Analyzing the economic development-driven ecological deficit in the EU-15 countries: new evidence from PSTR approach

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Abstract
This paper empirically analyzes the non-linear effect of economic activities on ecological balance indicators that estimate the balance between economies’ pressure on nature and the biologically productive resource areas affected by human activity and the earth’s ecological carrying capacity. In measuring this balance, ecological balance sheet indicators are divided into four sub-components: cropland, fishing grounds, forest area, and grazing land. The sample of the study consists of the EU-15 countries over the period 1995–2016. To render the study robust with respect to econometric issues such as potential endogeneity bias, cross-country heterogeneity, non-linearity, and time instability, the panel smooth transition regression (PSTR) method is adapted. The empirical findings reveal that up to a certain threshold level, economic activities do not affect the ecological balance as nature can compensate for the resulting externalities, but beyond this threshold, waste accumulation and pollution exceed nature’s capacity to absorb. Consequently, these findings do not empirically support the EKC hypothesis with an inverted U-shaped curve and suggest that active environmental policies are needed to improve the environment.

Introduction
Evidence regarding the interrelation between ecological and economic systems shows that economic activities and ecological changes have mutual effects on each other. In particular, disposal of the waste products of economic activities can be damaging to the ecology and interfere with the mutually supporting interactions within the ecological system. With the increasing pace and volume of economic activity ever increasing in volume, the burden on the ecological system is becoming correspondingly greater. Consumption of resources faster than the rate of renewal destroys the natural ecosystems on which the human life and biodiversity depend. As a result, many negative externalities contributing to global climate change such as decreasing forest areas, exhaustion of freshwater systems, and increased carbon dioxide emissions are becoming increasingly damaging (Özsoy and Dinç 2016). In recognition of this process, countries’ goals should encompass both economic growth as traditionally measured and the sustainability of the process as well. Economic policies aiming at increasing welfare must therefore consider the economy and the environment as interdependent systems. In other words, it is crucial for countries, regions, and any global initiative to understand that economic actions create pressure on ecosystems while they create economic growth and take into consideration this pressure before taking any policy decisions.

Human beings leave a mark on the earth while carrying out the production and consumption activities throughout their life. However, as they carry out these activities, they...
little notice the spillover effects on nature and the resulting pressure on the ecosystem, with the consequence that the ecological carrying capacity may be exceeded; such over-utilization in order that the biological capacity (biocapacity) be preserved to create a balance between the resource requirements of current and future generations. In other words, to achieve sustainability, natural resource consumption should not exceed the volume of the resources that the earth can regenerate within a given time frame. The earth, where all living things continue their lives, is inadequate to meet the needs due to the use of resources above the biological capacity (BC)—defined as the capacity of a geographical region to produce renewable natural resources. For this target, the ecological footprint (EF) measurements are seen as a vital measurement tool in the preparation of environmentally friendly policies required while ensuring economic development.

Following pioneering studies by Rees (1992) and Rees and Wackernagel (1994), the ecological footprint measure is being increasingly employed in studies applied to geographic regions, countries, and specific productive activities. Given prevailing technology and resource management practices, the EF, which measures the amount of the biologically productive soil and water areas (cropland, forest land, grazing land, built-up land, fishing grounds, and carbon uptake land) required to reproduce the natural resources consumed by individuals, countries, or activities in a given period and to absorb the waste creates, provides insight into how far the limits of the carrying capacity of the earth have been exceeded, rather than giving precise judgments. In this respect, the comparison of EF and BC indicates whether the earth can live within the limits of self-renewal.

An ecological deficit occurs when the EF of a country’s (a particular person, society, or economy) population exceeds the bio-capacity of that country’s area. Under deficit conditions, natural resources are exhausted, environmental problems appear and it becomes difficult for a population to meet its needs. In contrast, an ecological reserve exists whenever a population’s EC is below its region’s biocapacity.

The debate on whether ecological resources are sufficient to attain sustainable economic growth dates back to the beginning of the 1970s. The Club of Rome’s “Limits to Growth” approach, which claims that environmental quality will deteriorate with economic activity, has resonated in political and academic platforms and has become an effective argument. It is argued that the earth cannot provide the need for natural resources, clean, accessible water, and fresh air if the population growth and economic development continue in this way (Meadows et al. 1972). These debates are a considerable warning in terms of drawing attention to the damage caused by economic activities on the earth’s ecosystems and natural resources. This approach has also sparked the wick of a consensus on acting on a common platform against environmental issues.

