added to the social difficulties in adapting norms and habits to the regulations of the moment, generate a reduction in the effectiveness of recovery policies.

Another socio-political issue is the debate on the regulations and incentives existing among the different alternatives for treating waste, as discussed by Stumpf *et al.* (2021). In this regard, the promotion of landfill reduction and waste prevention leads to unexpected incentives that promote incineration instead of more sustainable treatments. Egüez (2021), for high-income countries, and Okumura *et al.* (2014) for Japan and Korea, find that there is a substitution effect from recycling to incineration. Another fact that could encourage the expansion of energy recovery through incineration is the international context of rising electricity and fossil fuel prices, leading to incineration being considered a more profitable way of producing electricity. For this reason, even if waste disposal is reduced, incentives and regulation are determinant for boosting sustainable alternatives such as recovery instead of incineration.

In summary, OECD countries are facing a situation of stagnation in waste recovery promotion policies, partially explained by the existence of multiple (physical, technological, and socio-political) limitations. If the aim is truly to achieve a more circular economy, it is mandatory to comply with the international objectives and agreements and to consolidate these policies with a shared and holistic perspective. Unfortunately, some waste recovery policies have been relegated by many institutions to the background as a consequence of the Great Recession.

3.5. Conclusions

This paper has studied the relationship between waste generation and economic development for a sample of 14 OECD countries. Unlike previous papers, we have disaggregated the total waste into recovered and unrecovered waste. The use of the methodology proposed by Qu and Perron (2007) has allowed us to find the existence of two structural breaks in this relationship, with the Great Recession playing a crucial role in this regard.

Once these breaks were identified, we obtained a number of interesting insights. Our results confirm that waste management policies have been quite effective for achieving relative decoupling between total waste generation and the evolution of the economy. Likewise, the disaggregation of waste has permitted us to observe that this goal has been mostly attained thanks to the superior performance of unrecovered waste, in the sense that the estimated URW/GDP elasticities are mostly negative. By contrast, the recovered waste elasticities show a more ambiguous behavior. It is true that they have reduced across the sample, and this is positive news because it implies the existence of relative decoupling in total waste at the end of the sample. However, it also shows that there are some limitations to waste recovery that do not allow the recovered waste to grow as much as would be desirable to advance the construction of more sustainable and circular economies.

There are some explanations for this fact. Some are related to the economic restrictions caused by the GR, which has conditioned the development of more aggressive waste policies, as well as to human limitations, to legislation and, more importantly, to education. Other factors are related to the physical and technological aspects of waste management. In particular, the difficulty of recovering plastic, whose importance is growing in the composition of the total waste, suggests the need for more research to solve this problem. If these problems are not solved, it will be quite difficult for economies to be genuinely circular. We would need to think in terms of spiral economies, given the inevitable losses of material.

We conclude that the transition to a truly circular economy will require a greater effort by institutions, in order to strengthen public policies aimed at increasing waste recovery, so that materials can be reintegrated into the production cycle rather than being dumped or incinerated. This shift will not only require a continued emphasis on innovation, but also a redefinition of the incentives and behaviors by which companies and citizens guide their production, consumption, and disposal patterns.

Finally, we would like to note that this work has certain limitations, most of them related to the short time span employed in the paper and the uncertainty of the evolution of the different waste streams. We intend to continue research on these limitations, especially on disaggregation by the different waste streams, paying special attention to plastic waste given its clear increase in recent years.

3.6. Figures and tables

Figure 1. Elasticities distribution by period and country

Figure 1.1 RW and URW elasticities during 1995

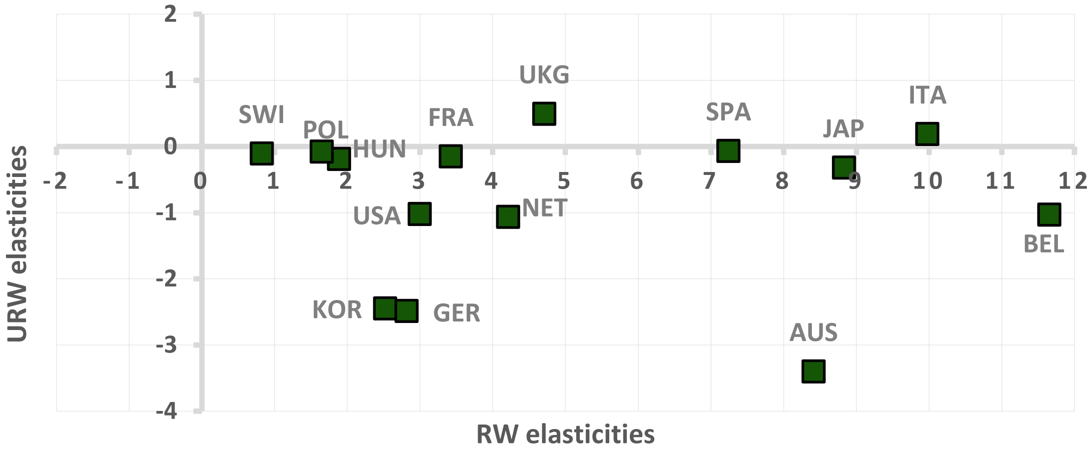
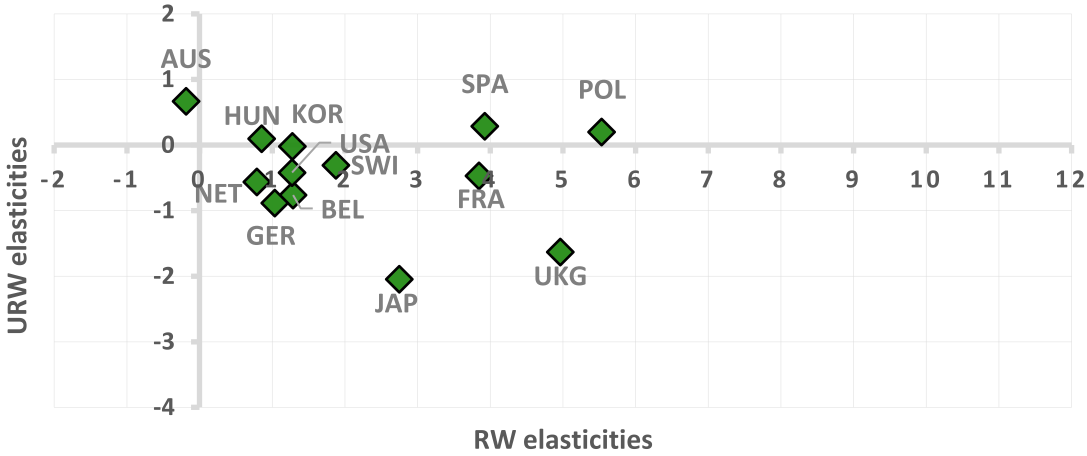


Figure 1.2 RW and URW elasticities during 2007



*Italy (-6.0, 2.8) has been omitted to clarify the interpretation.

Figure 1.3 RW and URW elasticities during 2018

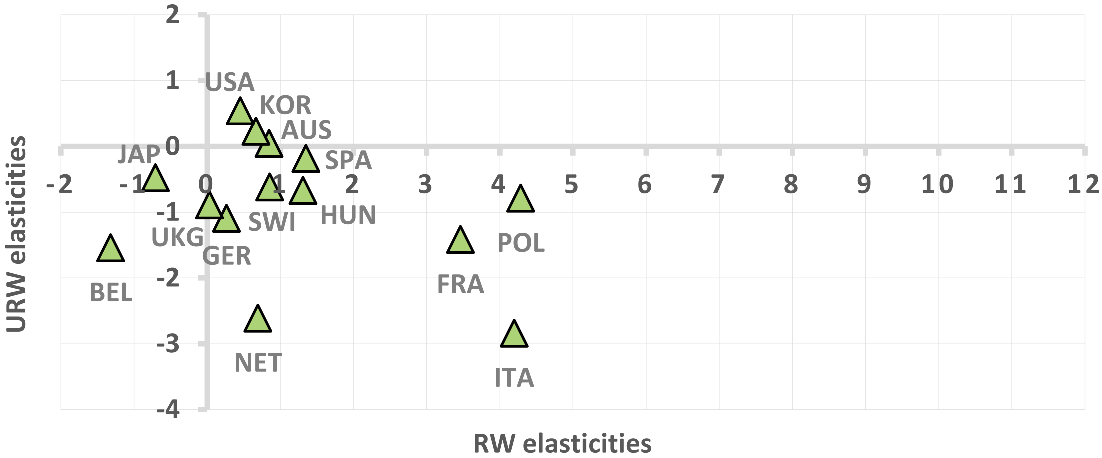


Table 1. Testing for breaks and estimation of the equations (RW and URW as a function of GDP)

Country	Sample	WD_{max} LRT	Model	a_0	β_0	TB_1	a_1	β_1	TB_2	a_2	β_2
AUS	90-19	373	*RW*	-90.29	8.41	1998	0.97	-0.19	2010	-10.29	0.85
				18.70	1.77		3.03	0.28		2.57	0.24
			URW	34.36	-3.40		-8.70	0.67		-1.89	0.04
				7.00	0.66		3.21	0.30		1.46	0.13
BEL	90-19	107	*RW*	-124.40	11.65	1998	-16.17	1.29	2008	12.70	-1.32
				6.80	0.65		3.51	0.33		4.19	0.39
			URW	9.63	-1.03		6.63	-0.76		14.94	-1.54
				9.05	0.86		2.45	0.23		1.90	0.18
FRA	95-19	639	*RW*	-38.08	3.42	2002	-42.52	3.85	2011	-38.30	3.46
				2.17	0.21		8.09	0.76		3.07	0.29
			URW	0.59	-0.15		3.96	-0.47		13.88	-1.42
				0.56	0.54		0.46	0.44		3.02	0.28
GER	95-19	140	*RW*	-30.90	2.81	2002	-12.07	1.04	2011	-3.74	0.27
				4.61	0.44		1.67	0.16		3.52	0.33
			URW	25.12	-2.48		7.95	-0.88		10.24	-1.10
				2.36	0.22		5.77	0.54		1.77	0.16
HUN	95-19	94	*RW*	-23.27	1.89	2003	-11.44	0.86	2011	-15.49	1.31
				6.70	0.68		1.13	0.11		2.37	0.23
			URW	1.03	-0.18		-1.98	0.10		5.50	-0.67
				0.85	0.09		5.50	0.54		1.89	0.18
ITA	95-19	2,075,221	*RW*	-108.19	9.98	2003	61.69	-6.01	2011	-45.72	4.20
				13.31	1.26		25.27	2.39		6.87	0.65
			URW	-2.87	0.19		-30.26	2.77		28.61	-2.84
				1.22	0.12		9.86	0.93		5.81	0.55
JAP	90-18	2,494	*RW*	-95.70	8.83	1999	-31.63	2.75	2008	4.87	-0.71
				18.93	1.81		4.09	0.39		1.13	0.11
			URW	2.35	-0.32		20.51	-2.05		3.80	-0.48
				1.22	0.12		3.63	0.34		0.41	0.04
KOR	90-18	2,539	*RW*	-27.02	2.52	1998	-14.71	1.28	2009	-8.53	0.67
				1.87	0.19		0.27	0.03		2.39	0.23
			URW	22.78	-2.45		-1.35	-0.02		-4.33	0.23
				2.46	0.25		7.84	0.78		0.72	0.07
NET	91-19	14,264	*RW*	-46.10	4.21	1999	-9.81	0.79	2009	-8.82	0.70
				17.24	1.63		1.30	0.12		1.93	0.18
			URW	10.14	-1.06		4.88	-0.56		26.93	-2.61
				3.89	0.37		1.66	0.15		3.37	0.31
POL	95-19	34	*RW*	-20.69	1.65	2002	-58.59	5.54	2010	-46.31	4.28
				5.61	0.59		5.67	0.57		9.75	0.95
			URW	-0.521	-0.07		-3.277	0.20		6.56	-0.79
				2.08	0.22		2.65	0.27		5.17	0.51
SPA	95-19	78	*RW*	-77.35	7.24	2003	-42.91	3.93	2011	-16.01	1.35
				3.87	0.37		19.39	1.85		3.03	0.29
			URW	-0.07	-0.06		-3.99	0.29		0.75	-0.18
				2.30	0.22		7.55	0.72		1.74	0.17
SWI	90-19	5,458,773	*RW*	-10.50	0.82	1998	-21.77	1.88	2009	-10.48	0.85
				19.34	1.78		5.01	0.45		4.73	0.43
			URW	0.15	-0.10		2.34	-0.31		5.81	-0.62
				11.45	1.05		4.45	0.40		0.35	0.03
UK	95-19	3104	*RW*	-52.16	4.71	2002	-54.46	4.97	2010	-1.93	0.03
				4.98	0.48		27.12	2.55		1.49	0.14
			URW	-5.91	0.50		16.36	-1.63		8.19	-0.89
				0.81	0.08		12.54	1.18		1.95	0.18
USA	90-18	101	*RW*	-33.63	3.00	1998	-15.26	1.27	2007	-6.31	0.45
				6.06	0.57		0.57	0.05		0.85	0.08
			URW	10.27	-1.02		3.92	-0.42		-6.62	0.54
				2.15	0.20		0.76	0.07		3.76	0.34

This table presents the estimation of equations (3) and (4), with TBj (j=1,2) being the periods when the break appears. WDmax LRT statistic tests for the null hypothesis of no structural breaks, rejecting it for all the considered countries. Sample covers the period 19XX-20XX. Robust standard deviations are presented below the estimated parameters.

Chapter IV. A study of the evolution of packaging waste for European countries with a special focus on plastic packaging waste and the effects of the Great Recession and COVID-19.

4.1. Introduction

The development of market economies, productive specialization and the expansion of trade have made it more and more necessary to introduce packaging to hold the properties of products until they reach final consumers. Whereas initially it was established as a system in which packaging was returned to the producer for reuse, it has been gradually replaced, although with greater intensity in the 1980s and 1990s, by single-use packaging, in which plastics and their derivatives played a crucial role.

Whilst consumers and producers initially accepted this change towards single-use packaging, it is also true that European societies became worrying about the increase in waste generated by the use of this new packing formula. We should take into account that this waste increment is a movement against the sustainability of the economic model and, therefore, contrary to the circular economy and waste prevention concepts that predominate the European environmental policies since the end of the 20th century. Moreover, this new packaging formula was also going against the introduction of new municipal waste regulations that seek more effective sorting and recycling systems with which to prevent a purely linear use of materials. Another element that was established during these years was the implementation of EPR (Extended Producer Responsibility) systems, with the application of packaging manufacturing fees and the assignment of the responsibility for the packaging collection, treatment, and recycling to certain national companies as Lorang *et al.* (2022) points out.

As a consequence, it comes as no surprise that single-use packaging methods have been identified as a global environmental pollution crisis (Chen *et al.*, 2021), as noted by the UNEP (2018) and also reflected in the waste programs of the European Union. In this latter regard, we should note that the concepts and objectives of the circular economy in the European Union are clearly reflected in the Packaging and Packaging Waste Directive 94/62/EC and its amendments 2004/12/EC and EU/2018/852, as well as in the Waste Framework Directive 2008/98/EC and the 2018 EU Strategy for Plastics in a Circular Economy. The EU has set targets in this legislation to reduce waste generation, as well as to establish recycling targets for packaging waste of 65% for 2025 and 70% for 2030 and for specific fractions as plastic packaging (50% in 2025 and 55% in 2030) as a culmination of this commitment with a sustainable agenda. This effort is being adapted to the United States, which is currently working on a law on plastic pollution (EPA, 2023) based on the National Recycling Strategy (EPA, 2021).

However, it seems that despite the interest of international institutions and the setting of strict targets, recycling and recovery rates are stagnant in most countries or with a much lower growth than it would be expected to achieve true circularity in the production-consumption processes since the Great Recession (GR hereafter). Authors such as Nicolli *et al.* (2012) suggested that to achieve the 2030 targets and reach absolute decoupling in waste generation it was needed more active and deeper policies beyond the EU given the prominent heterogeneity. More recently, Fitch-Roy *et al.* (2020) have pointed at the problems of the incremental process in European legislation which is unable to transform more deeply the production and consumption system and achieve these objectives. Moreover, this ongoing seems really concerning for plastic waste, whose recycling seems to be more constrained (EU was near 40% at 2019) and whose disposal generates serious health (Prata *et al.,* 2020) and ecological problems (Kögel *et al.*, 2020), most notably in the seas as can be seen in Eriksen *et al.* (2014), Jambeck *et al.* (2015) or Geyer *et al.* (2017) among others.

The issue of packaging waste, especially the one related to the use of plastics, has attracted the attention of the literature in the fields of attitudes towards prevention and recycling of consumers or companies (Tencati *et al*., 2016; Khan *et al.*, 2020; Jacobsen *et al.*, 2022), in the engineering disciplines on the introduction of more effective recycling procedures for packaging and plastics (Larrain et al., 2021), material selection and design (Zhu *et al*., 2022), an economic discussion on taxation and municipal collection systems (Hage *et al*., 2018) or some descriptive analysis (Chioatto and Sospiro, 2023). Bradley and Corsini (2023) and Miao *et al*. (2023) have analyzed the possibilities of reusable or biodegradable packaging as more sustainable alternatives to plastic. Most of these studies are based on the use of cross-sectional data, which may offer very interesting results, although they do not allow to capture certain aspects such as the dynamics of relationships or, very importantly in our view, the possible presence of structural changes over time. Consequently, we can appreciate a certain lack of literature that addresses the study of the performance of packaging waste management with respect to the economic cycle from a time series perspective.

Against this background, the aim of this paper is to analyze the evolution of packaging waste management with respect to the economics from a time series perspective. We want to pay especial attention to the possible effect of some important events on this relationship (here, we are thinking in the Great Recession and the COVID-19 pandemic), also disaggregating the packaging waste management into its plastic and non-plastic components. The results of this research can provide very useful information in order to clarify which is the real degree of circularity in packaging waste management, whilst can also help us to determine how close European countries are to the packaging waste targets imposed by Europe.

The rest of the paper is organized as follows. Section 2 presents the database, a brief descriptive analysis of the variables employed in the study, and the econometric methods used. Section 3 presents the results, whilst the social and policy implications are discussed in Section 4. The paper ends summarizing the most important conclusions.

4.2. Data and methods

4.2.1. Data

The data employed in this study have been obtained from the ENV_WASPAC Dataset constructed by Eurostat (Eurostat, 2023). In particular, we have used the per capita gross domestic product at constant 2020 prices (GDP hereafter), per capita total packaging waste (TPW) and per capita recycled packaging waste (RPW). We have subsequently disaggregated these two variables into their plastic and non-plastic components (This includes paper and cardboard, glass, metals, and wood). Therefore, we will also consider the per capita total plastic packaging waste (TPPW), the per capita recycled plastic packaging waste (RPPW), the per capita total non-plastic packaging waste (TNPPW) and the per capita recycled plastic packaging waste (RNPPW).

The available sample varies for the different countries included in the study. The starting year mostly is 1997, although Portugal only has information since 1999. The last observation is the one of 2020, but some countries only provide information up to 2018 or 2019. Finally, the EU countries included in the study are the following ones: Austria, Belgium, Denmark, Finland, France, Germany, Greece, Ireland, Italy, Luxemburg, the Netherlands, Portugal, Spain, Sweden, and the United Kingdom.

4.2.2. Descriptive analysis

Before applying the methodology described in the next Section, it seems to be appropriate to carry out a descriptive statistic of the variables included in our data set. The main results are presented in the Tables A1-A4 of the Appendix 4. For the sake of make the interpretation of the results easier, we have considered the sample 1998-2019, except for the case of the United Kingdom, which is calculated until 2018.

Table A1 shows that Total per capita package waste (TPW) shows a stable growth during the period 1998-2019 (0.8%), with Portugal (2.6%) and France (-0.2%) presenting the extreme growth rates. France is the unique country with a negative growth rate considering the period as a whole. If we consider the initial year, we can observe that the generation of packages in 1998 goes from 194 Kg (France) to 74 Kg (Greece), whilst it goes from 228 Kg (Germany and Ireland) to 81 Kg (Greece) at the end of the sample. This reveals that packaging per capita increased across the sample while sustaining very similar heterogeneity within countries.

The second part of Table A1 focus on RPW. We can see that the behavior of the per capita recycling package across the sample is quite disclosive (See Figure 4.1). We can see that the growth rates are moves from the highest value of Ireland (8.1%) to the lowest on of Sweden (0.1%). The country with the highest recycling quantities in 1998 was Germany (137 per capita Kg) and the one with the lowest was Greece (26 per capita kgs). The situation in 2019 is a bit different and the recycling quantities go from 155 per capita kgs (Luxembourg) to 49 per capita kgs (Greece). If we now consider the relative recycling rate (Rpac/Tpac), we can see that the average package recycling increased from 46.7% in 1998 to 67.2% in 2019. The Figure 5.1 reflects the evolution of the recycling rates for each country. Belgium (83.5%) reached in 2019 the highest ratio and Greece (60.1%) the lowest. It is promising to see these signs of progress being made towards circularity.

Despite of that, it is not a reality applicable to all cases. Austria have maintained the same proportion of recycled packages, while Germany and Sweeden have reduce their fraction by at least 10 percentage points.

Tables A2-A3 help to analyze the evolution of plastic and non-plastic packaging waste. We should note first that TPPW (Table A2) represents in 2019 around 20,6 % of the total amount of packaging. The growth rate of this fraction has been 1.5% along the sample, with Germany and Luxembourg showing the highest growth rates (3.3% and 3.2%, respectively), whilst Netherlands (-0.3%) and Greece (0%) present the lowest ones. We can also see that plastic packages quantities in 1998 are distributed on a range that goes from 45 per capita kgs (Ireland) to 16 per capita kgs (Sweeden). The distribution goes from 65 per capita kgs (Ireland) to 24 per capita kgs (Finland and Sweden) in 2019. The picture that emerges from that is that plastic packages have grown parallel to an increase in heterogeneity among countries.

If we now consider RPPW (Also in Table A2), we see that the values of this variable have jumped from a range of 12 per capita kgs (Germany) to 1 per capita kgs (Greece, Ireland and Portugal) in 1998 to a variety between 18 per capita kgs (Ireland and Spain) and 8 per capita kgs (Greece) in 2019. The evolution of the natural logs of RPPW could be seen in Figure 4.2. Throughout this period, plastics recycling grew for all sample countries at an average annual rate of 8.3%, with Portugal (14%) and Germany (1.8%) presenting the highest and the lowest value, respectively. If we now focus on the proportion of the total plastic packaging, we can see that it has risen from an average of 14.8% of recycled packaging in 1998 to 40.9% in 2019 (For more country details, check Figure 5.2). At this point only the Netherlands, Spain and Sweden exceed 50% recycling rates. Germany led the recycling of this type of waste at the beginning of the sample (59.1%) and it is the only country that has reduced the fraction it recycles at the end of the sample (43.3%). However, the highest recycling rate in 2019 was the one of the

Netherlands (57.2%). Recycling rates for plastic packaging are still at low levels, although the rate at which they are growing is high, with respect to total packaging.

The evolution of TNPPW is somewhat different, as can be noted from the analysis of Table A.3. This table shows that this variable follows a more modest growth compared to the one of TPPW with an average annual growth of 0.7%. The growth are also very heterogenous, varying from 2.7% (Portugal) to -0.5% (France). The United Kingdom also shows a negative growth rate for this period (-0.1%). The per capita quantities have increased from an average of 122 kg, with the rank headed by France (167 kg) in 1998 to an average of 138 kg topped by Germany (188 kg) in 2019. It is also interesting to note that Greece generated the lowest amount of packaging during the period, from 53 kg per capita in 1998 to 60 kg in 2019.

Recycling of non-plastic packaging (Table A.3) has also experienced an average annual increase of 2.6%, led by Ireland (7.6%). Sweden alone is the only country with a negative growth rate (-0.5%). Figure 4.3 reflects the evolution of RNPPW during the sample period. The amounts recycled have increased from 65 per capita kgs in 1998 to 103 per capita kgs in 2019. With respect to total non-plastic packaging, the recycling fraction in 1998 accounted for 83.5% in Sweden or 82.3% in Germany, whereas Ireland had the lowest proportion (18.8%). By 2019, Belgium had the highest rate at 92.1% and Sweden the lowest at 63.1%. The average recycling rate stood at 74.3% in 2019 (For more details, see Figure 5.3). Only Germany and Sweden drop their recycling rates along this period for non-plastic packaging.

The evolution of the per capita GDP reflects a standard measure of the productive sphere of any economy. If we consider the values at the beginning of our sample, we can see in Table A4 that the per capita GDP goes from 36,789€ (Luxemburg) to 13,884€ (Portugal). The average growth during 1998-2019 for our sample was

2.8%. Luxemburg maintains as the richest country in 2019 (78,681€) and Greece emerges as the country with the smallest per capita GDP (20,556€).

Regarding the evolution of the variables over time, up to 2004 we can find a process of parallel growth in both GDP and the different types of packaging around 3%. From 2004 onwards, the generation of packaging began to decrease (on average -0.9%), especially non-plastic packaging (-1%), whereas plastic packaging continued to grow at a lower rate (0.5%). This phenomenon seems to be prior to the arrival of the GR and persists until approximately 2011, when the growth rates of packaging start to rebound (1%), always led by the growth of plastics (1.4%), although already lower than the average growth of GDP (2.1%).

This disruption in packaging generation with a first period of reduction in generation and a second period of marked recovery after the GR coincides with a declining evolution in the growth rates of packaging recycling, both plastic and non-plastic The average growth rates for total recycled packaging were 7% for 1998-2003, 2.7% for 2004-2010 and 1.2% for 2011-2019. In the case of plastic, the case of Austria, Germany or Luxembourg is striking, with growth rates that have fallen to close to zero by the end of the sample. Denmark, Finland, Portugal, Spain, Sweden and the United Kingdom seem to have managed to maintain strong growth in plastic packaging recycling (between 6% and 8% in 2019). In the case of non-plastic packaging, growth rates in the period 2011-2019 have fallen to a modest 0.9% on average, with France (-0.8%) and Greece (-0.6%) being the only countries where recycling of this fraction decreases (-0.2% and -0.5%, respectively). Finland and Portugal lead the growth rates between 2011 and 2019 with 1.9%.

Based on this descriptive analysis, we can conclude that it seems to be a clear relation between packaging waste and the evolution of the European economies. The plastic packaging waste shows strong recycling growth rates, but it could be only the result of a later process of implementing their recycling technologies.

However, we should recognize ourselves that this is not the best procedure to analyze if the series are decoupling or presenting structural changes across the sample. Then, it seems to be appropriated to carry out a deeper study of the relationship between the generation of packaging, differentiating between those that are plastic and those that are not. This distinction seems particularly significant, given the different recycling rates of the two waste fractions. Moreover, while the generation of the packaging fractions seems to be more linked to the economic cycle, recycling seems to follow its own particular evolution of exhaustion in its growth, even though more pronounced with the onset of the GR.

4.2.3. Methodology

As previously mentioned, the aim of the paper is to estimate the relationships between the evolution of total and recycled packaging waste with respect to the per capita GDP for each one of the 15 selected European economies. Later, we will disaggregate these two variables into their plastic and non-plastic components. Then, the general model specification can be stated as follows:

$$\ln\ \mathrm{T_{it}} = \alpha_{1i} + \beta_{1i}\ln\left(\mathrm{GDP_{it}}\right) + u_{it} \qquad (1)$$

$$\ln\ \mathrm{R_{it}} = \alpha_{2i} + \beta_{2i}\ln\left(\mathrm{GDP_{it}}\right) + \mathrm{v_{it}} \qquad (2)$$

With i and t denoting the countries considered in the sample (i= 1, 2, …,15) and the sample period (t=1997/1998, 1999, …, 2018/2019), respectively. T_{it} and R_{it} represent the Total and the Recycled packaging waste considered; total packaging waste (PW), plastic packaging waste (PPW) and non-plastic packaging waste (NPPW).

The descriptive analysis has alerted us about the possible existence of structural breaks in the previous system of equations, especially due to the Great Recession. Consequently, we consider that the stability of this relationship may be questionable, as Alcay *et al.* (2021) show for total municipal solid waste generation. Therefore, it seems sensible to estimate the previous system allowing for the presence of some breaks. Thus, the model can be specified as follows:

$$ln\ T_{it} = a_{ij} + b_{ij} ln\ (GDP_{it}) + u_{it} \quad (3)$$

$$ln\ R_{it} = \alpha_{ij} + \beta_{ij} ln\ (GDP_{it}) + v_{it} \quad (4)$$

with $j=1, \ldots, m+1$ and m being the number of breaks that occur at periods $TB_1, \ldots, TB_m$.

We have several possibilities to estimate the system of equations composed of (3)-(4). We have opted to use the methodology proposed by Qu and Perron (2007) (QP hereafter). This methodology allows for the presence of multiple structural changes that may occur at unknown periods, whilst these breaks can affect the regression coefficients and/or the covariance matrix of the errors. Moreover, the distribution of the regressors does not have to remain constant across regimes and the method of estimation is quasi-maximum likelihood based on normal errors.

The general approach of the methodology of QP is the following. Let us consider that we have N cross-sections and the sample size is of dimension T. Then, let the vector y_t be the one that includes the endogenous variables of the system, in such a way that $y_t= (y_{1t}, \ldots y_{nt})$. Similarly, let z_t be the (qx1) vector that contains the regressors: $z_t= (z_{1t},\ldots, z_{qt})'$. We should assume that the variables included in y_t an z_t do not exhibit unit roots. The selection matrix "S" is of dimension $nq\ x\ p$ with full column rank, where p is the total number of parameters. It involves elements that take the values 0 and 1 indicating which regressors appear in each equation. The total number of structural changes in the system is m and the break dates are denoted by the m vector $M = (TB_1; \ldots; TB_m)$. The subscript j indexes a regime $(j =1, \ldots, m + 1)$, the subscript t indexes the temporal observation $(t =1, \ldots T)$, and the subscript e indexes the equation $(e =1, \ldots, f)$ to which a scalar dependent variable y_{it} is related.

The general model proposed is of the form:

$$y_t = z_t' S \beta_j + u_t \tag{5}$$

with u_t having mean 0 and covariance matrix Σ_j for $T_{j-1}+1 \leq t \leq T_j$. If we compare equation (5) with the system (3)-(4), it is clear that $y_{it} = \{ln\ T_{it},\ ln\ R_{it}\}$ and $z_t = \{ln\ (GDP_{it})\}$.

To determine the number of breaks in the system, we have used the UD_{max}LRT and WD_{max} LRT statistics to test whether at least one break is present. When the tests reject it, the test SEQ_t (ℓ +1 | ℓ) is sequentially applied for ℓ=1, 2 until the test fails to reject the null hypothesis of no additional structural breaks. Following critical values derived from response surface regressions, the tests offer evidence of the presence of two breaks in the system of equations for each country.

The next Section present the results of the application of the QP methodology for the countries included in our sample. Previous to the analysis of the results, we should note that, as previously mentioned, the use of this methodology requires the variables included in the system to be non-integrated. Then, we should verify in a first step that the variables are stationary. For this purpose, we have employed both the Dickey Fuller-GLS statistic, proposed by Elliott *et al.* (1996), and the statistics developed in Carrion-i-Silvestre *et al.* (2009), which allows the trend function of the variables to present structural breaks at unspecified periods. The number of lags has been included by considering the statistic MIC proposed in Ng and Perron (2001). The results of the tests are reported in Tables A5-A11 in the Appendix 4. As we can observe, we can reject the presence of unit roots in the series once some structural changes are considered. Consequently, the QP methodology can be used in our framework.

Finally, we consider of interest to analyze whether the COVID-19 has generated a change in the habits of the European consumers. Then, we have followed

Salkever (1975) in order to measure this possible COVID-19 effect. Following this author, we have estimated the following system:

$$ln\ (TP_k)_{it} = a_{1ik3} + \beta_{1ik3}\ ln(GDP_{it}) + \delta_{1ik3}D_t + u_{1ikt} \tag{6}$$

$$ln\ (RP_k)_{it} = a_{2ik3} + \beta_{2ik3} ln\ (GDP_{it}) + \delta_{1ik3}D_t + u_{2ikt} \tag{7}$$

We should note that t now takes values up to 2020 (t=1997/1998, 1999, …, 2019, 2020) and Dt is a dummy variable that takes the value 1 for the period 2020 and 0 otherwise. with value 1 for 2020 (and zero elsewhere). Under this modelling, $\hat{\delta}$ means the prediction error for the year 2020 using the information available until 2019 and $\frac{\hat{\delta}}{\widehat{\sigma_{\hat{\delta}}}}$ analyze the null hypothesis of absence of (post-sample) structural change, providing very useful information on the impact of the COVID-19 in the evolution of the Total/Recycled packaging waste.

4.3. Results

Tables 1-3 present the results of applying the methodology discussed in section 2. Table 1 reflects the results for the case of total packaging, whilst Table 2 and Table 3 focus on the cases of plastic packaging waste and non-plastic packaging waste, respectively.

The first interesting result that emerges from the analysis of these tables is the presence of two structural breaks. We can see that the statistics WDmax LTR allows us to reject the null hypothesis of non-structural breaks for the three-system considered (total packaging, plastic and non-plastic packaging waste). Furthermore, we can also observe that the sequential procedure estimates the presence of two breaks for the three systems. The first one is located around 2004, whilst the second one appears around 2011. This latter break is clearly related to the turning point after the Great Recession that involved the beginning of the recovery period of the European economies. Consequently, these initial results reinforce our idea that the GR plays a key role in the relationship between the evolution of the economies, the generation of packages and their recycling.

We now analyze the estimations obtained for the three considered cases, beginning by the case of the total packaging, whose results are presented in Table 1. The different elasticities for 1998, 2007 and 2019 can be found in Figures 1.1 to 1.3 for TPW and in Figures 1.4 to 1.6 for RPW respectively. If we compare the results at the beginning and at the end of the sample, we can see clear reduction of the per capita GDP elasticities, which can be interpreted as evidence in favor of relative decoupling. This decoupling was more intense during the GR as a consequence of the severe drop in packaging waste quantities (-0.9% yearly in our sample average during 2004-2010). Ireland is the only exception with a more intense decoupling relationship after the GR. By contrast, we can observe that this decoupling procedure has weakened with the recovery of the economies and the per capita GDP elasticities raises in the last segment of the sample (See Belgium, Denmark, Germany, Greece, Italy, Luxembourg, the Netherlands Spain, Sweden, or United Kingdom). If we now focus on packaging recycling, we should expect the RPW elasticities to take higher values than TPW ones, and overcoming 1, so that the fraction of non-recycled packaging will decrease over time. However, we can see that the estimated elasticities are mostly positive (France is the exception), but the estimations are higher than 1 just for the cases of Denmark, Finland, Greece, Luxembourg and the Netherlands.

The case of the plastic packaging waste also offers very interesting results, as deduced from the analysis of Table 2. The different elasticities for 1998, 2007 and 2019 can be found in Figures 2.1 to 2.3 for TPPW and in Figures 2.4 to 2.6 for RPPW respectively We can observe that the degree of coupling does not reduce despite the GR. For instance, we can see that the per capita GDP elasticities at the end of the sample are all positive and they are greater than 1 for 6 countries. The case of the recycled plastic packaging shows very large elasticities too. This is not a quite surprising result if we take into account that people began recycling plastics very recently. We should note that only a 10% of plastic packaging were recycled at the beginning of the sample, whilst the current value has risen to remarkable 40%. Germany and Austria exhibit a complete standstill in the expansion of recycling, with Austria even showing declines. The remaining countries are still showing vigorous growth, although this is gradually decelerating over time as can be shown in Figure 5.2.

The last considered system is the one that considers the non-plastic packaging waste. The results for this case are reflected in Table 3, where we can see a relative decoupling with respect to the per capita GDP, except for SWE, GRE and IRE. Similarly, to the other models, the elasticities for 1998, 2007 and 2019 can be found in Figures 3.1 to 3.3 for TNPPW and in Figures 3.4 to 3.6 for RNPPW respectively. The growth in total non-plastic package quantities is similar to the growth of the recycling ones, so it is not possible to increase the recycled fraction proportion of the total. Recycling growth rates for these types of packaging are weaker, maybe due to the higher recycling rates for many countries (70-80%). At the end of the sample, Belgium, Finland, Ireland, Luxembourg, Netherlands, Portugal and Spain presented a higher elasticity for RNPPW than for TNPPW. In some way regarding also Figure 5.3, this reflects that the recycled fraction of this type of packages appears to be stabilizing.

Finally, it seems to be interesting to analyze whether the COVID-19 pandemic has modified the results presented in the pattern of behavior of the European agents. To that end, Table 4 presents the estimation of the parameter d and its corresponding standard deviation. Following Salkever (1975), the ratio of this values is equivalent to test the null hypothesis of the absence of a post-sample structural break.

The first result that emerges for the analysis of this tables is that we can reject this null hypothesis for all the countries, with the exception of Ireland. Then, we should conclude that there exists a structural break a consequence of the COVID-19 pandemic. Luxembourg shows a reduction in the different packaging fractions for both generated and recycled packaging waste, while Spain shows just the opposite, all showing a positive impact. Moreover, the effect in Spain is more positive for generation than for recycling. However, Spain is the sole country that shows an improvement in recycling for both plastic and non-plastic packaging. Portugal shows an improvement only for non-plastic packaging and Belgium reports positive effects only for plastic packaging recycling. Austria, Denmark, France, Luxembourg and Sweden show a decrease in plastics and/or non-plastics recycling during 2020. Another finding present in Belgium, France, Germany, Portugal, and Spain is a positive effect on the use of plastic packaging. suggesting that the pandemic is linked to a more intense use of plastic. In conclusion, the pandemic has led in most countries to an increase in packaging, especially plastic, while recycling has decreased or, in the cases where it has increased, it has increased less than the increase in packaging, so that, as a general rule, waste recovery has suffered.

4.4. Discussion

The results presented in the previous are a combination of positive and negative news so far, the effect of packaging waste management on the sustainability of the economies. On the one hand, it is encouraging to note that waste prevention policies have resulted in achieving a relative decoupling for total packaging with respect to GDP. However, this good news fundamentally comes from the non-plastic packaging side, whilst plastic packaging waste shows a very persistent and unpleasant growth. Even worse, many countries kept this growth during the GR.

It is also positive to note that recycled plastic packaging waste has been growing very strongly for last 20 years, moving from a 10% at the beginning of the sample to around 40%. Nevertheless, it should be also notice that the growth has gradually decelerated in the last segment of the sample. The recycling of the non-plastic packaging is growing slightly, and it is stabilizing around 70-80% recycling rates. Following Rigamonti (2018), although these materials have theoretical recycling rates close to 100% in practice the processes are not being efficient from a collection and separation point of view, so that the process inputs result impure or directly discard a significant fraction that is not recycled. This situation is more complicated for plastics, with lower theoretical recycling rates. According to Hahladakis and Iacovidou (2019) and Antonopoulos *et al.* (2021), to further increase plastic recycling, it would be necessary to revise the packaging sorting system, its collection and the recycling plant processes to reduce impurities, bottlenecks and ensure that less and less of the collected plastics are landfilled. This should raise concern about the laxity in the application and progressive improvement of public policies to promote the collection, sorting and recycling of packaging waste.

The substitution of heavy packaging with lighter plastic ones could be playing a dangerous trick towards decoupling in mass as Tsiamis *et al.* (2018) point out, at the cost of making packaging harder to be recycled. The replacement of plastic with reusable or biodegradable packaging may be one of the most successful public policies to reduce the growth of plastic use, however, there is still pending a tremendous amount of research on new varieties of compostable materials and on the actual sustainability of each of the alternatives to plastic packaging (Bradley and Corsini, 2023).

The link between the economic cycle and packaging waste management appears to be volatile and vulnerable to the impact of major events such as the GR or COVID-19. Given the results, both events have boosted the use of plastic packaging (or have not reduced it) while they have tended to decelerate packaging recycling growth or sometimes reduce it. This fragility should warn us about the need to invigorate environmental policies at European level to be a priority, even in times of crisis.

Heterogeneity between countries is also a common characteristic in environmental issues, considering the European decentralization in the application, monitoring and enforcement of waste generation and recycling regulations. In this sense, we see how initial leaders in recycling, Austria and Germany, have lost their position as leaders in the recycling of packaging: Austria and Germany, are no longer so, stagnating or even receding. The current leaders in this field are Belgium, Denmark, the Netherlands and Sweden, with the appearance of Spain in the recycling of plastic packaging.

4.5. Conclusions

This paper analyzes the relationship between the economic cycle and the generation of packaging waste for a sample of EU countries. To that end, we have considered total packaging waste and recycled packaging waste and we have disaggregated these variables into plastic and non-plastic packaging waste, in order to better appreciate the evolution of plastics, given their relevance in waste management.

Our main interest is to model the relationship between these waste variables and the per capita GDP in order to estimate their elasticities and interpret them in terms of decoupling, in that these estimations offer valuable information to study the effect of waste on the sustainability of the economies. In order to take into account, the possible presence of changes in the elasticities, we have considered the possible presence of structural breaks. The evidence is quite favorable to this hypothesis, and we have found two breaks, the second one clearly related to the recovery period that occurred after the Great Recession. Finally, we have taken advantage of the estimations of the different waste-GDP relationships in order to analyze the effect of COVID-19 and to know whether it has positively or negatively affected sustainability.

The results of our analysis show a relative decoupling between total packaging waste and per capita GDP. We can also observe a slight increase in the fraction of recycled packaging, although the increase is slowdown after the Great Recession. If we disaggregate by type of packaging waste, we can observe that plastics has grown across the sample. Moreover, we have also observed that this growth was very persistent, in the sense that packaging waste maintaining its growth despite the GR. The growth of plastic packaging recycling has been particularly high during the first years of the 21st century, although it is beginning to show a certain attrition, still at proportions of 40-50% of recycled packaging. Non-recycled packaging has shown a more stagnant behavior, with a situation of relative decoupling with respect to the per capita GDP. The recycling of non-recycled packaging is increasing very slightly, at around 60-70% of the recycled proportion.