However, the findings that emerged in many studies on the potential environmental impacts of economic growth such as Grossman and Krueger (1991, 1995), Beckerman (1992), Shafik and Bandyopadhyay (1992), Panayotou (1993), and Selden and Song (1994) in the early 1990s have led to doubts about the Limits of Growth paradigm, and new arguments have been put forward. This approach suggests that there is no positive or negative linear relationship between economic activities and environmental quality; in fact, environmental quality follows an inverted U-shaped pattern that initially falls and then rises as a function of per capita income. The basic notion of this approach which is also referred to as the environmental Kuznets curve (henceforth EKC) hypothesis in the literature is that as an economy develops, environmental deteriorations initially increase during the early development or industrialization stages and then tend to fall as they reach higher income per capita due to awareness of people and increased demand for a clean environment.

The fact that many natural disasters (such as hurricanes, floods, droughts, extreme cold, and heat), which are thought to be caused by global climate changes in recent decades, and the effect of human-made activities on environmental degradation led to re-thinking on the economic growth process and re-questioning the EKC hypothesis. Some researchers who find it wrong to have very optimistic expectations about the future of the world by looking at the data that environmental quality is improving in developed countries draw attention to the global pollution, the situation of developing countries, and the pollutants that lead to irreversible consequences.

Although this approach is interesting in terms of modeling the environment growth relationship or providing an alternative perspective on explaining the relationship, the EKC hypothesis and its policy implications have begun to be questioned and criticized by many theoretical and empirical studies seeking answers to the question of how to achieve sustainable development. Criticisms of the EKC approach may generally be grouped under three headings.

The first point of these criticisms is that the shape of a typical EKC curve is based on the assumption that the environmental pollution is not cumulative or that its effects can be reversed (Tisdell 1993, 2001; Arrow et al. 1995; Bimonte 2002; Tao et al. 2008; Czech 2008; Fodha and Zaghboud 2010; Aydin et al. 2019a; Esen et al. 2020). Some pollutants are cumulative, that is, they tend to accumulate in the environment (Tisdell 1993, 2000). Furthermore, even if pollutants do not accumulate initially, they may begin to accumulate after a
certain threshold capacity level is exceeded. In addition, the direct effects of some pollutants on the environment may be relatively small individually, but major environmental effects may occur when combined with others (Cooper 2004; Solomon et al. 2016). Therefore, cumulative impacts may be difficult to predict and monitor due to insufficient environmental data, nature’s reaction taking relatively long times, and complex ecological processes (Clark 1994). Also, even if environmental investments increase as incomes rise, resources such as biodiversity may not be renewed to the extent that they are lost. In this case, biodiversity loss is, on any reasonable scale, essentially irreversible and monotonic as there is no threshold point as a function of income per capita (Tisdell 1993; Asafu-Adjaye 2003; Dietz and Adger 2003; Simpson et al. 2005; Czech 2008; Mills and Waite 2009; Iritié 2015; Ruiz–Agudelo et al. 2019).

The second consideration is that the results of empirical studies on EKC may be valid for pollutants that have local and regional dimensions but not at the global level and their negative impact can be controlled with only limited effects on economic growth. Critics of the EKC often argue that these are the pollutants that involve local, direct, and short-term costs, which have destructive effects which are limited to the environment of the area where they are being released (for example, SOx, NOx, SPM). However, the EKC-type relation may not be meaningful for resource stocks, for the accumulated stocks of waste, or for pollutants involving relatively long-term and more dispersed costs (such as CO2), which are often increasing functions of income (e.g., Arrow et al. 1995; Holtz–Eakin and Selden 1995; Stern et al. 1996; Panayotou 1997; Cavlovic et al. 2000; Lieb 2003; York et al. 2003; Salvati and Zitti 2008; Esen et al. 2020).

Thirdly, studies have been critical of the EKC evidence on the grounds that the functional specifications and econometric techniques employed fail to capture the actual shape of the environment–income relationship (Stern 1998; Bagliani et al. 2008; Romero-Ávila 2008; Galeotti et al. 2009; Esteve and Tamarit 2012; Chiu 2012; Aydin et al. 2019a; Esen et al. 2020). Most of the studies cited so far have usually adopted reduced-form models, in which the pollutants per capita are a quadratic or cubic function of the real income per capita. In these studies, the aim is to find the potential non-linearity of the underlying function with strategies such as logarithmic transforms or cubic functions (He and Richard 2010). However, such an approach may not have the flexibility to determine the true form of the relationship. Imposing any parametric (e.g., linear, non-linear, quadratic, or cubic) relationship in advance or predetermining the types of models can lead to the selection of inappropriate models that can lead to biased findings.