Our results have also allowed us to analyze the effect of the COVID-19 on packaging waste. They do not appear to be much more optimistic about packaging recycling, since the countries included in our sample have experienced an increase in the generation of plastic packaging during 2020 and, apart from exceptional cases, a decline in recycling. The results for the years after 2020 will be very important in order to determine whether the pattern of behavior of the agents has really changed or we have only observer a transitory change.

In view of the results, it is necessary to rethink a revision of the policies aimed at promoting the circular economy to make them less dependent on the economic cycle and on the conjunctural disposition of the EU member countries. This latter point seems us crucial to us in the believe that environmental policies should be a fundamental pillar of the European policies. To increase both the prevention of packaging and its subsequent recycling, it is necessary to review both regulatory policies on packaging design and materials, as well as policies aimed at the separate collection, sorting and recycling of the different types of packaging.

Greater harmonization and simplification are necessary in order to reach the 2025 and 2030 packaging recycling targets so that packaging that is not subsequently recyclable can be avoided, while facilitating the separation of packaging that is recyclable to prevent it from being discarded from the economic cycle.

4.6. Figures and tables

Table 1. Estimation of the system composed by (3)-(4) equations. Case: Total packaging waste.

Country	Sample	WD_{max} LRT	Model	a_0	β_0	TB_1	a_1	β_1	TB_2	a_2	β_2
AUS	97-19	54.50	*TPW*	5.95*	-0.10	2005	-0.64	0.54*	2012	-1.06*	0.58*
				(0.84)	(0.08)		(0.91)	(0.09)		(0.37)	(0.04)
			RPW	4.98*	-0.05		0.14	0.43*		-0.17	0.46*
				(1.04)	(0.1)		(1.4)	(0.14)		(0.69)	(0.07)
BEL	97-19	67.14	*TPW*	2.88	0.21	2003	5.55*	-0.05	2011	3.21*	0.18*
				(1.44)	(0.14)		(0.55)	(0.05)		(0.35)	(0.03)
			RPW	-3.38*	0.79*		0.79	0.39*		-0.58	0.52*
				(1.15)	(0.12)		(0.95)	(0.09)		(0.57)	(0.05)
DEN	97-19	37.54	*TPW*	7.11*	-0.20	2003	11.19*	-0.59*	2010	-3.77	0.84*
				(1.96)	(0.19)		(2.70)	(0.27)		(3.95)	(0.38)
			RPW	-9.57*	1.39*		0.97	0.35*		-6.78	1.10*
				(3.27)	(0.32)		(1.19)	(0.12)		(4.08)	(0.39)
FIN	97-19	57.76	*TPW*	-3.45	0.80	2004	2.85*	0.20	2011	5.74*	-0.08
				(4.09)	(0.41)		(1.05)	(0.10)		(0.75)	(0.07)
			RPW	-6.54*	1.03*		-15.71*	1.94*		-8.33*	1.23*
				(1.77)	(0.18)		(3.04)	(0.30)		(0.93)	(0.09)
FRA	97-19	38.70	*TPW*	2.13	0.32*	2003	5.74*	-0.04	2011	4.96*	0.03
				(1.30)	(0.13)		(0.99)	(0.10)		(1.57)	(0.15)
			RPW	-3.59*	0.81*		-6.04*	1.05*		5.37*	-0.05
				(1.43)	(0.14)		(1.48)	(0.15)		(2.07)	(0.20)
GER	97-19	50.84	*TPW*	-0.99	0.62*	2004	1.26	0.39*	2012	-0.03	0.52*
				(0.70)	(0.07)		(0.83)	(0.08)		(0.97)	(0.09)
			RPW	5.16*	-0.02		-1.59	0.63*		4.85	0.02
				(1.88)	(0.19)		(1.32)	(0.13)		(2.82)	(0.27)
GRE	97-19	61.89	*TPW*	-7.17*	1.19*	2003	3.18	0.13	2011	-9.37*	1.38*
				(2.38)	(0.25)		(1.72)	(0.17)		(2.94)	(0.30)
			RPW	-4.42*	0.80*		-11.14	1.49*		-20.16*	2.43*
				(1.41)	(0.15)		(6.96)	(0.69)		(5.67)	(0.57)
IRE	97-19	128.69	*TPW*	-1.33	0.66*	2003	-3.23*	0.83*	2010	2.37*	0.28*
				(0.77)	(0.08)		(0.75)	(0.07)		(0.58)	(0.05)
			RPW	-22.89*	2.64*		-7.29	1.17*		3.87*	0.10
				(3.67)	(0.37)		(4.24)	(0.40)		(0.58)	(0.05)
ITA	97-19	110.89	*TPW*	-3.08	0.83*	2003	5.53*	-0.02	2011	-3.41*	0.85*
				(3.38)	(0.34)		(2.32)	(0.23)		(1.12)	(0.11)
			RPW	-136.52*	14.05*		-7.33*	1.19*		-5.21*	0.99*
				(59.53)	(5.93)		(1.87)	(0.18)		(1.23)	(0.12)
LUX	97-19	44.13	*TPW*	4.64*	0.05	2003	4.31*	0.09	2011	-4.19*	0.85*
				(0.65)	(0.06)		(1.34)	(0.12)		(1.14)	(0.10)
			RPW	-8.38*	1.20*		2.09	0.25		-15.61*	1.83*
				(3.26)	(0.31)		(1.58)	(0.14)		(2.98)	(0.26)
NET	97-19	36.23	*TPW*	-1.74	0.68*	2004	16.67*	-1.11*	2011	3.01	0.20
				(3.07)	(0.30)		(3.39)	(0.33)		(3.43)	(0.33)
			RPW	-0.77	0.54*		3.34*	0.14		-6.97	1.12*
				(1.33)	(0.13)		(0.91)	(0.09)		(3.63)	(0.35)
POR	99-19	58.10	*TPW*	-6.31*	1.15*	2004	-6.82*	1.20*	2011	-3.08*	0.82*
				(1.90)	(0.20)		(1.70)	(0.17)		(0.63)	(0.06)
			RPW	-11.20*	1.55*		-30.86*	3.57*		-4.42*	0.90*
				(2.19)	(0.23)		(3.73)	(0.38)		(1.49)	(0.15)
SPA	97-19	58.17	*TPW*	1.73	0.34*	2005	-0.23	0.53*	2012	-2.90*	0.78*
				(1.28)	(0.13)		(2.23)	(0.22)		(0.46)	(0.05)
			RPW	-8.77*	1.32*		-5.74*	1.02*		-5.42*	0.99*
				(1.06)	(0.11)		(1.12)	(0.11)		(1.06)	(0.10)
SWE	97-19	42.88	*TPW*	-10.03*	1.47*	2005	24.04*	-1.84*	2012	-16.30*	2.02*
				(4.67)	(0.46)		(9.21)	(0.90)		(2.26)	(0.22)
			RPW	-3.72	0.80		5.33	-0.09		-4.03	0.81
				(4.68)	(0.46)		(5.99)	(0.58)		(5.59)	(0.54)
UKG	97-18	50.28	*TPW*	6.24*	-0.11	2004	3.55*	0.16*	2011	1.19	0.38*
				(1.83)	(0.18)		(0.75)	(0.07)		(1.94)	(0.19)
			RPW	-18.61*	2.27*		-13.18	1.74		-1.01	0.55*
				(2.76)	(0.27)		(10.18)	(1.07)		(1.40)	(0.14)

This table presents the results of the estimation of equations (3) and (4), with TBj (j=1,2) being the estimated periods when the break appears. WDmax LRT statistic tests for the null hypothesis of no structural breaks, rejecting it for all the considered countries (CV at 5% level is 21.37). Sample covers the period 19XX-20XX. Robust standard deviations are presented below the estimated parameter.

Table 2. Estimation of the system composed by (3)-(4) equations. Case: Plastic packaging waste.

Country	Sample	WD_{max} LRT	Model	α_0	β_0	TB_1	α_1	β_1	TB_2	α_2	β_2
AUS	97-19	64.13	*TPPW*	-2.84	0.60*	2003	-4.73*	0.79*	2012	2.52	0.10
				(2.19)	(0.22)		(0.77)	(0.08)		(2.12)	(0.20)
			RPPW	-21.94*	2.36*		-10.24*	1.21*		9.02*	-0.63*
				(5.81)	(0.58)		(0.91)	(0.09)		(2.79)	(0.27)
BEL	97-19	58.15	*TPPW*	-5.00*	0.81*	2003	-0.88	0.41*	2012	0.09	0.32*
				(0.59)	(0.06)		(0.57)	(0.06)		(0.66)	(0.06)
			RPPW	-12.31*	1.41*		-13.23*	1.52*		-9.04*	1.11*
				(2.35)	(0.23)		(2.68)	(0.26)		(1.76)	(0.17)
DEN	97-19	30.73	*TPPW*	13.82*	-1.03*	2003	1.96	0.15	2012	-8.82*	1.18*
				(1.59)	(0.16)		(2.53)	(0.25)		(2.18)	(0.21)
			RPPW	-39.01*	4.00*		-16.42*	1.78*		-27.81*	2.88*
				(4.73)	(0.47)		(2.52)	(0.25)		(3.89)	(0.37)
FIN	97-19	48.77	*TPPW*	5.31*	-0.25*	2004	-6.73*	0.95*	2011	-7.85*	1.06*
				(0.44)	(0.04)		(2.48)	(0.24)		(1.05)	(0.10)
			RPPW	-12.75*	1.36*		-38.11*	3.85*		-41.99*	4.23*
				(1.67)	(0.17)		(4.00)	(0.40)		(8.52)	(0.83)
FRA	97-19	79.00	*TPPW*	-2.41*	0.58*	2003	3.39*	0.01	2011	-6.91*	1.01*
				(0.50)	(0.05)		(1.60)	(0.16)		(1.52)	(0.15)
			RPPW	-40.52*	4.18*		-20.03*	2.16*		-14.77*	1.64*
				(1.30)	(0.13)		(1.44)	(0.14)		(2.52)	(0.25)
GER	97-19	88.36	*TPPW*	-14.85*	1.79*	2004	-9.03*	1.22*	2011	-3.40*	0.67*
				(0.97)	(0.10)		(1.78)	(0.17)		(0.42)	(0.04)
			RPPW	-4.59*	0.70*		-19.77*	2.19*		0.16	0.26
				(1.57)	(0.16)		(4.02)	(0.39)		(2.45)	(0.23)
GRE	97-19	70.77	*TPPW*	-9.42*	1.29*	2003	11.93*	-0.87	2010	-12.95*	1.60*
				(2.75)	(0.28)		(4.40)	(0.44)		(3.14)	(0.32)
			RPPW	-8.50	0.84		-67.41*	6.83*		-16.96	1.90
				(4.81)	(0.49)		(20.85)	(2.09)		(11.96)	(1.14)
IRE	97-19	87.54	*TPPW*	-6.77*	1.02*	2005	-7.22*	1.03*	2012	-16.25*	1.94*
				(1.86)	(0.19)		(2.16)	(0.22)		(4.40)	(0.45)
			RPPW	-17.69*	1.79*		26.67*	-2.53*		-36.50*	3.89*
				(7.24)	(0.75)		(10.56)	(1.04)		(3.09)	(0.31)
ITA	97-19	66.72	*TPPW*	-2.56*	0.61*	2003	-0.98	0.45*	2011	-5.79*	0.92*
				(0.40)	(0.04)		(0.57)	(0.06)		(0.32)	(0.03)
			RPPW	-52.82*	5.45*		-21.88*	2.39*		-20.30*	2.25*
				(4.36)	(0.44)		(4.44)	(0.44)		(2.11)	(0.20)
LUX	97-19	84.06	*TPPW*	3.07*	0.00	2003	2.25	0.14	2012	13.16*	-0.83
				(0.04)	(0.00)		(2.70)	(0.24)		(5.69)	(0.51)
			RPPW	-56.96*	5.49*		-3.70	0.57		18.91*	-1.44*
				(9.09)	(0.86)		(7.72)	(0.70)		(3.31)	(0.30)
NET	97-19	67.92	*TPPW*	6.74*	-0.32	2005	21.13*	-1.71*	2012	-4.20*	0.72*
				(3.38)	(0.33)		(7.63)	(0.73)		(1.41)	(0.13)
			RPPW	-9.69*	1.13*		-19.85*	2.13*		-15.43*	1.72*
				(2.42)	(0.24)		(1.51)	(0.15)		(3.29)	(0.31)
POR	99-19	46.95	*TPPW*	-7.41*	1.11*	2003	-1.53	0.51*	2010	-5.77*	0.94*
				(1.27)	(0.13)		(1.61)	(0.16)		(0.59)	(0.06)
			RPPW	-61.29*	6.40*		-41.57*	4.39*		-12.63	1.52*
				(9.30)	(0.96)		(8.73)	(0.89)		(6.64)	(0.66)
SPA	97-19	69.62	*TPPW*	-3.96*	0.75*	2005	-2.83*	0.63*	2012	-9.30*	1.26*
				(0.45)	(0.05)		(1.35)	(0.13)		(1.21)	(0.12)
			RPPW	-33.90*	3.61*		-3.90	0.60		-25.53*	2.78*
				(4.35)	(0.44)		(3.79)	(0.37)		(3.78)	(0.37)
SWE	97-19	45.13	*TPPW*	-2.81	0.56	2005	-3.55*	0.64*	2012	-1.71	0.47*
				(1.68)	(0.16)		(1.61)	(0.16)		(1.91)	(0.18)
			RPPW	-13.26	1.43		-10.09	1.17		-26.55*	2.77*
				(8.33)	(0.82)		(9.50)	(0.92)		(12.61)	(1.20)
UKG	97-18	49.56	*TPPW*	4.06*	-0.07	2005	-5.76	0.91	2012	13.43*	-0.96*
				(1.32)	(0.13)		(9.37)	(0.91)		(3.75)	(0.36)
			RPPW	-40.42*	4.18*		-22.73	2.43		-36.21*	3.76*
				(8.57)	(0.85)		(24.51)	(2.39)		(3.70)	(0.36)

This table presents the results of the estimation of equations (3) and (4), with TBj (j=1,2) being the estimated periods when the break appears. WDmax LRT statistic tests for the null hypothesis of no structural breaks, rejecting it for all the considered countries (CV at 5% level is 21.37). Sample covers the period 19XX-20XX. Robust standard deviations are presented below the estimated parameter.

Table 3. Estimation of the equations (3)-(4) system. Case: Non-Plastic packaging waste.

Country	Sample	WD_{max} LRT	Model	a_o	β_o	TB_1	a_1	β_1	TB_2	a_2	β_2
AUS	97-19	51.11	*TNPPW*	7.44*	-0.27*	2005	0.34	0.42*	2012	-2.71*	0.71*
				(0.87)	(0.09)		(0.99)	(0.10)		(0.67)	(0.06)
			RNPPW	6.78*	-0.23*		1.03	0.33*		-1.65*	0.59*
				(1.05)	(0.10)		(1.41)	(0.14)		(0.66)	(0.06)
BEL	97-19	61.73	*TNPPW*	3.86*	0.09	2011	6.49*	-0.16*	2011	3.43*	0.14*
				(1.61)	(0.16)		(0.68)	(0.07)		(0.55)	(0.05)
			RNPPW	-3.01*	0.74*		1.91*	0.27*		-0.09	0.46*
				(1.07)	(0.11)		(0.93)	(0.09)		(0.64)	(0.06)
DEN	97-19	41.53	*TNPPW*	4.83	0.00	2002	2.91	0.20	2008	-3.73	0.82*
				(3.10)	(0.31)		(2.59)	(0.25)		(4.08)	(0.39)
			RNPPW	-11.13*	1.55*		1.28	0.32*		-2.47	0.68*
				(3.28)	(0.33)		(0.93)	(0.09)		(2.68)	(0.26)
FIN	97-19	78.98	*TNPPW*	-0.59	0.48*	2002	-5.96	1.04	2009	5.36	-0.07
				(0.43)	(0.04)		(6.53)	(0.64)		(2.41)	(0.23)
			RNPPW	-5.61*	0.93*		-14.70*	1.83*		-9.02*	1.29*
				(2.26)	(0.23)		(1.28)	(0.13)		(2.64)	(0.26)
FRA	97-19	38.41	*TNPPW*	2.42	0.27	2003	5.67*	-0.05	2011	6.89*	-0.18
				(1.42)	(0.15)		(0.90)	(0.09)		(1.67)	(0.16)
			RNPPW	-2.41	0.68*		-5.40*	0.99*		6.63*	-0.18
				(1.51)	(0.15)		(1.54)	(0.15)		(2.10)	(0.20)
GER	97-19	55.54	*TNPPW*	-0.12	0.52*	2003	3.43*	0.16	2012	-0.74	0.57*
				(0.94)	(0.09)		(1.19)	(0.12)		(1.84)	(0.17)
			RNPPW	3.14*	0.17		1.15	0.36*		4.85	0.00
				(1.42)	(0.14)		(1.56)	(0.15)		(2.86)	(0.27)
GRE	97-19	54.09	*TNPPW*	-7.12*	1.16*	2003	0.88	0.34	2009	-9.73*	1.39*
				(2.34)	(0.24)		(1.95)	(0.19)		(1.27)	(0.13)
			RNPPW	-4.44*	0.80*		-10.28*	1.39*		-7.52	1.13*
				(1.40)	(0.15)		(3.95)	(0.39)		(4.77)	(0.48)
IRE	97-19	122.12	*TNPPW*	-7.12*	1.16*	2003	0.27	0.40*	2011	-8.14*	1.23*
				(2.34)	(0.24)		(1.65)	(0.16)		(3.00)	(0.30)
			RNPPW	-4.45*	0.80*		-9.05	1.27*		-17.87*	2.18*
				(1.40)	(0.15)		(5.43)	(0.54)		(6.17)	(0.62)
ITA	97-19	118.40	*TNPPW*	-3.79	0.89*	2003	6.37*	-0.12	2011	-3.47*	0.84*
				(4.17)	(0.42)		(3.03)	(0.30)		(1.29)	(0.13)
			RNPPW	-172.84*	17.66*		-6.25*	1.07*		-3.74*	0.83*
				(81.60)	(8.13)		(1.66)	(0.16)		(1.38)	(0.13)
LUX	97-19	48.88	*TNPPW*	4.43*	0.06	2003	4.24*	0.08	2011	-8.63*	1.22*
				(0.74)	(0.07)		(1.30)	(0.12)		(1.31)	(0.12)
			RNPPW	-6.38	1.01*		2.31*	0.22*		-19.48*	2.17*
				(3.96)	(0.38)		(0.98)	(0.09)		(3.83)	(0.34)
NET	97-19	36.65	*TNPPW*	-4.19*	0.91*	2005	16.21*	-1.08	2011	4.22	0.07
				(2.11)	(0.21)		(7.32)	(0.70)		(3.78)	(0.36)
			RNPPW	-0.72	0.53*		5.96*	-0.12		-6.27	1.05*
				(0.95)	(0.09)		(2.89)	(0.28)		(4.10)	(0.39)
POR	99-19	45.37	*TNPPW*	-6.08	1.10*	2003	-9.44*	1.44*	2010	-2.58*	0.74*
				(3.96)	(0.41)		(1.11)	(0.11)		(0.81)	(0.08)
			RNPPW	-6.35*	1.04*		-32.24*	3.70*		-3.54	0.79*
				(2.76)	(0.29)		(2.16)	(0.22)		(1.90)	(0.19)
SPA	97-19	60.12	*TNPPW*	5.98*	-0.12	2003	4.84*	0.01	2011	-0.98	0.57*
				(1.96)	(0.20)		(0.40)	(0.04)		(1.14)	(0.11)
			RNPPW	-4.99*	0.92*		-9.51*	1.39*		-2.43*	0.68*
				(1.12)	(0.12)		(1.09)	(0.11)		(1.10)	(0.11)
SWE	97-19	42.61	*TNPPW*	-11.61*	1.61*	2005	28.58*	-2.30*	2012	-20.42*	2.39*
				(5.32)	(0.53)		(11.20)	(1.09)		(2.62)	(0.25)
			RNPPW	-3.44	0.76		6.40	-0.20		-1.13	0.52
				(4.40)	(0.43)		(6.13)	(0.60)		(5.88)	(0.56)
UKG	97-18	54.74	*TNPPW*	6.12*	-0.12	2004	5.04*	-0.01	2011	-2.82	0.75*
				(2.34)	(0.23)		(0.22)	(0.02)		(2.21)	(0.21)
			RNPPW	-17.55*	2.16*		-12.74	1.69		3.23*	0.13
				(2.46)	(0.24)		(9.58)	(0.94)		(1.30)	(0.13)

This table presents the results of the estimation of equations (3) and (4), with TBj (j=1,2) being the estimated periods when the break appears. WDmax LRT statistic tests for the null hypothesis of no structural breaks, rejecting it for all the considered countries (CV at 5% level is 21.37). Sample covers the period 19XX-20XX. Robust standard deviations are presented below the estimated parameter.

Table 4. Testing for COVID-19 influence on packaging waste.

Country	TPW	RPW	TPPW	RPPW	TNPW	RNPW
AUS	0.00	-0.04*	-0.01	-0.26*	0.01	-0.01*
	(0.00)	(0.00)	(0.01)	(0.02)	(0.00)	(0.00)
BEL	0.05*	-0.00	0.05*	0.04*	0.06*	-0.01
	(0.00)	(0.00)	(0.00)	(0.02)	(0.01)	(0.01)
DEN	0.02	-0.12*	-0.03	-0.55*	0.03	-0.05
	(0.03)	(0.05)	(0.03)	(0.05)	(0.04)	(0.04)
FRA	-0.01	-0.12*	0.04*	-0.18*	-0.03*	-0.12*
	(0.01)	(0.02)	(0.01)	(0.02)	(0.01)	(0.02)
GER	-0.01	0.00	0.02*	0.00	-0.01*	0.00
	(0.01)	(0.03)	(0.00)	(0.03)	(0.01)	(0.03)
IRE	0.01	-0.01	-0.04	-0.07	0.03	0.00
	(0.02)	(0.02)	(0.06)	(0.07)	(0.02)	(0.01)
LUX	-0.08*	-0.06*	-0.14*	-0.08*	-0.07*	-0.06*
	(0.01)	(0.00)	(0.03)	(0.02)	(0.01)	(0.01)
POR	0.05*	0.06*	0.05*	-0.03	0.05*	0.07*
	(0.01)	(0.02)	(0.01)	(0.06)	(0.01)	(0.02)
SPA	0.11*	0.05*	0.15*	0.12*	0.09*	0.03*
	(0.00)	(0.01)	(0.01)	(0.03)	(0.01)	(0.01)
SWE	-0.02	-0.09	-0.00	-0.46*	-0.02	-0.03
	(0.03)	(0.05)	(0.01)	(0.03)	(0.03)	(0.05)

This table presents the estimations of the parameter δ for the system composed by equations (5)-(6), The corresponding robust standard deviations into parenthesis.

*Means rejection of the null hypothesis of non-significance for a 5% significance level.

TPW: Total packaging waste
RPW: Recycled packaging waste
TPPW: Total plastic packaging waste
RPPW: Recycled plastic packaging waste
TNPW: Total non-plastic packaging waste
RNPW: Recycled non-plastic packaging waste

Figure 1.1. Elasticities for GDP and Total Packaging 1998.

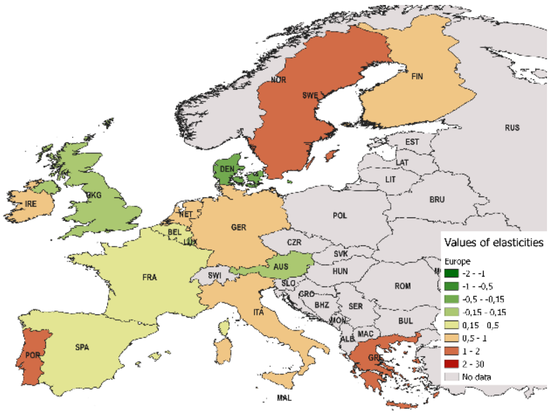


Figure 1.2. Elasticities for GDP and Total Packaging 2008.

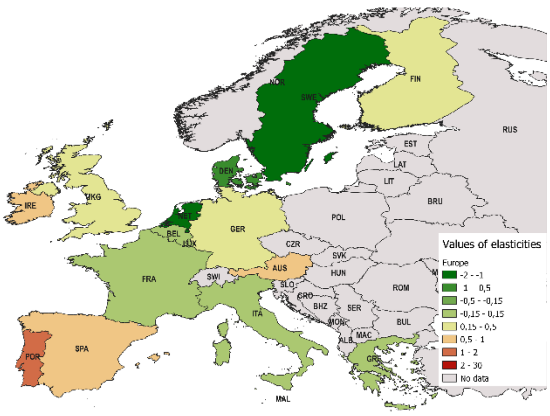


Figure 1.3. Elasticities for GDP and Total Packaging 2018.

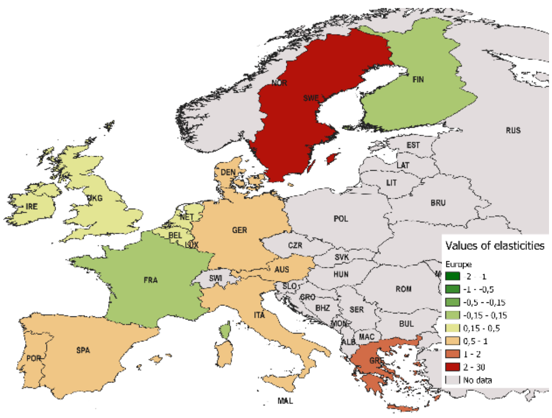


Figure 1.4. Elasticities for GDP and Recycled Packaging 1998.

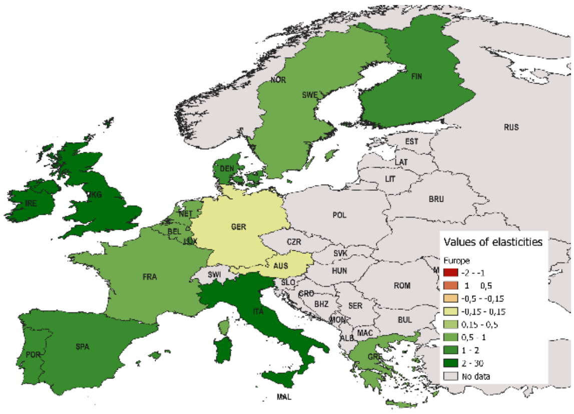


Figure 1.5. Elasticities for GDP and Recycled Packaging 2008.

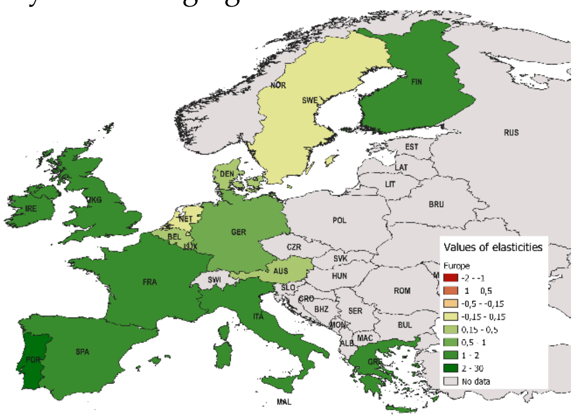


Figure 1.6. Elasticities for GDP and Recycled Packaging 2018.

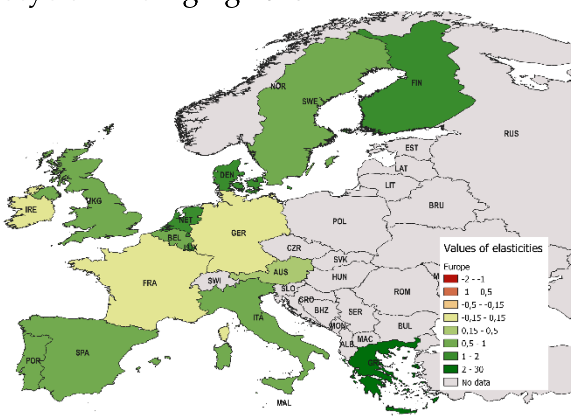


Figure 2.1. Elasticities for GDP and Total Plastic Packaging 1998.

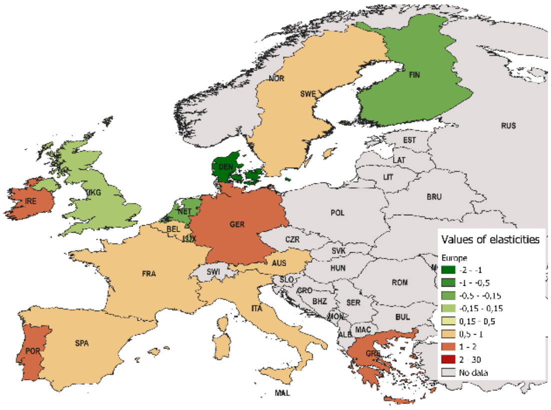


Figure 2.4. Elasticities for GDP and Recycled Plastics 1998.

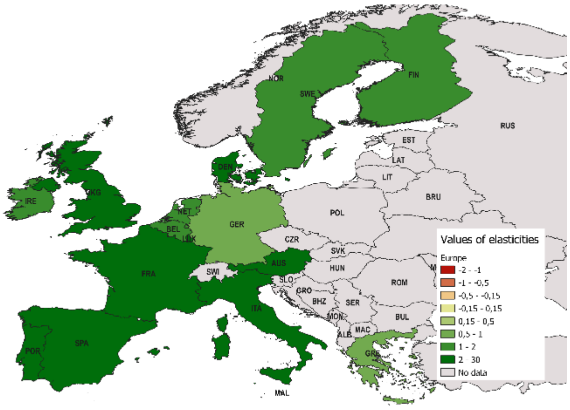


Figure 2.2. Elasticities for GDP and Total Plastic Packaging 2008.

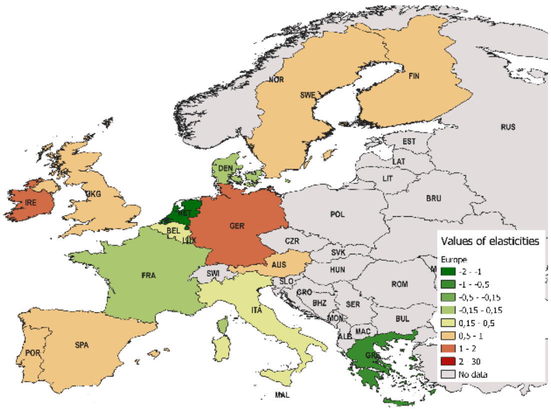


Figure 2.5. Elasticities for GDP and Recycled Plastics 2008.

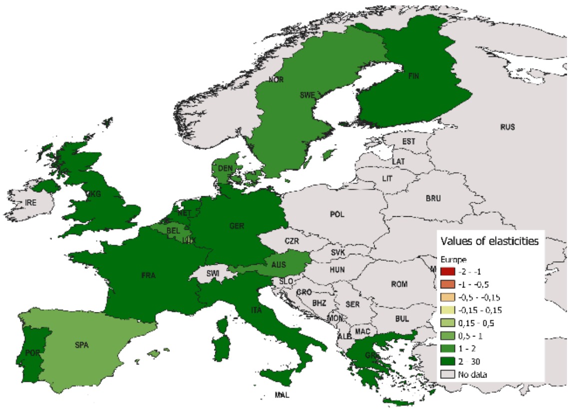


Figure 2.3. Elasticities for GDP and Total PlasticPackaging 2018.

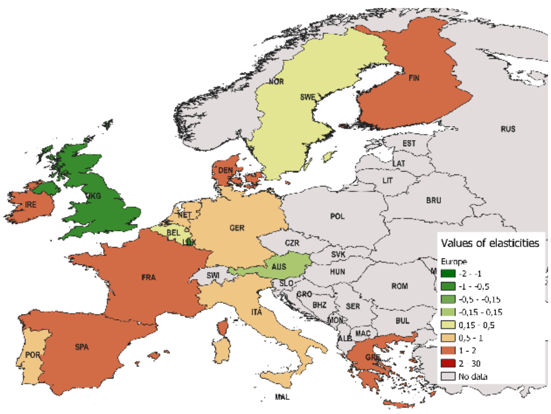


Figure 2.6. Elasticities for GDP and Recycled Plastics 2018.

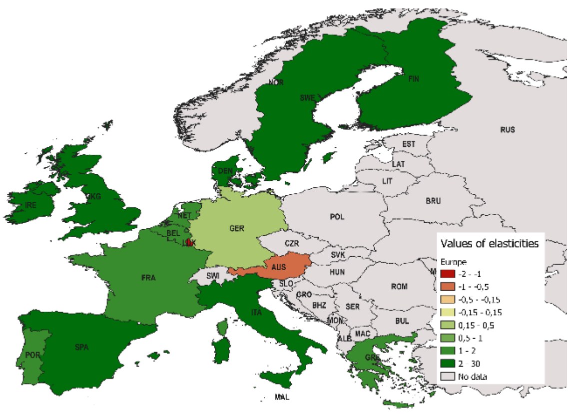


Figure 3.1. Elasticities for GDP and Total Non-Plastic Packaging 1998.

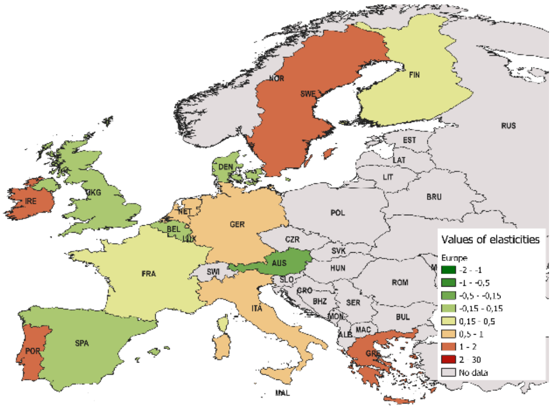


Figure 3.2. Elasticities for GDP and Total Non-Plastic Packaging 2008.

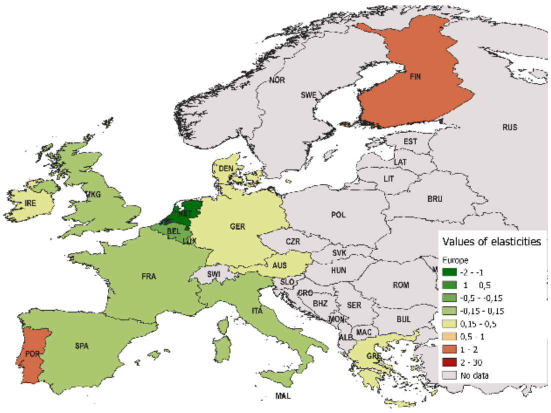


Figure 3.3. Elasticities for GDP and Total Non-Plastic Packaging 2018.

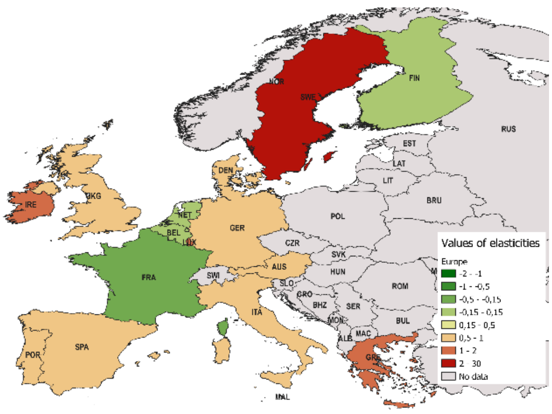


Figure 3.4. Elasticities for GDP and Recycled Non-Plastics 1998.

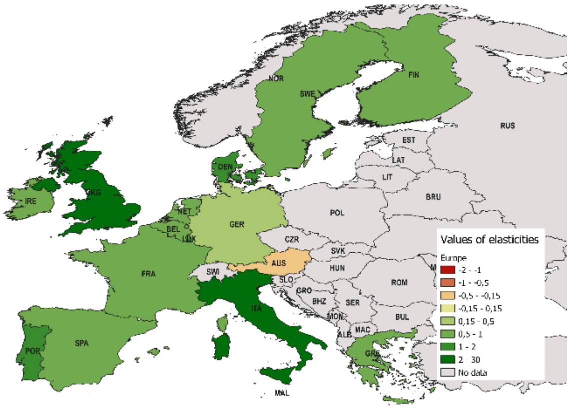


Figure 3.5. Elasticities for GDP and Recycled Non-Plastics 2008.

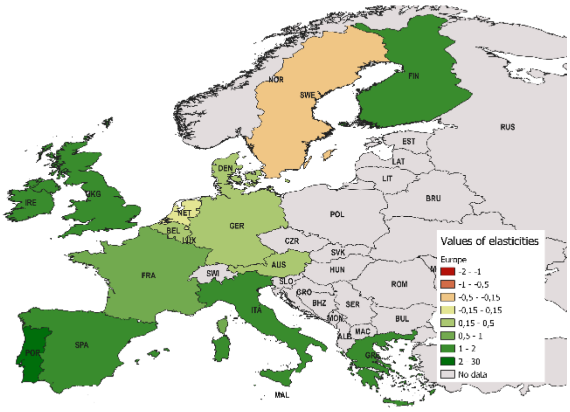


Figure 3.6. Elasticities for GDP and Recycled Non-Plastics 2018.

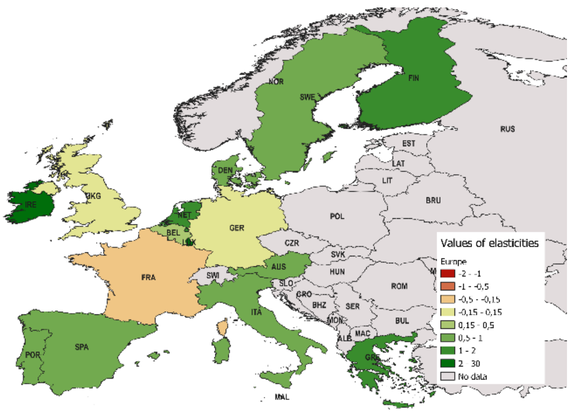


Figure 4.1. Recycled packaging waste (RPW) by country and sample average*.

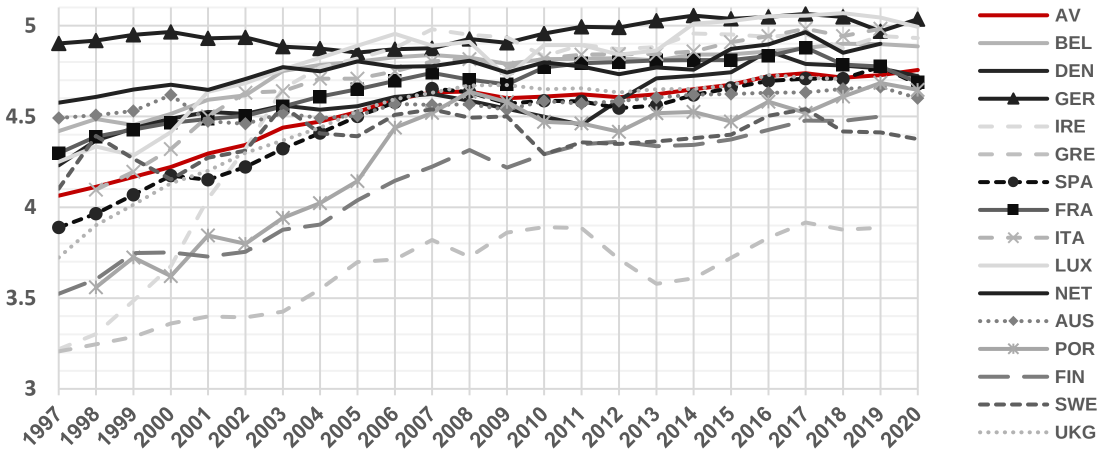


Figure 4.2. Recycled plastic packaging waste (RPPW) by country and sample average*.

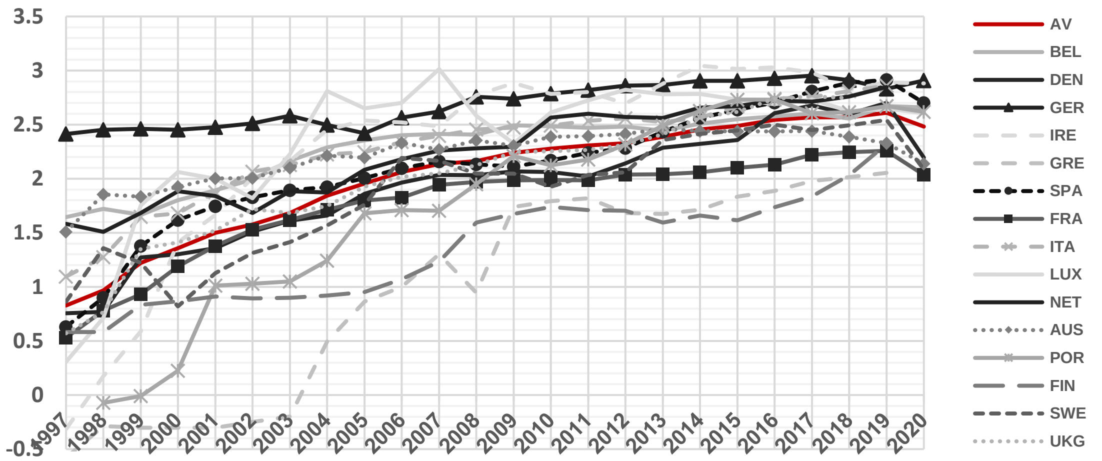
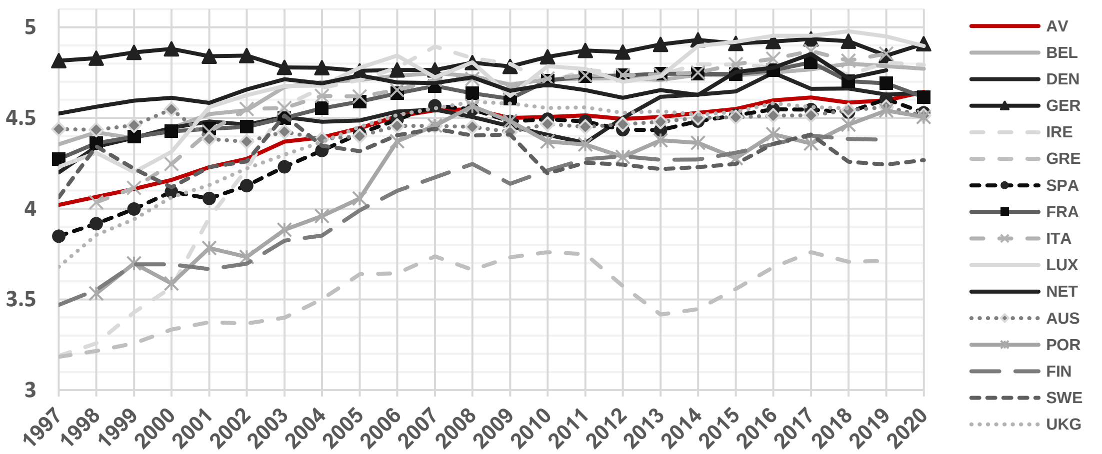


Figure 4.3. Recycled non-plastic packaging waste (RNPPW) by country and sample average*.