The emergence of a series of environmental disasters such as global warming and climate change and the increase in public support for environmental development have forced policy-makers and legislators to take action at the international level. In this process, the EU is at the forefront of international efforts to promote economically, environmentally, and socially sustainable development to address environmental issues. Environmental policies for the protection of the ecosystem are at the center of EU policy. In terms of environmental regulations, the community legislation of the EU sets the basic standards that are mandatory for all member states. However, members can often take more stringent measures at the national level, provided that they do not hinder the common internal market. In addition, the EU plays an important role in the preparation and implementation of multilateral environmental agreements and other environmental negotiations and processes (Alberton 2012).

To sum up, the aim of this paper is to empirically examine the effect of income per capita on the ecological deficit by using a large dataset for the EU-15 countries—Austria, Belgium, Denmark, Finland, France, Germany, Greece, Ireland, Italy, Luxembourg, Netherlands, Portugal, Spain, Sweden, and United Kingdom (UK)—covering the period 1995–2016. The starting period of the study is determined by taking into account the White Paper “An energy policy for the European Union” outlining a common energy and environmental policy among the member states, adopted in 1995 (European Commission 1995). To obtain a homogenous group of countries, each with a similar development status and policy approach, the sample involves the EU-15 countries that have taken common actions with regard to energy and environmental policy during the sample period. Failure to identify the appropriate country group in panel data analysis methods can have a seriously misleading effect on the findings and their policy implications, a problem this study avoids by employing data for countries following similar policies.

More specifically, this paper makes two innovative contributions to ongoing debates regarding the long-run analysis of the EKC hypothesis in the literature. Firstly, the inverse U-shaped pattern may be valid only for certain types of pollutants, especially atmospheric ones, while not being valid for the accumulation of stocks of waste. It is also claimed that the EKC-type relation is likely to be more limited or weak where the feedback effects of resource stocks such as those containing soil and its cover, forests, and other ecosystems are significant (Arrow et al. 1995). To provide a more complete perspective, this study adopts both the total ecological deficit and its disaggregated components as new indicators of environmental degradation, which reveals the quantity of pressure on natural resource stocks (ecological assets) and where they originate, comparing the ecological footprint from human activities with the number of natural resources that can be produced in the same period, that is, with biological capacity. It is an important environmental indicator that determines to what
extent human activities exceed basically two types of environmental limits, such as resource production and waste absorption (Wackernagel et al. 2004; Schaefer et al. 2006; Rugani et al. 2014). To contribute to the determination of the areas where the ecological deficit is concentrated and to better plan the efficient use of resources, the present study examines the ecological deficit both in total and in the sub-components, separated according to the major types of ecologically productive areas—cropland, grazing land, forest area, fishing grounds, and carbon land.

In this study, non-linear panel data analysis will be used to investigate the effects of economic activities on ecological balance indicators. Non-linear relationships in panel data analysis were first tackled with the panel threshold regression (PTR) model developed by Hansen (1999). However, the PTR approach assumes that parameters change abruptly between regimens. Such an approach may not always be valid in the relationship between economy and environment because economic activities disrupt the ecosystem over time. This means that the predicted parameters do not change abruptly but smoothly. Secondly, in contrast to traditional non-linear threshold approaches, this paper employs a panel smooth transition regression (PSTR) model as an innovative econometric technique that can estimate the threshold level endogenously and also the smoothness of the transition from a low-income regime to a high-income regime. Another advantage of this model is its ability to consider econometric issues such as potential endogeneity biases, non-linearity, cross-country heterogeneity, and time instability (Chiu 2012; Wu et al. 2013). To the best of the authors’ knowledge, this paper is the first to apply a PSTR model to the linkages between income and an ecological deficit. The motivation behind the choice of the PSTR model is based on the fact that contrary to the econometric methods applied in previous studies, this empirical method puts forward a strong solution to the EKC hypothesis and its empirical research, especially methodological criticisms.

To this end, the remainder of this paper is organized as follows: The “Literature review” section presents a review of the literature on the environment–income relationship. The “Methodology” section describes relevant data, the methodology, and the empirical model used in the study. The “Model, data, empirical results, and implications” section provides and discusses the empirical findings and the “Conclusion” section concludes.