*The quantities are referred in natural logs for each country and for the sample average. The slope could be interpreted as the growth rate of the variable.

Figure 5.1. Recycled packaging waste as percentage of total.

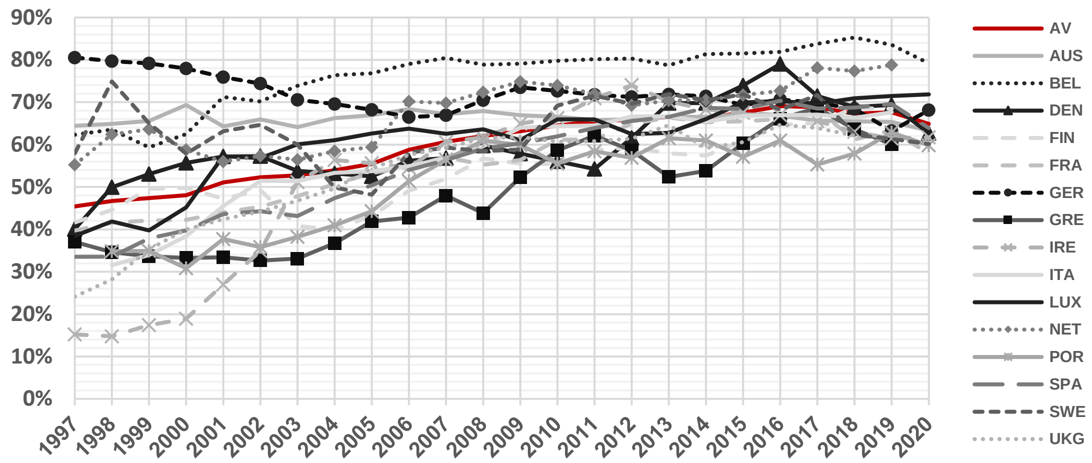


Figure 5.2. Recycled plastic packaging waste as percentage of total.

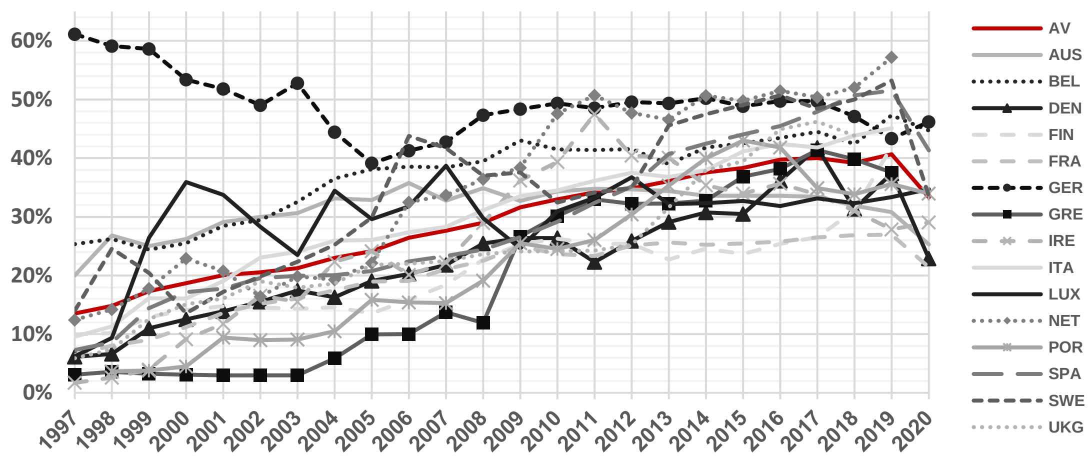
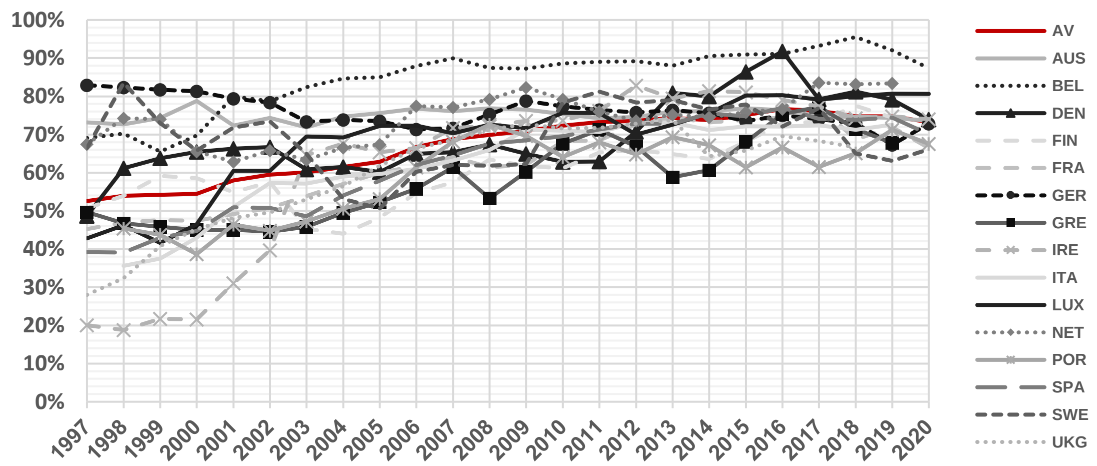


Figure 5.3. Recycled non-plastic packaging waste as percentage of total.



Conclusiones (Spanish)

A lo largo de los capítulos de esta tesis doctoral se han estudiado diversos aspectos que atañen a la relación de los residuos urbanos con el desarrollo económico a la par que se analiza el grado de avance hacia la sostenibilidad de dicha relación a lo largo del tiempo.

En primer lugar, hemos comprobado que España, al igual que ocurre con los países europeos, presenta una profunda heterogeneidad si medimos dos variables de eficiencia en la generación de residuos urbanos, rechazando la convergencia absoluta de las regiones españolas. Gracias a un conjunto de variables explicativas de los grupos de convergencia, hemos podido constatar que las regiones más ineficientes son aquellas con una menor renta per cápita, un menor nivel educativo, medido como los años promedio de educación recibida, y también presentan un marcado perfil regional, siendo las regiones mediterráneas de la costa y sur de España las que presentan una menor eficiencia. Este fenómeno parece guardar una fuerte relación con las zonas más turísticas del país. También se ha podido constatar que la ideología de los gobiernos autonómicos influye en la generación de residuos, más concretamente, parece que los años de gobiernos de partidos o coaliciones de izquierda presentan una mayor sensibilidad por aumentar la eficiencia en la generación de residuos urbanos.

En segundo lugar, hemos analizado la relación desde finales del siglo XX hasta la actualidad de la generación de residuos urbanos en relación con el PIB per cápita y con el Índice de Desarrollo Humano (IDH). Este estudio del desacople nos ha reflejado las dificultades para que todos los países europeos consigan uniformemente éxitos hacia un desacople absoluto. La tónica general parece ser un avance progresivo hacia un desacople relativo entre generación de residuos y

PIB per cápita, lo que quiere decir que con el tiempo los residuos aumentan en una proporción cada vez menor con el mismo aumento relativo del PIB. No obstante, eventos como la Gran Recesión de 2008 (GR) parecen frenar o romper esta senda de avance, al menos transitoriamente. Este avance hacia una desmaterialización de la economía no se hace tan evidente cuando estudiamos la relación con el IDH. En concreto, parece que aumentos en calidad de vida ligados a una mayor educación, o en salud como una mayor esperanza de vida, están más ligados a un consumo material de lo que puede estar el aumento del PIB, cada vez más basado en una economía de servicios.

Además, los países que presentan unas políticas públicas más ambiciosas en el ámbito de la prevención de residuos, así como de educación y sensibilización ambiental, presentan relaciones más intensas de desacople, lo que puede servir de guía para proponer mejoras en los países más rezagados en los objetivos ambientales. La Gran Recesión ha afectado de forma significativa en esta relación, perjudicando a los países más constreñidos por la austeridad fiscal, lo que se tradujo en una menor ambición a la hora de diseñar e implementar políticas ambientales, así como para definir objetivos menos claros y ambiciosos.

En el tercer capítulo, se ha expuesto la extensión a un sistema en la que se mide la relación del ciclo económico (evolución del PIBpc) con las variables de residuos recuperados (compostaje + reciclaje) y de los residuos no recuperados (resto de residuos urbanos). Con este modelo, buscábamos ver si las relaciones a lo largo del tiempo tendían a un trasvase de los residuos no recuperados hacia los recuperados, es decir, que cada vez una mayor cantidad de los residuos urbanos eran tratados de formas sostenibles de forma que pueden reincorporarse al ciclo productivo con posterioridad. En este punto, los resultados muestran que, de nuevo, la GR ha frenado la intensidad en la promoción del reciclaje y del compostaje y las tasas han tendido a estancarse con posterioridad a la GR salvo unos pocos países que han seguido incrementando el reciclaje. De nuevo, el debilitamiento de las políticas públicas y el cambio de las políticas europeas hacia la prevención, en vez de hacia un aumento del reciclaje, han conducido a que los residuos recuperados no presenten una elasticidad tan positiva como cabría esperar, si bien para la mayoría de los países se aprecian avances importantes durante los últimos 20 años si consideramos conjuntamente la relación de los residuos recuperados y no recuperados.

Finalmente, en el cuarto capítulo se ha extendido la relación de los envases con respecto a la evolución del PIBpc. Esta relación la hemos planteado, simétricamente, para los envases de plástico y el resto. Los envases de plástico han presentado un crecimiento elevado durante los últimos 20 años, y presentan un comportamiento muy persistente, hasta el punto de seguir aumentando la cantidad recogida a pesar de la GR, que sí parece afectar en la reducción del resto de envases. El COVID-19 ha supuesto un incremento súbito de la generación de plásticos en la mayoría de los países europeos. El reciclaje de los envases, por el contrario, sí que se ha visto negativamente afectado por la GR, reduciendo sus tasas de crecimiento. El plástico presenta unas tasas bajas de reciclaje, por debajo del 50%, para la inmensa mayoría de los países europeos. Si bien estas tasas se han incrementado con una fuerte intensidad, con la GR y la pandemia del COVID-19, las tasas de reciclaje han visto frenado su crecimiento de forma general. El debilitamiento de las políticas públicas en materia de reciclaje de envases parece haberse unido a una aproximación al umbral tecnológico del reciclado, especialmente para los envases que no son de plástico, que vienen presentando unas tasas de reciclaje más elevadas y cercanas al 60-80%.

Como corolario de todos los resultados y conclusiones obtenidas, destaca, en primer lugar, la fuerte heterogeneidad y dependencia de la situación de cada país y región a la hora de hacer frente a una renovación de las políticas en materia de prevención de residuos y promoción del reciclaje. En esta línea, sobresale la necesidad de un enfoque holístico que establezca unos objetivos comunes para grandes espacios económicos como la Unión Europea (UE), que sirva de guía para la investigación y el desarrollo de alternativas, así como la promoción de políticas e iniciativas exitosas, sin perjuicio de que las estrategias y políticas se ejecuten y se adapten necesariamente a las situaciones, ventajas y voluntades de cada una de las regiones y cada uno de los países en materia de gestión de residuos.

Otro de los puntos que destacan es la necesidad de reforzar las políticas destinadas a implementar una economía más circular, promoviendo el aumento de reciclaje, el ecodiseño o la contratación pública con criterios ambientales y ecológicos. Este refuerzo debería de ser promocionado a escala europea, de forma que se adapte a perseguir los objetivos deseados, si bien luego estos programas puedan ser delegados a los estados y regiones como los fondos de cohesión o la política agraria común. Un aspecto que queda soslayado en los diversos capítulos es que como consecuencia de la GR y/o del COVID-19, en términos de resultados, las políticas en estos ámbitos han terminado por ser menos efectivas que en los periodos de mayor crecimiento económico, lo que hace pensar que dichas políticas no constituyen un pilar sólido dentro de las políticas nacionales o europeas, al estar muy sujeto al ciclo económico. Para corregirlo, sería necesario un mayor desarrollo de normativas, asignación presupuestaria para establecer líneas de inversión e incentivos, objetivos más claros y ambiciosos, así como unos mecanismos de control supranacionales compatibles con la suficiente libertad para que los estados y regiones implementen sus propias estrategias.

Una última conclusión, se entronca con la necesidad de implementar políticas de continuidad en el ámbito de la desmaterialización y la mejora de la eficiencia, renovando los esfuerzos, sin olvidar la perspectiva de equidad, que exigiría un mayor compromiso transformador en el sistema económico y social, repensando todas las áreas de las políticas públicas, introduciendo transversalmente los criterios de sostenibilidad ligados a los ODS, de forma que la política medioambiental no quede como un método de corrección o minimización de los impactos ambientales del sistema económico, que no se vea modificado. Para este cambio, será esencial promover una formación y concienciación que favorezcan la innovación tanto social, decisional, organizativa como tecnológica para favorecer nuevos modelos institucionales y empresariales que estén lo más alineados posible con unos objetivos de satisfacción de necesidades humanas considerando también la sostenibilidad en un mundo finito.

Conclusions

Throughout the chapters of this doctoral thesis, we have studied various aspects of the relationship between municipal waste and economic development, as well as analyzing the degree of progress towards the sustainability of this relation over time.

Firstly, we have found that Spain, as is the case with the European member states, shows a marked heterogeneity if we measure two efficiency variables in the generation of urban waste, rejecting the absolute convergence of the Spanish regions. Thanks to a set of explanatory variables of the convergence groups, we have been able to find that the most inefficient regions are those with a lower per capita income, a lower educational level, measured as the average years of education received, and present a marked regional profile, with the Mediterranean regions of the coast and south of Spain being the least efficient.

This phenomenon seems to be strongly related to the most touristic areas of the country. It has also been found that the ideology of the autonomous governments influences waste generation; more specifically, it seems that the years of governments of left-wing parties or coalitions show a greater awareness of the need to increase efficiency in the generation of municipal waste.

Secondly, we have analyzed the relationship between the end of the 20th century and the present of municipal waste generation in relation to GDP per capita and Human Development Index (HDI). This study of decoupling has shown the difficulties for all European countries to achieve uniform success towards absolute decoupling. The general trend seems to be a progressive move towards a relative decoupling between waste generation and GDP per capita, meaning that over time waste increases at a decreasing rate with the same relative increase in GDP. However, events such as the Great Recession of 2008 (GR) seem to slow down or break this path of progress, at least temporarily. This progress towards a dematerialization of the economy is not so evident when we study the relationship with the HDI. In particular, it seems that increases in quality of life linked to higher education, or in health such as longer life expectancy, are more closely linked to material consumption than the increase in GDP, which is increasingly based on a service economy.

In addition, countries with more ambitious public policies in the area of waste prevention and environmental education and awareness have stronger decoupling relationships, which can be used as a guide for proposing improvements in laggard countries in terms of environmental objectives. The Great Recession has significantly affected this relationship, damaging the countries most constrained by fiscal austerity, which resulted in less ambition when designing and implementing environmental policies, as well as in defining less clear and ambitious objectives.

In the third chapter, we presented the extension to a system in which we measured the relationship between the economic cycle (evolution of GDPpc) and the variables of recovered waste (composting + recycling) and non-recovered waste (rest of municipal waste). Using this model, the aim was to see whether the relationships over time tended to shift from non-recovered waste to recovered waste, i.e., that an increasing amount of municipal waste was treated in sustainable ways so that it can be reintroduced into the production cycle afterwards. On this point, the results show that, again, the GR has slowed down the intensity of recycling and composting promotion and rates have tended to stagnate post-GR except for a few countries that have continued to increase recycling. Once again, the weakening of government policies and the change in European policies towards prevention rather than more recycling have resulted in recovered waste not showing such a positive elasticity as might be expected.

Although this point, significant progress can be seen over the last 20 years for most countries when the ratio of recovered to non-recovered waste is considered together.

Lastly, in the fourth chapter we have developed the relationship of packaging with respect to the evolution of per capita GDP. This relationship is symmetrical for plastic packaging and the rest. Plastic packaging has shown strong growth over the last 20 years, and has a very persistent behavior, to the point of still increasing in quantity collected despite the GR, which does seem to affect the reduction of the rest of packaging. COVID-19 has led to a sudden increase in the generation of plastics in most European countries. Packaging recycling, on the other hand, has been negatively affected by the GR, reducing growth rates. Plastic has low recycling rates, below 50%, for the vast majority of European countries. While these rates have increased sharply, with the GR and the COVID-19 pandemic, recycling rates have generally slowed. The weakening of public policies on packaging recycling seems to have come together with an approaching recycling threshold, especially for non-plastic packaging, linked to organizational and sorting issues more than technological, which has been showing higher recycling rates close to 60-80%.

As a corollary of all the results and conclusions obtained, first of all, is underlined the strong heterogeneity and dependence on the situation of each country and region when facing a renovation of waste prevention and recycling policies. In this line, it is necessary to highlight the need for a holistic approach that establishes common objectives for large economic areas such as the European Union (EU), which serves as a guide for research and development of alternatives, as well as the promotion of successful policies and initiatives, without prejudice to the fact that strategies and policies are necessarily implemented and adapted to the situations, advantages and will of each of the regions and each of the countries in terms of waste management.

Another point that stands out is the need to strengthen policies aimed at implementing a more circular economy, promoting increased recycling, eco-design or public procurement with environmental and ecological criteria. This reinforcement should be promoted at the European level, so that it is adapted to pursue the desired objectives, although these programs can then be delegated to Member States and regions such as cohesion funds or the common agricultural policy. Another important aspect is that as a result of the GR and/or COVID-19 policies in these areas have ended up being less effective in terms of results than in periods of higher economic growth, which suggests that these policies do not constitute a robust foundation within national or European policies, as they are highly subject to the economic cycle. To rectify this, it would be necessary to further develop regulations, budget allocations to establish lines of investment and incentives, clearer and more ambitious objectives, as well as supranational control mechanisms compatible with sufficient freedom for the states and regions to implement their own strategies.

A final conclusion is linked to the need maintain dematerialization and the improvement of efficiency without forgetting the perspective of equity, which would require a greater transformative commitment in the economic and social system. This will require introducing the sustainability criteria linked to the SDGs across all areas of public policies with the objective that environmental policy does not remain as a method of correcting or minimizing the environmental impacts of the economic system, which is never modified. For this change, it will be essential to promote education, training and awareness that promote social, decision-making, organizational and technological innovation in order to develop new institutional and business models that are as aligned as possible with the objectives of satisfying human needs, also considering sustainability in a finite world.

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Appendixes

Appendix 1

This Appendix includes the variables we have employed as explanatory variables in the probit analysis of Section 3.

1) Economic and development indicators
   a) Regional GDP per capita. Source: National Statistical Institute of Spain.
   b) Population of the region. Source: National Statistical Institute of Spain.
   c) Average expenditure of the regions. Source: National Statistical Institute of Spain.
   d) Median income of the regions. Source: Living Conditions Survey, National Statistical Institute of Spain.
   e) Continuing education. Proportion of people between the ages of 25 and 64 engaged in educational studies. Source: Living Conditions Survey, National Statistical Institute of Spain.
   f) Higher Education. Proportion of graduates over the total population. Source: Living Conditions Survey, National Statistical Institute of Spain.
   g) Regional Transparency index. Source: INCAU, Transparency International Spain
   h) Proportion of homes with problems of insecurity, theft and vandalism. Source: Living Conditions Survey, National Statistical Institute of Spain.
   i) Proportion of the population in municipalities with less than 10,000 inhabitants. Source: Living Conditions Survey, National Statistical Institute of Spain.
   j) Proportion of GDP generated by primary sector. Source: National Statistical Institute of Spain.
   k) Proportion of GDP generated by industrial and manufacturing sector. Source: National Statistical Institute of Spain.
   l) Proportion of GDP generated by services sector. Source: National Statistical Institute of Spain.
2) Environmental indicators and waste
   a) Number of water treatment plants in the region. Source: Ministry of Ecological Transition.
   b) Number of landfills in the region. Source: Ministry of Ecological Transition

c) Recycling and composting plants, both mixed waste and separate collection. Source: Ministry of Ecological Transition.
d) Proportion of people who report environmental and pollution problems. Source: Living Conditions Survey, National Statistical Institute of Spain.

3) Economic policy and government indicators
   a) Total Regional expenditure of local entities on environmental programs. Source: General State Budgets.
   b) Per capita regional expenditure on environment. Source: General State Budgets.
   c) Per capita total regional EELL expenditure. Source: General State Budgets.
   d) Percentage of the environmental expenditure over the total EELL expenditure. Source: General State Budgets.
   e) Number of years of right-wing government. Source. Own elaboration.
   f) Number of years of left-wing government. Source. Own elaboration.
   g) Number of years of government of regionalist parties. Source: Own elaboration

Appendix 2

Table A1. Descriptive análisis.

	MSW (thousands of Kg)							GDP (euros)							HDI (between 0 and 1)						
	1995	2018	95-18	95-07	07-18	07-14	14-18	1995	2018	95-18	95-07	07-18	07-14	14-18	1995	2018	95-18	95-07	07-18	07-14	14-18
EU27	0.465	0.492	0.3%	0.9%	-0.5%	-1.2%	0.8%	19,707	27,681	1.5%	2.2%	0.7%	0.0%	2.1%	0.780	0.890	0.6%	0.8%	0.4%	0.4%	0.4%
Austria	0.438	0.580	1.2%	2.6%	-0.3%	-0.7%	0.5%	27,604	37,873	1.4%	2.2%	0.5%	0.1%	1.1%	0.817	0.914	0.5%	0.6%	0.3%	0.4%	0.3%
Belgium	0.455	0.412	-0.4%	0.7%	-1.7%	-2.1%	-0.8%	26,267	35,686	1.3%	2.1%	0.5%	0.0%	1.3%	0.851	0.919	0.3%	0.5%	0.2%	0.2%	0.2%
Bulgaria	0.693	0.422	-2.1%	-1.9%	-2.4%	-3.1%	-1.1%	3,227	6,526	3.1%	3.5%	2.7%	1.8%	4.3%	0.697	0.816	0.7%	0.8%	0.6%	0.6%	0.6%
Cyprus	0.600	0.640	0.3%	1.4%	-1.1%	-2.2%	1.6%	17,643	23,927	1.3%	2.9%	-0.6%	-2.9%	7.5%	0.783	0.873	0.5%	0.6%	0.3%	0.2%	0.9%
Czechia	0.302	0.352	0.7%	-0.2%	1.6%	0.7%	3.2%	10,163	17,651	2.4%	3.5%	1.3%	0.1%	3.5%	0.753	0.891	0.7%	1.0%	0.4%	0.5%	0.3%
Denmark	0.522	0.767	1.7%	3.5%	-0.3%	-0.3%	-0.2%	37,222	48,372	1.1%	1.8%	0.4%	-0.4%	1.8%	0.831	0.930	0.5%	0.7%	0.3%	0.4%	0.1%
Estonia	0.368	0.406	0.4%	1.7%	-0.9%	-3.2%	3.2%	5,398	15,086	4.6%	7.8%	1.2%	-0.2%	3.7%	0.724	0.882	0.9%	1.2%	0.5%	0.4%	0.5%
Finland	0.414	0.552	1.3%	1.7%	0.8%	-0.7%	3.4%	24,130	36,909	1.9%	3.7%	-0.1%	-1.1%	1.7%	0.816	0.925	0.5%	0.8%	0.2%	0.3%	0.2%
France	0.476	0.527	0.4%	1.1%	-0.3%	-0.7%	0.4%	25,707	33,002	1.1%	1.7%	0.4%	-0.1%	1.3%	0.825	0.891	0.3%	0.4%	0.2%	0.3%	0.1%
Germany	0.624	0.616	-0.1%	-0.6%	0.5%	1.2%	-0.7%	26,308	35,907	1.4%	1.5%	1.2%	1.1%	1.4%	0.834	0.939	0.5%	0.8%	0.2%	0.2%	0.2%
Greece	0.304	0.503	2.3%	3.4%	1.0%	1.0%	1.1%	15,070	17,765	0.7%	3.5%	-2.6%	-4.1%	1.2%	0.768	0.872	0.6%	0.8%	0.3%	0.3%	0.3%
Hungary	0.460	0.381	-0.8%	-0.1%	-1.6%	-2.4%	-0.2%	6,756	12,554	2.7%	3.6%	1.8%	0.4%	4.1%	0.741	0.845	0.6%	0.8%	0.3%	0.3%	0.4%
Ireland	0.514	0.579	0.5%	3.6%	-3.0%	-4.6%	0.8%	22,491	58,326	4.1%	5.3%	2.7%	-0.6%	16.6%	0.795	0.942	0.8%	1.0%	0.4%	0.3%	1.0%
Italy	0.454	0.499	0.4%	1.8%	-1.0%	-1.9%	0.6%	24,814	26,729	0.3%	1.3%	-0.7%	-1.9%	1.3%	0.800	0.883	0.4%	0.7%	0.2%	0.1%	0.3%
Latvia	0.263	0.406	1.9%	3.3%	0.4%	-1.0%	2.8%	3,824	12,132	5.1%	8.5%	1.6%	0.1%	4.2%	0.673	0.854	1.0%	1.6%	0.4%	0.3%	0.5%
Lithuania	0.424	0.463	0.4%	-0.2%	1.0%	0.5%	1.8%	3,999	13,279	5.4%	7.7%	2.9%	2.1%	4.3%	0.703	0.869	0.9%	1.4%	0.5%	0.4%	0.5%
Luxembourg	0.592	0.616	0.2%	1.4%	-1.1%	-1.4%	-0.7%	56,874	84,410	1.7%	3.4%	-0.1%	-0.8%	1.1%	0.817	0.909	0.5%	0.7%	0.2%	0.1%	0.4%
Netherlands	0.541	0.513	-0.2%	1.0%	-1.5%	-1.9%	-0.8%	29,268	41,666	1.5%	2.5%	0.6%	-0.2%	1.9%	0.862	0.934	0.3%	0.4%	0.3%	0.3%	0.2%
Poland	0.285	0.329	0.6%	1.0%	0.2%	-2.4%	4.9%	4,938	12,575	4.1%	4.7%	3.6%	3.2%	4.3%	0.740	0.872	0.7%	0.8%	0.6%	0.6%	0.6%
Portugal	0.353	0.507	1.6%	2.5%	0.7%	-0.6%	3.0%	13,659	18,101	1.2%	2.0%	0.4%	-0.9%	2.8%	0.760	0.850	0.5%	0.5%	0.4%	0.5%	0.3%
Romania	0.342	0.271	-1.0%	1.0%	-3.2%	-6.1%	2.2%	3,692	8,715	3.8%	4.1%	3.5%	2.3%	5.6%	0.687	0.816	0.8%	1.1%	0.4%	0.4%	0.4%
Slovakia	0.295	0.414	1.5%	0.0%	3.2%	1.2%	6.7%	6,600	15,564	3.8%	5.1%	2.4%	1.8%	3.4%	0.751	0.857	0.6%	0.7%	0.5%	0.5%	0.4%
Slovenia	0.596	0.488	-0.9%	-1.0%	-0.7%	-2.8%	3.1%	11,435	20,216	2.5%	4.2%	0.7%	-0.8%	3.5%	0.782	0.902	0.6%	0.9%	0.3%	0.2%	0.4%
Spain	0.506	0.476	-0.3%	1.2%	-1.8%	-3.7%	1.5%	17,961	24,913	1.4%	2.7%	0.1%	-1.5%	2.9%	0.800	0.893	0.5%	0.5%	0.4%	0.4%	0.4%
Sweden	0.386	0.436	0.5%	2.0%	-1.0%	-1.3%	-0.5%	28,446	44,045	1.9%	3.0%	0.8%	0.3%	1.6%	0.857	0.937	0.4%	0.5%	0.3%	0.4%	0.2%
U.K.	0.499	0.465	-0.3%	1.1%	-1.8%	-2.3%	-1.0%	23,176	32,777	1.5%	2.6%	0.4%	0.0%	1.1%	0.839	0.920	0.4%	0.5%	0.3%	0.4%	0.1%

This table presents the initial value, the final value and the average rates of growth of the variables for the indicated periods.

Table A.2. Testing for unit roots. Per capita MSW

	ADF-GLS	CKP1	TB1	CKP2	TB1	TB2	CKP3	TB1	TB2	TB3
EU27	-1.56	-2.26	2000	*-3.72**	2003	2010	**-6.72****	2003	2008	2014
Austria	-0.90	**-4.27****	2000	**-4.48****	1998	2004	**-5.73****	1998	2004	2010
Belgium	-0.56	-2.87	2008	**-7.83****	2007	2016	**-10.37****	1999	2008	2015
Bulgaria	-1.86	**-4.37****	2011	**-5.19****	1998	2011	**-5.96****	1998	2009	2013
Cyprus	-1.56	-2.42	2010	-2.54	2010	2015	**-6.65****	2001	2010	2015
Czechia	-1.92	**-3.43****	2001	**-5.52****	2001	2012	**-4.81****	1998	2001	2012
Denmark	-0.48	**-6.98****	2007	**-4.49****	2009	2012	**-8.08****	2003	2009	2012
Estonia	-1.70	**-4.85****	2009	**-5.75****	2007	2011	**-5.26****	2001	2008	2012
Finland	*-2.91*	*-3.20**	2001	**-3.88****	2001	2009	**-5.73****	2001	2009	2014
France	-1.01	*-3.13**	2008	-2.32	2003	2008	**-4.33****	2003	2007	2016
Germany	-1.68	-2.86	2003	-3.21	2003	2007	**-5.08****	2003	2007	2016
Greece	-1.51	*-3.13**	2010	-3.12	1999	2010	**-5.30****	1999	2010	2013
Hungary	-1.57	*-3.25**	2010	-2.77	2000	2010	**-5.34****	2000	2008	2012
Ireland	-1.15	-2.10	2008	**-4.05****	2007	2010	**-4.58****	2001	2006	2011
Italy	-1.13	-2.05	2007	-2.96	2007	2012	-3.19	1999	2007	2012
Latvia	-2.46	**-4.15****	2010	**-4.11****	2007	2011	**-5.66****	2001	2007	2011
Lithuania	-0.79	**-4.41****	1999	**-4.03****	1999	2009	**-4.79****	1999	2009	2013
Luxembourg	-0.68	-2.95	2009	*-3.50**	2008	2013	**-6.09****	1998	2009	2013
Netherlands	-0.79	-2.91	2006	**-4.10****	2001	2010	**-6.48****	2001	2007	2013
Poland	*-3.17*	-2.43	2005	-2.09	2001	2005	**-5.65****	2001	2005	2014
Portugal	-2.48	*-3.11**	2012	**-4.49****	2002	2012	**-5.28****	2001	2008	2012
Romania	-1.53	-2.91	2010	**-4.83****	2009	2012	**-6.58****	1999	2009	2012
Slovakia	-1.01	-2.07	2015	*-3.61**	2002	2016	**-5.63****	2002	2008	2015
Slovenia	-2.74	-1.94	2004	-3.13	2004	2011	*-3.97**	2002	2009	2013
Spain	-1.95	-1.20	2000	-2.67	2002	2014	**-6.19****	2000	2008	2014
Sweden	-0.95	**-3.69****	2007	**-4.88****	2008	2011	**-5.87****	1999	2008	2011
U. Kingdom	-1.07	-1.72	2002	*-3.73**	2004	2013	**-6.28****	2002	2007	2013

ADF-GLS is the statistic proposed by Elliot *et al.* (1996) when the specification includes an intercept and a deterministic trend. CKPi is the ADF type statistic proposed by Carrion-i-Silvestre *et al.* (2009) when the specification includes i breaks that affect both the intercept and the deterministic trend, with i=1,2,3.

** rejection of the unit root null hypothesis for a 5% significance level

*: rejection of the unit root null hypothesis for a 10% significance level

Table A.3. Testing for unit roots. Per capita GDP

	ADF-GLS	CKP1	TB1	CKP2	TB1	TB2	CKP3	TB1	TB2	TB3
EU27	-1.33	-2.24	2009	*-3.50**	2009	2013	**-5.47****	2003	2009	2013
Austria	-0.91	-2.43	2009	-3.12	2003	2009	**-4.16****	2003	2009	2013
Belgium	-1.26	**-3.43****	2009	**-4.15****	2009	2013	**-6.85****	2002	2009	2013
Bulgaria	-2.25	-1.99	1999	**-3.88****	2001	2009	**-8.16****	2001	2009	2014
Cyprus	-2.33	-1.51	2009	**-4.37****	2009	2014	**-4.56****	2009	2011	2014
Czechia	-2.28	-1.76	2009	-2.39	2003	2009	**-4.15****	2003	2009	2013
Denmark	-1.98	-2.91	2009	-3.33	2009	2014	**-4.64****	2002	2009	2014
Estonia	-1.54	-2.74	2009	-2.57	2006	2009	*-3.85**	2003	2007	2009
Finland	-1.25	-2.39	2009	*-3.42**	2009	2014	-3.06	2009	2012	2015
France	-1.11	-2.17	2009	-3.01	2000	2009	**-4.41****	2000	2009	2014
Germany	-1.13	-2.94	2009	-2.55	2009	2012	**-5.23****	2003	2009	2012
Greece	-2.35	-1.94	2009	-2.20	2008	2013	**-6.40****	2003	2008	2013
Hungary	-1.71	-2.21	2009	-3.39	2009	2013	-3.41	1998	2009	2013
Ireland	-1.50	-2.76	2009	*-3.56**	2008	2015	**-4.77****	2000	2008	2015
Italy	-1.46	-2.06	2009	*-3.49**	2008	2015	**-4.54****	2000	2009	2014
Latvia	-1.65	-2.41	2009	**-6.49****	2002	2009	*-3.84**	2003	2008	2010
Lithuania	-2.26	-2.10	2009	**-4.57****	2002	2009	**-4.30****	2002	2007	2009
Luxembourg	-0.87	*-3.39**	2009	**-5.09****	1999	2009	**-4.39****	2000	2007	2009
Netherlands	-1.53	-1.80	2009	-2.31	2002	2009	**-4.56****	2002	2009	2013
Poland	-1.17	-2.06	2006	*-3.60**	2007	2016	**-5.24****	2002	2007	2016
Portugal	-1.90	-1.15	2000	*-3.83**	2001	2012	**-6.15****	2000	2007	2014
Romania	-2.12	-1.86	1999	-2.83	2001	2010	-3.75	2001	2008	2012
Slovakia	-2.14	-2.23	2007	-3.47	2004	2009	**-4.34****	2004	2009	2013
Slovenia	-1.70	-1.34	2009	-2.84	2009	2015	**-4.99****	2005	2009	2014
Spain	-2.02	-1.95	2009	-2.72	2007	2013	**-6.33****	2000	2007	2013
Sweden	-1.06	**-3.83****	2009	-3.31	2008	2010	*-3.61**	2008	2010	2012
U. Kingdom	-1.49	*-3.13**	2009	-2.69	2007	2009	**-4.29****	2000	2008	2010

ADF-GLS is the statistic proposed by Elliot *et al.* (1996) when the specification includes an intercept and a deterministic trend. CKPi is the ADF type statistic proposed by Carrion-i-Silvestre *et al.* (2009) when the specification includes i breaks that affect both the intercept and the deterministic trend, with i=1,2,3.

**: rejection of the unit root null hypothesis for a 5% significance level

*: rejection of the unit root null hypothesis for a 10% significance level

Table A.4. Testing for unit roots. HDI

	ADF-GLS	CKP1	TB1	CKP2	TB1	TB2	CKP3	TB1	TB2	TB3
Austria	-1.67	**-4.52****	2007	**-4.78****	2002	2007	**-4.66****	2001	2003	2007
Belgium	-1.11	**-3.94****	2008	*-3.57**	1998	2006	**-6.37****	1998	2006	2009
Bulgaria	-1.78	*-3.23**	2003	*-3.84**	2001	2009	**-6.02****	2001	2009	2015
Cyprus	-1.97	-2.81	2010	**-3.98****	2002	2010	**-5.70****	2002	2010	2015
Czechia	-0.48	**-4.41****	2007	**-3.87****	1998	2007	**-4.86****	1998	2004	2007
Denmark	-0.71	*-3.26**	2003	*-3.62**	2005	2011	**-8.49****	2001	2005	2011
Estonia	-0.89	*-3.30**	2005	**-4.83****	2001	2008	**-8.21****	2001	2007	2011
Finland	-0.78	-3.09	2004	-3.48	2001	2004	-3.55	2001	2004	2009
France	-2.13	-3.06	2005	-3.24	2000	2005	**-4.74****	2000	2005	2014
Germany	-0.30	**-4.63****	2006	**-4.20****	2004	2006	**-4.16****	1998	2004	2006
Greece	-1.03	**-3.64****	2005	**-3.76****	1999	2006	**-5.93****	2001	2006	2011
Hungary	-0.76	**-4.11****	2006	**-5.00****	1998	2007	**-4.95****	1998	2007	2011
Ireland	-1.76	-1.39	1998	*-3.78**	1998	2009	-3.77	1998	2006	2010
Italy	-1.08	-2.86	2007	**-4.72****	2006	2013	**-5.55****	2001	2007	2013
Latvia	-1.44	-2.89	2006	**-4.22****	2007	2010	**-5.98****	2001	2006	2010
Lithuania	-1.05	*-3.35**	2009	**-5.30****	2006	2009	**-6.87****	2006	2009	2014
Luxembourg	-0.83	-2.86	2001	-3.26	1999	2009	**-5.74****	1999	2009	2013
Netherlands	-1.92	-2.70	2011	**-5.36****	2006	2011	**-4.97****	2005	2008	2011
Poland	-1.42	**-5.07****	2004	**-7.55****	1999	2004	-3.07	2004	2012	2014
Portugal	-2.01	-2.36	1999	-2.87	1999	2013	**-5.15****	1999	2004	2013
Romania	-1.88	-2.12	2008	-2.74	2002	2008	**-4.76****	2002	2008	2012
Slovakia	-1.98	-2.54	2007	-3.25	2003	2009	**-5.29****	2003	2009	2012
Slovenia	-0.61	*-3.28**	2005	**-4.18****	2005	2012	**-5.80****	2003	2009	2012
Spain	-1.27	*-3.38**	2010	*-3.66**	2000	2005	**-4.29****	2000	2005	2010
Sweden	-1.90	-2.61	2004	**-4.92****	2000	2013	**-4.57****	1998	2004	2013
U. Kingdom	-0.93	**-4.08****	2006	-2.46	2011	2013	**-4.81****	2006	2011	2013

ADF-GLS is the statistic proposed by Elliot *et al.* (1996) when the specification includes an intercept and a deterministic trend. CKPi is the ADF type statistic proposed by Carrion-i-Silvestre *et al.* (2009) when the specification includes i breaks that affect both the intercept and the deterministic trend, with i=1,2,3.

**: rejection of the unit root null hypothesis for a 5% significance level

*: rejection of the unit root null hypothesis for a 10% significance level

Appendix 3

Table A1: Descriptive analysis for per capita Gross Domestic Product (GDP) and Municipal Solid Waste (MSW)

	GDP						MSW					
	1995	2018	g (95-07)	g (08-18)	g (08-13)	g (14-18)	1995	2018	g (95-07)	g (08-18)	g (08-13)	g (14-18)
AUS	38,126	52,243	2.2%	0.4%	-0.1%	1.1%	437.3	579.2	2.6%	-0.4%	-0.7%	0.6%
BEL	35,296	47,748	2.1%	0.5%	-0.1%	1.2%	455.1	409.3	0.7%	-1.6%	-1.9%	-0.9%
FRA	33,159	42,492	1.7%	0.5%	-0.1%	1.2%	474.6	532.5	1.1%	-0.1%	-0.7%	0.8%
GER	36,802	49,776	1.6%	1.0%	0.6%	1.3%	625.9	606.2	-0.5%	0.1%	0.5%	-1.0%
HUN	16,174	30,279	3.6%	1.9%	-0.4%	4.2%	460.1	381.5	-0.1%	-1.7%	-3.6%	-0.2%
ITA	35,707	38,530	1.2%	-0.5%	-2.0%	1.3%	453.5	498.9	1.7%	-0.9%	-2.3%	0.6%
JAP	35,306	42,364	1.0%	0.8%	0.4%	1.3%	416.3	337.8	-0.4%	-1.1%	-1.3%	-0.8%
KOR	17,688	40,947	4.7%	2.7%	2.8%	2.5%	386.7	396.3	-0.2%	0.2%	-1.9%	2.5%
NET	37,496	53,218	2.5%	0.4%	-0.8%	1.8%	539.3	511.0	1.0%	-1.6%	-2.6%	-0.8%
POL	12,141	30,259	4.6%	3.4%	2.5%	4.4%	287.0	325.0	1.0%	0.2%	-1.7%	4.9%
SPA	27,124	37,700	2.6%	0.3%	-2.0%	2.9%	505.5	475.7	1.1%	-1.5%	-3.8%	1.5%
SWI	52,047	68,580	1.6%	0.7%	0.2%	1.1%	597.2	706.1	1.6%	-0.4%	-0.8%	-0.9%
UKG	30,917	43,720	2.6%	0.5%	-0.2%	1.1%	498.1	463.4	1.1%	-1.5%	-2.3%	-1.0%
USA	41,719	59,801	2.2%	1.1%	0.3%	1.9%	739.4	811.2	0.4%	0.8%	-0.6%	2.5%

This table presents the initial value, the final value, and the average rates of growth of the variables for the indicated periods. GDP is presented in Purchasing power parity dollars of 2015 and MSW in kilograms, both of them in per capita terms. g(xx-yy) indicates the cumulative growth rate between period xx and yy.

Table A2: Descriptive analysis for Recovered and Unrecovered Waste (RW and URW)

	RW								URW							
	1995	**%RW**	**2018**	**%RW**	**g (95-07)**	**g (08-18)**	**g (08-13)**	**g (14-18)**	**1995**	**%URW**	**2018**	**%URW**	**g (95-07)**	**g (08-18)**	**g (08-13)**	**g (14-18)**
AUS	220.9	50.5	334.0	57.7	4.1%	-1.3%	-2.5%	1.2%	216.4	49.5	245.2	42.3	0.8%	1.1%	2.1%	-0.2%
BEL	87.1	19.1	222.5	54.4	10.2%	-1.3%	-1.8%	-0.7%	368.0	80.9	186.9	45.6	-4.4%	-1.9%	-2.0%	-1.2%
FRA	84.2	17.7	239.9	45.0	6.3%	2.9%	2.1%	4.0%	390.4	82.3	292.6	55.0	-0.5%	-1.9%	-2.2%	-1.5%
GER	246.7	39.4	406.8	67.1	3.5%	0.6%	0.5%	-0.5%	379.2	60.6	199.5	32.9	-4.5%	-0.8%	0.5%	-2.1%
HUN	7.4	1.6	142.6	37.4	18.2%	7.5%	7.7%	5.0%	452.7	98.4	238.9	62.6	-1.0%	-4.7%	-6.3%	-2.8%
ITA	21.9	4.8	248.3	49.8	16.6%	6.6%	8.0%	5.2%	431.6	95.2	250.7	50.2	-0.3%	-5.0%	-6.7%	-3.1%
JAP	40.7	9.8	67.5	20.0	5.9%	-1.2%	-1.0%	-1.5%	375.7	90.2	270.3	80.0	-1.4%	-1.0%	-1.4%	-0.6%
KOR	91.5	23.7	245.8	62.0	7.7%	0.6%	-2.1%	3.8%	295.2	76.3	150.6	38.0	2.1%	-0.3%	-1.5%	0.6%
NET	214.0	39.7	285.6	55.9	2.6%	-0.2%	-2.0%	1.6%	325.3	60.3	225.4	44.1	-0.3%	-3.1%	-3.1%	-3.4%
POL	5.3	1.8	111.4	34.3	13.8%	12.7%	5.7%	11.9%	281.8	98.2	213.6	65.7	0.4%	-2.9%	-2.8%	2.0%
SPA	35.6	7.0	165.5	34.8	14.3%	-2.7%	-7.5%	4.6%	469.8	93.0	310.2	65.2	-1.3%	-0.7%	-1.6%	0.0%
SWI	231.0	38.7	370.7	52.5	3.9%	0.1%	-0.4%	-1.4%	366.2	61.3	335.4	47.5	-0.3%	-0.9%	-1.2%	-0.4%
UKG	34.8	7.0	204.4	44.1	15.2%	0.4%	1.1%	-0.6%	463.2	93.0	259.0	55.9	-1.7%	-2.8%	-4.5%	-1.3%
USA	189.7	25.7	260.6	32.1%	2.5%	0.4%	0.0%	0.7%	549.7	74.3	550.6	67.9	-0.5%	0.9%	-0.9%	3.4%

This table presents the initial value, the final value, and the average rates of growth of the variables for the indicated periods. RW and URW are presented in kilograms per capita and as a percentage of MSW. g(xx-yy) indicates the cumulative growth rate between period xx and yy.

Table A3. Testing for unit roots. Variable: Recovered waste (per capita)

	DF-GLS	CKP1	TB1	CKP2	TB1	TB2	CKP3	TB1	TB2	TB3
AUS	-1.31	-3.43**	1996	-3.68**	1994	1997	-3.39	1993	1996	2004
FRA	-0.88	-3.32*	2012	-4.22**	2003	2013	-5.89**	2003	2009	2016
GER	-0.67	-3.06	1999	-5.88**	2003	2016	-5.96**	2001	2003	2016
ITA	-0.60	-2.52	2002	-5.52**	2002	2007	-4.72**	2002	2007	2009
JAP	-1.37	-3.41**	2006	-4.62**	2000	2008	-4.29**	2000	2005	2008
KOR	-1.98	-3.21*	2010	-5.16**	2008	2014	-6.72**	1995	2008	2014
SPA	-2.52	-2.67	2002	-2.89	1998	2002	-6.19**	1998	2002	2010
SWI	-1.66	-3.15*	2008	-3.34	2008	2014	-3.33	2008	2013	2015
UKG	-0.76	-2.49	2010	-3.88**	2002	2008	-4.96**	2002	2007	2017
USA	-0.88	-2.25	2000	-5.10**	1994	2008	-4.68**	1994	2007	2010
BEL	-1.94	-2.50	2004	-3.35	1997	2008	-2.84	1995	2000	2008
HUN	-1.62	-2.53	2004	-2.15	2003	2005	-7.05**	2003	2005	2014
NET	-1.55	-2.58	1997	-2.27	1994	2000	-3.49	1994	1997	2010
POL	-2.60	-2.79	2014	-2.64	2014	2016	-5.74**	2003	2011	2015

This table presents the DF-GLS statistic proposed by Elliot *et al.* (1996), whilst the CKP statistics are the ones proposed by Carrion-i-Silvestre *et al.* (2009) for 1, 2 and 3 breaks in both the intercept and the trend of the model.

* and ** mean rejection of the unit root null hypothesis at 10% and 5% significance level, respectively.

Table A4. Testing for unit roots. Variable: Unrecovered waste (per capita)

	DF-GLS	CKP1	TB1	CKP2	TB1	TB2	CKP3	TB1	TB2	TB3
AUS	-1.72	-2.63	1995	-2.63	1994	1996	-4.55**	1994	1996	2004
FRA	-2.54	-3.68**	2008	-2.22	2003	2005	-4.56**	2003	2006	2015
GER	-0.53	-2.46	2005	-3.57*	1999	2005	-5.92**	1999	2005	2010
ITA	-1.11	-2.57	2007	-9.13**	2006	2012	-7.39**	2007	2009	2012
JAP	-2.01	-1.65	2008	-4.04**	2003	2011	-5.18**	2001	2008	2011
KOR	-2.70	-3.95**	1995	-3.97**	1998	2008	-6.19**	1995	2006	2008
SPA	-2.06	-2.43	1995	-1.47	1995	2002	-4.24**	1995	2002	2008
SWI	-2.76	-3.47**	2002	-3.39	2001	2003	-4.29**	1999	2002	2004
UKG	-2.98	-2.05	2005	-5.25**	2002	2011	-3.60	2002	2008	2012
USA	-1.52	-2.09	2016	-3.44	1997	2016	-5.82**	1997	2009	2016
BEL	-2.67	-1.85	1997	-4.59**	1996	1999	-5.36**	1996	1999	2008
HUN	-1.49	-3.36**	2010	-5.45**	1997	2010	-6.16**	1997	2006	2014
NET	-3.96**	-2.05	1993	-5.11**	1995	2007	-5.02**	1993	1996	2007
POL	-2.61	-2.97*	2014	-2.37	2005	2014	-4.99**	2001	2005	2014

This table presents the DF-GLS statistic proposed by Elliot *et al.* (1996), whilst the CKP statistics are the ones proposed by Carrion-i-Silvestre *et al.* (2009) for 1, 2 and 3 breaks in both the intercept and the trend of the model.

* and ** mean rejection of the unit root null hypothesis at 10% and 5% significance level, respectively.

Table A5. Testing for unit roots. Variable: GDP (per capita)

	DF-GLS	CKP1	TB1	CKP2	TB1	TB2	CKP3	TB1	TB2	TB3
AUS	-1.49	-2.51	2009	-2.47	2009	2012	-2.97	1993	2009	2013
FRA	-1.70	-2.35	2009	-3.46	1998	2009	-4.33**	1998	2009	2017
GER	-4.81**	-2.98	2009	-3.79*	2003	2009	-4.43**	1993	2003	2009
ITA	-1.81	-2.29	2008	-2.17	2006	2009	-5.39**	2000	2008	2013
JAP	-2.89	-3.68**	2009	-4.37**	1998	2009	-5.03**	1996	2000	2009
KOR	-0.62	-1.19	1998	-4.79**	1998	2008	-4.18**	1998	2007	2010
SPA	-2.13	-1.83	2009	-2.49	2008	2014	-4.04*	1998	2007	2014
SWI	-2.16	-3.25*	2005	-3.10	2004	2009	-5.05**	1996	2002	2009
UKG	-1.34	-2.28	2008	-4.02**	1997	2009	-4.63**	1997	2006	2009
USA	-1.85	-3.02	2008	-4.58**	1998	2009	-4.32**	1995	2001	2009
BEL	-1.81	-3.05	2009	-3.77*	1993	2009	-3.15	1993	2007	2009
HUN	-3.85**	-1.97	2009	-2.23	1997	2009	-3.56	1997	2009	2014
NET	-1.76	-1.88	2009	-2.24	2009	2013	-2.31	1995	2002	2009
POL	-4.40**	-2.32	2004	-2.28	1993	2016	-3.97*	1993	2007	2016

This table presents the DF-GLS statistic proposed by Elliot et al (1996), whilst the CKP statistics are the ones proposed by Carrion-i-Silvestre et al (2009) for 1, 2 and 3 breaks in both the intercept and the trend of the model.

* and ** mean rejection of the unit root null hypothesis at 10% and 5% significance level, respectively.

Table A6: Estimation of the relations between RW and URW with GDP following Bai and Perron alternative procedure.

	Sample	Model	R^2	DW	a_0	b_0	TB_1	a_1	b_1	TB_2	a_2	b_2	TB_3	a_3	b_3
Austria	1990-19	RW	0.995	1.764	-65.5	6.05	1995	-13.43*	1.16*	2003	-16.69*	1.45*	2009	-6.71	0.52
					51.83	4.93		1.06	0.1		2.45	0.23		3.51	0.32
		URW	0.999	1.735	27.87	-2.78	1995	-8.58*	0.65*	2001	-6.04*	0.43*			
					24.74	2.35		2.03	0.19		1.51	0.14			
France	1995-19	RW	0.999	1.301	-38.70*	3.48*	2008	-37.46*	3.38*						
					1.07	0.1		2.06	0.19						
		URW	0.999	1.741	0.59	-0.15*	2002	1.1	-0.2	2008	17.81*	-1.78*	2013	9.45*	-1.00*
					0.56	0.05		3.17	0.3		6.86	0.65		1.58	0.15
Germany	1995-19	RW	0.999	2.202	-41.67*	3.83*	1999	-10.57*	0.89*	2008	-6.74*	0.54*	2013	4.24*	-0.47
					6.3	0.6		1.95	0.18		1.12	0.1		2.97	0.27
		URW	0.999	2.491	25.12*	-2.48*	2002	15.47*	-1.59*	2009	11.83*	-1.24*			
					2.37	0.22		5.15	0.48		0.88	0.08			
Italy	1995-19	RW	0.999	1.166	-109.08*	10.06*	2006	32.75*	-3.27*	2013	-39.82*	3.64*			
					10.84	1.02		3.04	0.29		2.99	0.28			
		URW	0.999	1.493	-4.02*	0.30*	2006	-32.39*	2.97*	2013	24.60*	-2.46*			
					1.07	0.1		4.31	0.41		2.1	0.2			
Japan	1990-18	RW	0.999	0.807	-82.45*	7.56*	2001	-42.40*	3.77*	2008	0.29	-0.27*	2013	9.65*	-1.16*
					14.39	1.37		6.54	0.62		0.6	0.06		0.71	0.07
		URW	0.999	2.269	2.50*	-0.33*	2000	16.07*	-1.63*	2007	4.53*	-0.55*			
					1.07	0.1		1.23	0.12		1.17	0.11			
Korea	1990-18	RW	0.999	2.678	-25.77*	2.39*	1998	-14.40*	1.25*	2009	-8.53*	0.67*			
					1.78	0.18		0.26	0.03		2.36	0.22			
		URW	0.995	2.399	21.15*	-2.27*	1997	1.09	-0.26	2002	-42.46*	4.01*	2007	-2.14	0.02
					2.13	0.22		0.75	0.58		14.05	1.37		1.23	0.12
Spain	1990-19	RW	0.997	1.186	78.44*	-8.01*	1995	-66.00*	6.14*	2009	-17.30*	1.47*			
					23.91	2.35		5.5	0.53		3.47	0.33			
		URW	0.999	2.108	-5.15*	0.43*	1996	15.54*	-1.57*	2002	-0.89	-0.02	2007	-0.84	-0.03
					1.62	0.16		3.86	0.37		10.03	0.96		0.95	0.09
Switzerland	1990-19	RW	0.999	1.188	60.67*	-5.73*	1995	-23.68*	2.05*	2001	-12.83*	1.07*	2013	8.84*	-0.88*
					15.66	1.44		1.58	0.15		1.93	0.18		3.43	0.31
		URW	0.997	1.929	-38.78*	3.48*	1995	1.23	-0.21						
					8.29	0.76		1.45	0.13						
United Kingdom	1995-19	RW	0.999	1.731	-47.35*	4.25*	2000	-93.57*	8.64*	2006	8.97	-1.00*	2012	1.21	-0.26
					5.72	0.55		3.4	0.32		4.87	0.46		1.54	0.14
		URW	0.999	2.104	-5.91*	0.50*	2002	42.65*	-4.10*	2008	14.89*	-1.52*			
					0.81	0.08		7.92	0.75		3.53	0.33			
United States	1990-18	RW	0.999	1.716	-60.02*	5.49*	1994	-12.58*	1.03*	2002	-11.40*	0.92*	2009	-5.71*	0.40*
					12.08	1.14		0.51	0.05		0.68	0.06		0.85	0.08
		URW	0.999	1.758	16.49*	-1.61*	1996	1.65	-0.21*	2007	-6.62	0.54			
					2.63	0.25		1.13	0.1		3.55	0.33			

Appendix 4

Table A1: Descriptive analysis for Total and Recycled package waste

	TPW						RPW							
	1998	2019	g (98-19)	g (98-03)	g (04-10)	g (11-19)	1998	%Tpac	2019	%Tpac	g (98-19)	g (98-03)	g (04-10)	g (11-19)
AUS	140	162	0.7%	0.4%	0.4%	1.1%	91	64.9%	106	65.4%	0.7%	0.2%	1.0%	0.9%
BEL	140	161	0.7%	2.3%	-0.2%	0.4%	89	63.5%	134	83.5%	2.0%	5.4%	0.9%	0.9%
DEN	158	169	0.3%	2.4%	-4.9%	3.4%	79	50.0%	117	69.3%	1.9%	3.9%	1.4%	1.2%
FIN	82	131	2.3%	7.5%	1.6%	-0.1%	37	44.6%	90	68.5%	4.4%	5.6%	6.1%	2.3%
FRA	194	187	-0.2%	0.5%	-0.4%	-0.4%	81	41.5%	118	63.4%	1.9%	3.4%	3.1%	0.0%
GER	172	228	1.3%	1.8%	0.6%	1.7%	137	79.7%	144	63.2%	0.2%	-0.7%	1.0%	0.1%
GRE	74	81	0.4%	4.6%	-1.5%	-0.3%	26	34.6%	49	60.1%	3.1%	3.7%	6.9%	0.0%
IRE	184	228	1.0%	2.2%	-1.1%	2.1%	27	14.8%	140	61.4%	8.1%	31.0%	2.6%	1.2%
ITA	191	216	0.6%	1.1%	-0.6%	1.3%	60	31.6%	146	67.6%	4.3%	11.4%	2.6%	1.8%
LUX	182	217	0.8%	1.3%	0.6%	0.8%	76	41.8%	155	71.5%	3.4%	8.9%	1.9%	1.7%
NET	161	170	0.3%	5.4%	-3.4%	0.4%	100	62.4%	134	78.8%	1.4%	3.3%	0.4%	1.1%
POR	101	172	2.6%	5.9%	2.3%	1.0%	35	34.8%	108	62.8%	5.5%	7.9%	7.9%	2.4%
SPA	157	170	0.4%	2.2%	-1.4%	0.8%	53	33.6%	118	69.6%	3.9%	7.4%	3.9%	2.1%
SWE	108	134	1.0%	8.0%	-5.7%	2.7%	81	74.9%	82	61.3%	0.1%	3.4%	-3.7%	1.3%
UKG*	175	178	0.1%	-0.8%	0.3%	0.4%	49	28.2%	111	62.1%	4.1%	9.8%	4.1%	0.7%
AV	148	174	0.8%	3.0%	-0.9%	1.0%	68	46.7%	117	67.2%	3.0%	7.0%	2.7%	1.2%
SD	39	40	0.8%	2.7%	2.2%	1.1%	31	18.6%	28	6.7%	2.2%	7.5%	2.9%	0.8%

This table presents the initial value, the final value, and the rates of growth of the variables for the indicated periods. The sample average (AV) and the standard deviation (SD) of the metrics are presented at the bottom of the table. g(xx-yy) indicates the cumulative growth rate between period xx and yy. Tpac and Rpac are presented in kilograms per capita and as a percentage of Tpac.
*Last observation is 2018.

Table A2: Descriptive analysis for Total and Recycled plastic packaging waste

	TPPW								RPPW							
	1998	**%Pac**	**2019**	**%Pac**	**g (98-19)**	**g (98-03)**	**g (04-10)**	**g (11-19)**	**1998**	**%Tpla**	**2019**	**%Tpla**	**g (98-19)**	**g (98-03)**	**g (04-10)**	**g (11-19)**
AUS	24	17.0%	33	20.6%	1.6%	2.3%	2.5%	0.6%	6	26.8%	10	30.8%	2.3%	5.1%	4.1%	-0.7%
BEL	21	15.3%	31	19.0%	1.7%	4.7%	1.1%	0.6%	6	26.2%	14	47.3%	4.6%	9.3%	4.7%	2.1%
DEN	32	20.5%	40	23.5%	1.0%	-2.4%	0.5%	3.3%	2	6.7%	15	37.4%	9.6%	18.3%	6.7%	7.3%
FIN	18	21.3%	24	18.4%	1.5%	-0.4%	3.4%	1.2%	2	10.2%	10	42.0%	8.6%	6.6%	12.7%	6.7%
FRA	27	14.0%	35	19.0%	1.3%	3.0%	-0.3%	1.6%	2	8.0%	10	27.0%	7.3%	18.3%	5.4%	3.0%
GER	20	11.4%	39	17.2%	3.3%	5.0%	3.9%	1.9%	12	59.1%	17	43.3%	1.8%	2.7%	3.0%	0.5%
GRE	21	28.1%	21	25.6%	0.0%	5.7%	-4.5%	0.5%	1	3.6%	8	37.6%	11.8%	1.8%	32.9%	3.0%
IRE	45	24.7%	65	28.4%	1.7%	4.3%	-4.3%	5.2%	1	2.6%	18	27.8%	13.8%	48.9%	9.2%	1.2%
ITA	32	16.6%	39	18.0%	1.0%	2.0%	0.0%	1.2%	4	11.3%	17	45.1%	7.8%	18.5%	5.3%	4.2%
LUX	22	12.0%	42	19.5%	3.2%	12.6%	1.5%	-0.4%	2	9.3%	14	33.4%	9.6%	35.4%	5.7%	0.4%
NET	32	19.8%	30	17.7%	-0.3%	0.9%	-2.8%	1.1%	5	14.2%	17	57.2%	6.6%	7.8%	10.2%	3.2%
POR	25	25.2%	41	23.5%	2.2%	4.4%	1.1%	1.9%	1	3.7%	14	35.6%	14.0%	25.2%	16.6%	6.2%
SPA	29	18.2%	36	21.0%	1.1%	3.2%	-1.5%	2.0%	2	8.6%	18	51.5%	10.1%	22.0%	4.0%	8.6%
SWE	16	14.7%	24	17.8%	2.0%	3.1%	2.0%	1.4%	4	24.6%	13	53.2%	5.8%	1.1%	7.6%	7.1%
UKG*	29	16.6%	36	19.9%	1.0%	0.7%	4.0%	-1.3%	2	7.4%	16	43.8%	10.4%	20.2%	8.5%	6.3%
AV	26	18.4%	36	20.6%	1.5%	3.3%	0.5%	1.4%	3	14.8%	14	40.9%	8.3%	16.1%	9.1%	3.9%
SD	8	4.9%	10	3.3%	1.0%	3.4%	2.7%	1.5%	3	14.6%	3	9.2%	3.7%	13.5%	7.5%	2.9%

This table presents the initial value, the final value, and the rates of growth of the variables for the indicated periods. The sample average (AV) and the standard deviation (SD) of the metrics are presented at the bottom of the table. g(xx-yy) indicates the cumulative growth rate between period xx and yy. Tpla and Rpla are presented in kilograms per capita and as a percentage of Pac or Tpla.

*Last observation is 2018.

Table A3: Descriptive analysis for Total and Recycled non-plastic package waste

	TNPPW								RNPPW							
	1998	**%Pac**	**2019**	**%Pac**	**g (98-19)**	**g (98-03)**	**g (04-10)**	**g (11-19)**	**1998**	**%Tpla**	**2019**	**%Tpla**	**g (98-19)**	**g (98-03)**	**g (04-10)**	**g (11-19)**
AUS	116	83.0%	129	79.4%	0.5%	0.0%	0.8%	1.2%	84	72.7%	96	74.3%	0.6%	-0.2%	0.6%	1.0%
BEL	118	84.7%	130	81.0%	0.4%	1.8%	-0.5%	0.4%	83	70.3%	120	92.1%	1.7%	5.1%	0.6%	0.8%
DEN	126	79.5%	129	76.5%	0.1%	3.5%	-1.5%	3.5%	77	61.2%	102	79.1%	1.4%	3.3%	1.0%	0.6%
FIN	65	78.7%	107	81.6%	2.4%	9.3%	-0.2%	-0.3%	35	53.9%	80	74.4%	4.0%	5.6%	5.7%	1.9%
FRA	167	86.0%	152	81.0%	-0.5%	0.1%	-1.1%	-0.8%	78	47.0%	109	71.9%	1.6%	2.8%	3.0%	-0.2%
GER	152	88.6%	188	82.8%	1.0%	1.3%	1.9%	1.6%	125	82.3%	127	67.3%	0.1%	-1.0%	0.8%	0.1%
GRE	53	71.9%	60	74.4%	0.6%	4.2%	-4.9%	-0.6%	25	46.8%	41	67.8%	2.4%	3.7%	5.3%	-0.5%
IRE	138	75.3%	163	71.6%	0.8%	1.5%	-2.1%	1.1%	26	18.8%	122	74.7%	7.6%	29.9%	1.8%	1.2%
ITA	159	83.4%	177	82.0%	0.5%	0.9%	-0.1%	1.3%	57	35.6%	128	72.6%	4.0%	10.9%	2.4%	1.5%
LUX	161	88.0%	175	80.5%	0.4%	-0.7%	0.2%	1.1%	74	46.3%	141	80.7%	3.1%	7.7%	1.6%	1.8%
NET	129	80.2%	140	82.3%	0.4%	6.4%	1.1%	0.3%	96	74.3%	117	83.4%	1.0%	3.1%	-0.4%	0.9%
POR	75	74.8%	132	76.5%	2.7%	6.4%	-1.7%	0.7%	34	45.4%	94	71.2%	4.9%	7.3%	7.2%	1.9%
SPA	129	81.8%	134	79.0%	0.2%	1.9%	-1.9%	0.5%	50	39.1%	100	74.4%	3.3%	6.5%	3.8%	1.2%
SWE	92	85.3%	110	82.2%	0.9%	8.8%	-5.4%	3.0%	77	83.5%	70	63.1%	-0.5%	3.5%	-4.5%	0.6%
UKG*	146	83.4%	143	80.1%	-0.1%	-1.0%	0.3%	0.9%	47	32.4%	95	66.6%	3.6%	9.2%	3.8%	0.0%
AV	122	81.6%	138	79.4%	0.7%	3.0%	-1.0%	0.9%	65	54.0%	103	74.3%	2.6%	6.5%	2.2%	0.9%
SD	36	4.9%	32	3.3%	0.8%	3.4%	2.0%	1.2%	29	19.4%	26	7.3%	2.1%	7.2%	2.8%	0.8%

This table presents the initial value, the final value, and the rates of growth of the variables for the indicated periods. The sample average (AV) and the standard deviation (SD) of the metrics are presented at the bottom of the table. g(xx-yy) indicates the cumulative growth rate between period xx and yy. Tnonpla and Rnonpla are presented in kilograms per capita and as a percentage of Pac or Tnonpla.

*Last observation is 2018.

Table A4: Descriptive analysis for per capita Gross Domestic Product (GDP)

	GDP					
	1998	**2019**	**g (98-19)**	**g (98-03)**	**g (04-10)**	**g (11-19)**
AUS	22194	39402	2.8%	3.5%	2.7%	2.4%
BEL	20277	36817	2.9%	4.6%	2.5%	2.2%
DEN	21483	39511	2.9%	3.3%	3.7%	2.2%
FIN	19617	34185	2.7%	4.0%	3.1%	1.7%
FRA	19439	33136	2.6%	3.5%	2.3%	2.2%
GER	21244	37870	2.8%	3.2%	2.7%	2.6%
GRE	15005	20556	1.5%	5.5%	1.1%	-0.3%
IRE	20869	59279	5.1%	7.4%	1.3%	6.8%
ITA	20818	30218	1.8%	2.8%	1.4%	1.5%
LUX	36789	78681	3.7%	6.1%	4.8%	1.6%
NET	23055	39724	2.6%	4.0%	2.8%	1.7%
POR	13884	24609	2.8%	4.3%	2.7%	2.0%
SPA	15893	28460	2.8%	5.3%	2.2%	1.9%
SWE	21540	37215	2.6%	3.9%	2.9%	1.7%
UKG	19689	32602	2.5%	4.8%	1.5%	1.9%
AV	20786	38151	2.8%	4.4%	2.5%	2.1%
SD	5170	14191	0.8%	1.2%	1.0%	1.5%

This table presents the initial value, the final value, and the rates of growth of GDP for the indicated periods. The sample average (AV) and the standard deviation (SD) of the metrics are presented at the bottom of the table. g(xx-yy) indicates the cumulative growth rate between period xx and yy. GDP is presented in Purchasing power parity dollars of 2020 per capita.

Table A5. Testing for unit roots. Variable: Total packages (per capita)

	DF-GLS	CKP1	TB1	CKP2	TB1	TB2	CKP3	TB1	TB2	TB3
AUS	-2.63	-1.84	2000	-5.41**	2001	2003	-4.64**	1999	2001	2003
BEL	-2.19	-3.28*	2002	-1.80	2000	2002	-4.88**	2000	2002	2008
DEN	-2.65	-3.43**	2008	-4.07**	2008	2010	-5.12**	2002	2008	2010
FIN	-1.53	-2.49	2002	-6.66**	2002	2005	-6.64**	2002	2007	2009
FRA	-2.19	-3.29*	2000	-4.50**	2000	2011	-4.63**	2000	2008	2011
GER	-2.72	-2.73	2008	-3.79**	2008	2010	-5.23**	2000	2008	2013
GRE	-2.31	-2.63	2009	-2.98	2005	2013	-4.22**	2000	2007	2013
IRE	-2.50	-3.93**	2009	-4.42**	2006	2012	-6.79**	2002	2006	2012
ITA	-1.85	-5.02**	2008	-4.54**	1999	2008	-5.62**	1999	2008	2011
LUX	-1.89	-3.27*	2008	-3.57*	2008	2013	-5.06**	2003	2008	2013
NET	-2.90	-4.10**	2005	-5.74*	2005	2014	-6.17**	1999	2005	2014
POR	-4.64**	-3.41**	1999	-7.74**	1999	2009	-10.03**	1999	2005	2010
SPA	-4.88**	-3.49**	2008	-4.65**	2007	2012	-5.34**	2000	2007	2012
SWE	-2.05	-2.66	2009	-4.64**	2002	2009	-5.06**	2002	2009	2015
UKG	-2.53	-4.54**	2001	-6.28**	2001	2011	-6.90**	2001	2010	2013

This table presents the DF-GLS statistic proposed by Elliot *et al.* (1996), whilst the CKP statistics are the ones proposed by Carrion-i-Silvestre *et al.* (2009) for 1, 2 and 3 breaks in both the intercept and the trend of the model.

* and ** mean rejection of the unit root null hypothesis at 10% and 5% significance level, respectively.

Table A6. Testing for unit roots. Variable: Total recycled packages (per capita)

	DF-GLS	CKP1	TB1	CKP2	TB1	TB2	CKP3	TB1	TB2	TB3
AUS	-3.44*	-5.49**	2000	-6.57**	2000	2008	-6.38**	1999	2001	2008
BEL	-1.32	-2.91	2002	-4.57**	2002	2008	-6.17**	2002	2007	2012
DEN	-2.52	-2.95	2010	-4.45**	2010	2016	-5.67**	1999	2010	2016
FIN	-1.00	-4.41**	2005	-3.49	2004	2008	-4.58**	2004	2008	2012
FRA	-0.35	-2.94*	2016	-4.10**	2007	2016	-4.83**	2006	2009	2016
GER	-1.76	-2.41	2009	-4.24**	2004	2016	-5.75**	2000	2005	2016
GRE	-1.80	-3.85**	2011	-3.66**	2010	2012	-6.97**	2004	2011	2014
IRE	-0.91	-3.36*	2007	-4.34**	2002	2007	-4.32**	1999	2002	2007
ITA	-0.78	-5.16**	2000	-7.27**	1999	2002	-8.20**	1999	2002	2008
LUX	-1.71	-3.16*	2008	-5.86**	2006	2013	-5.68**	2006	2009	2013
NET	-2.04	-4.10**	2008	-7.51**	2004	2014	-8.53**	2004	2014	2016
POR	-2.17	-4.48**	2009	-5.02**	1999	2009	-2.24	1999	2005	2009
SPA	-1.03	-2.80	2008	-3.54*	2006	2013	-6.44**	2000	2007	2013
SWE	-1.52	-4.84**	2009	-5.07**	2002	2009	-5.51**	1999	2003	2009
UKG	-1.15	0.14	2006	-4.84**	2007	2015	-6.59**	2007	2011	2015

This table presents the DF-GLS statistic proposed by Elliot *et al.* (1996), whilst the CKP statistics are the ones proposed by Carrion-i-Silvestre *et al.* (2009) for 1, 2 and 3 breaks in both the intercept and the trend of the model.

* and ** mean rejection of the unit root null hypothesis at 10% and 5% significance level, respectively.

Table A7. Testing for unit roots. Variable: Plastic packages (per capita)

	DF-GLS	**CKP1**	**TB1**	**CKP2**	**TB1**	**TB2**	**CKP3**	**TB1**	**TB2**	**TB3**
AUS	-0.49	-3.87**	2012	-5.33**	2000	2012	-4.53**	1999	2002	2012
BEL	-1.59	-4.04**	2007	-9.11**	2002	2007	-6.67**	2000	2003	2007
DEN	-1.56	-2.32	2007	-7.38**	2001	2007	-10.45**	2001	2007	2013
FIN	-2.01	-2.57	2007	-3.56*	2007	2015	-4.70**	2004	2007	2015
FRA	-1.86	-3.04	2008	-6.38**	2007	2012	-4.52**	2007	2009	2012
GER	-0.72	-4.99**	2005	-4.78**	2005	2008	-6.88**	2005	2008	2010
GRE	-3.71**	-2.62	2007	-5.05**	2003	2013	-4.83**	2004	2007	2013
IRE	-2.19	-3.78**	2009	-5.64**	2005	2012	-5.70**	2005	2010	2013
ITA	-2.57	-2.59	2008	-3.62*	2007	2013	-4.35**	2006	2008	2013
LUX	-0.40	-3.52**	2002	-3.02	2002	2007	-6.64**	2002	2007	2012
NET	-1.57	-2.65	2005	-4.81**	1999	2005	-8.86**	1999	2005	2008
POR	-3.49**	-4.92**	1999	-8.92**	1999	2009	-11.04**	1999	2007	2012
SPA	-2.82	-3.47**	2008	-6.88**	2007	2011	-7.60**	2006	2009	2013
SWE	-1.17	-4.14**	2010	-5.25**	2000	2012	-6.68**	2000	2008	2013
UKG	-2.49	-2.87	2012	-4.53**	1999	2012	-5.91**	1999	2008	2012

This table presents the DF-GLS statistic proposed by Elliot *et al.* (1996), whilst the CKP statistics are the ones proposed by Carrion-i-Silvestre *et al.* (2009) for 1, 2 and 3 breaks in both the intercept and the trend of the model.

* and ** mean rejection of the unit root null hypothesis at 10% and 5% significance level, respectively.

Table A8. Testing for unit roots. Variable: Recycled plastic packages (per capita)

	DF-GLS	**CKP1**	**TB1**	**CKP2**	**TB1**	**TB2**	**CKP3**	**TB1**	**TB2**	**TB3**
AUS	-0.17	-1.91	2012	-3.61*	2006	2012	-8.36**	2001	2006	2014
BEL	-2.57*	-4.16**	2003	-3.14	2000	2004	-6.00**	2000	2005	2012
DEN	-0.99	-3.65**	2015	-6.29**	2009	2015	-7.68**	2005	2010	2015
FIN	-4.10**	-1.99	2016	-4.70**	2007	2016	-5.35**	2007	2012	2016
FRA	-0.03	-2.45	2007	-3.42*	2007	2016	-3.65	2004	2006	2016
GER	-2.44	-2.62	2007	-4.35**	2007	2016	-3.59	2004	2008	2016
GRE	-2.43	-2.55	2008	-3.22	2008	2011	-6.04**	2003	2008	2011
IRE	-0.86	-2.92	2009	-4.80**	2009	2013	-6.75**	2003	2007	2013
ITA	-1.83	-0.13	2001	-5.19**	2003	2012	-8.00**	2001	2008	2014
LUX	-1.49	-3.92**	2007	-4.72**	2006	2009	-3.09	2003	2006	2009
NET	-2.72	-3.35*	2009	-4.66**	2001	2009	-5.10**	2001	2009	2013
POR	-0.72	-4.14**	2013	-4.56**	2012	2016	-4.71**	2006	2009	2016
SPA	-0.89	-3.20*	2008	-3.86**	2001	2012	-5.95**	1999	2007	2012
SWE	-1.13	-2.47	2012	-3.75*	2005	2012	-5.34**	1999	2005	2012
UKG	-1.19	-4.08**	2013	-4.21**	2013	2015	-6.99**	2002	2010	2015

This table presents the DF-GLS statistic proposed by Elliot *et al.* (1996), whilst the CKP statistics are the ones proposed by Carrion-i-Silvestre *et al.* (2009) for 1, 2 and 3 breaks in both the intercept and the trend of the model.

* and ** mean rejection of the unit root null hypothesis at 10% and 5% significance level, respectively.

Table A9. Testing for unit roots. Variable: Non-plastic packages (per capita)

	DF-GLS	CKP1	TB1	CKP2	TB1	TB2	CKP3	TB1	TB2	TB3
AUS	-1.94	-4.37**	2003	-4.15**	2001	2003	-3.06	1999	2001	2003
BEL	-2.17	-3.51**	2002	-1.92	2000	2002	-4.85**	2000	2002	2008
DEN	-2.78	-3.33*	2008	-4.37**	2008	2010	-5.82**	2002	2008	2010
FIN	-1.74	-4.21**	2002	-5.38**	2002	2005	-6.16**	2002	2007	2009
FRA	-2.20	-3.97**	2000	-4.88**	2000	2011	-5.85**	2000	2011	2016
GER	-1.81	-2.69	2008	-3.97**	2008	2014	-5.25**	2000	2008	2013
GRE	-3.44**	-2.91	2009	-3.84**	2007	2013	-5.23**	2000	2007	2013
IRE	-2.72	-3.82**	2009	-3.90**	2002	2009	-8.02**	2002	2007	2012
ITA	-1.90	-5.20**	2008	-4.87**	1999	2008	-5.52**	1999	2008	2011
LUX	-1.66	-5.83**	2008	-6.19**	2008	2013	-5.60**	2007	2009	2013
NET	-4.19**	-3.97**	2005	-5.42**	2005	2014	-6.33**	1999	2005	2014
POR	-4.45**	-3.04*	1999	-6.55**	1999	2010	-9.90**	1999	2005	2010
SPA	-4.88**	-3.62**	2008	-4.60**	2002	2011	-5.75**	2000	2005	2012
SWE	-1.98	-2.70	2009	-4.48**	2002	2009	-5.70**	2002	2009	2015
UKG	-1.95	-4.18**	2013	-1.92	2001	2013	-8.30**	1999	2004	2013

This table presents the DF-GLS statistic proposed by Elliot *et al.* (1996), whilst the CKP statistics are the ones proposed by Carrion-i-Silvestre *et al.* (2009) for 1, 2 and 3 breaks in both the intercept and the trend of the model.

* and ** mean rejection of the unit root null hypothesis at 10% and 5% significance level, respectively.

Table A10. Testing for unit roots. Variable: Recycled non-plastic packages (per capita)

	DF-GLS	CKP1	TB1	CKP2	TB1	TB2	CKP3	TB1	TB2	TB3
AUS	-2.78	-4.95**	2000	-4.65**	1999	2001	-6.19**	1999	2001	2008
BEL	-1.39	-3.04	2002	-4.89**	2002	2008	-5.37**	2000	2002	2008
DEN	-2.72	-2.95	2010	-4.43**	2010	2016	-5.67**	1999	2010	2016
FIN	-0.99	-4.60**	2005	-3.61*	2004	2008	-3.89*	2004	2008	2012
FRA	-0.47	-3.20**	2016	-3.97**	2007	2016	-4.63**	2006	2009	2016
GER	-4.66**	-2.59	2009	-3.74**	2002	2016	-5.78**	2000	2005	2016
GRE	-2.05	-3.65**	2011	-3.29	2010	2012	-2.94	2010	2013	2016
IRE	-1.28	-3.44**	2007	-4.37**	2002	2007	-4.53**	1999	2002	2007
ITA	-0.44	-2.68	1999	-7.82**	1999	2002	-8.54**	1999	2002	2008
LUX	-1.86	-3.57**	2008	-5.98**	2006	2013	-8.82**	2000	2006	2013
NET	-2.46	-3.93**	2008	-6.99**	2005	2014	-8.32**	2002	2014	2016
POR	-2.26	-4.41**	2009	-4.74**	1999	2009	-1.86	1999	2005	2009
SPA	-1.07	-2.69	2008	-3.39	2006	2013	-6.29**	2000	2007	2013
SWE	-0.83	-4.91**	2009	-5.04**	2002	2009	-5.63**	1999	2003	2009
UKG	-2.79	-3.48**	2006	-6.96**	2007	2015	-7.89**	2007	2011	2015

This table presents the DF-GLS statistic proposed by Elliot *et al.* (1996), whilst the CKP statistics are the ones proposed by Carrion-i-Silvestre *et al. (*2009) for 1, 2 and 3 breaks in both the intercept and the trend of the model.

* and ** mean rejection of the unit root null hypothesis at 10% and 5% significance level, respectively.

Table A11. Testing for unit roots. Variable: GDP (per capita)

	DF-GLS	CKP1	TB1	CKP2	TB1	TB2	CKP3	TB1	TB2	TB3
AUS	-0.91	-2.99	2008	-3.21	2000	2008	-4.37**	2000	2008	2012
BEL	-1.13	-3.26*	2008	-3.67*	1999	2008	-5.92**	1999	2008	2012
DEN	-1.09	-2.97	2008	-3.18	2002	2008	-5.27**	2002	2008	2011
FIN	-1.21	-2.15	2008	-3.39	2008	2013	-4.14**	2006	2008	2013
FRA	-1.22	-2.40	2008	-3.46	2008	2016	-5.66**	2002	2008	2016
GER	-2.24	-3.80**	2008	-3.54*	2008	2011	-6.05**	2006	2008	2011
GRE	-1.14	-1.75	2009	-4.50**	2008	2011	-6.06**	2000	2008	2011
IRE	-1.68	-2.03	2014	-5.28**	2007	2014	-5.11**	2006	2009	2014
ITA	-1.60	-1.62	2008	-3.29	2008	2013	-5.42**	2003	2008	2012
LUX	-0.76	-3.93**	2005	-2.40	2005	2008	-4.85**	2002	2008	2014
NET	-1.17	-2.15	2008	-2.71	2002	2008	-6.10**	2002	2008	2016
POR	-1.35	-3.13*	2010	-3.80**	2008	2011	-2.78	2008	2010	2012
SPA	-1.13	-2.02	2008	-3.77**	2008	2012	-5.35**	2005	2008	2012
SWE	-1.06	-3.55**	2008	-4.40**	2006	2008	-4.62**	2004	2007	2009
UKG	-1.63	-3.33*	2008	-5.94**	2003	2008	-6.72**	1999	2005	2008

This table presents the DF-GLS statistic proposed by Elliot *et al.* (1996), whilst the CKP statistics are the ones proposed by Carrion-i-Silvestre *et al.* (2009) for 1, 2 and 3 breaks in both the intercept and the trend of the model.

* and ** mean rejection of the unit root null hypothesis at 10% and 5% significance level, respectively.