**Literature review**

Empirical findings in the literature testing the validity of the EKC differ and are quite mixed, depending on the types of pollutants selected, samples of countries/regions and time periods studied, the econometric techniques applied, and other explanatory variables used in the model. Among these studies, Panayotou (1993), Selden and Song (1994), Canas et al. (2003), Kahuthu (2006), Jalil and Mahmud (2009), Fodha and Zaghoud (2010), Iwata et al. (2010), Leitão (2010), Nasir and Rehman (2011), Saboori et al. (2012), Shahbaz et al. (2013), Tiwari et al. (2013), Lau et al. (2014), López-Menéndez et al. (2014), Shahbaz et al. (2014), Al-Mulali et al. (2015), Apergis and Ozterk (2015), Balaguer and Cantavella (2016), Li et al. (2016), Jibli et al. (2016), Wang et al. (2016), Apergis (2016), Ahmad et al. (2017), Solarin et al. (2017), Moutinho et al. (2017), Luo et al. (2017), and Dogan and Inglesi-Lotz (2020) have provided evidence of the validity of the EKC hypothesis and confirmed the inverted U-shape, whereas Agras and Chapman (1999), Harbaugh et al. (2002), Cole (2003), Richmond and Kaufmann (2006), Caviglia-Harris et al. (2009), He and Richard (2010), Kearsley and Riddel (2010), Ozturk and Acaravci (2010), Pao et al. (2011), Roca et al. (2001), Azlina et al. (2014), Baek (2015), Ozturk and Al-Mulali (2015), Robalino-López et al. (2015), Katz (2015), Zoundi (2017), Ozokcu and Ozdemir (2017), Liu et al. (2017), Aydin et al. (2019b), Yilanci and Pata (2020), and Erdogan et al. (2020) have found either no evidence or weak evidence in support of an inverted U-shape. However, most of the aforementioned studies (and many others) in the literature on EKC cited so far use either a specific pollution scale such as SOx, NOx, and SPM or a global pollution scale such as CO2 as indicators of environmental quality. The problem with focusing on these particular pollutants is that they represent only a small part of total environmental issues (Al-Mulali et al. 2015; Hervieux and Darné 2015; Destek et al. 2018; Imamoglu 2018).

In the literature, the association between income and environment is widely researched; however, the availability of studies related to an ecological deficit or its derivatives, which are considered to represent the overall human impact on the earth’s ecosystem and the current state of stock resources relatively more accurately, remains limited. Among these studies, York et al. (2004) examined the cross-national variation in the ecological footprint (EF) per unit of income utilizing data on 139 countries in 1999. The findings reveal that economic development leads to greater environmental impacts and is unlikely to ensure sustainability. A study by Hervieux and Darné (2015) analyzes the EKC hypothesis using traditional linear, quadratic, and cubic functions, with standard and logarithmic specifications using time series analysis for seven Latin American countries covering the 1961–2007 period. Similarly, the finding that environmental degradation does not improve when income increases emphasizes that the EKC hypothesis is not valid for EF. Adopting the panel
smooth transition regression (PSTR) model, Aydin et al. (2019a) have found only weak evidence of an inverted U-shaped relationship between EF and income. This evidence validates the EKC hypothesis in 26 EU countries. Similar findings were found in 141 countries by Bagliani et al. (2008) utilizing ordinary least squares (OLS) and weighted least squares (WLS) analysis on linear, quadratic, and cubic functions, in standard and logarithmic specifications, in 146 countries by Caviglia-Harris et al. (2009), in 150 nations with populations over 1 million by Wang et al. (2013) using a spatial econometric approach, and in 94 countries by Paolo Miglietta et al. (2017) using the OLS on linear and non-linear models.

In contrast, Aşçi and Acar (2016) examine the economy–environment relationship using the panel data set of 150 countries’ EF over the period 2004–2008. The findings confirm that there is an EKC-type relationship between per capita income and the footprint of domestic production. Similar results were found in Qatar by Mrabet and Alsamara (2017) employing the autoregressive distributed lag (ARDL) model with the presence of unknown structural breaks, by Charfeddine and Mrabet (2017) for a sample of MENA countries using panel cointegration and causality analysis, finds evidence in favor of an inverted U-shaped curve, in 15 EU countries by Destek et al. (2018) adopting second generation panel data methodologies which take into account the cross-sectional dependence among countries and in 11 newly industrialized countries by Destek and Sarkodie (2019) employing both augmented mean group (AMG) estimator and the heterogeneous panel causality method. Apart from these, Al-Mulali et al. (2015) found an EKC-type relation in upper-middle- and high-income countries, but not in low- and lower-middle-income countries, in a study of 93 countries using fixed effects and generalized moments (GMM) models, similar to Uddin et al. (2016) which provides evidence in support of the EKC hypothesis in 10 countries of 22, but in others either absent or weak evidence.

Ecological deficit measures are of great importance in terms of determining whether the earth’s biological resources and ecosystem services are used within the boundary of self-renewal and creating a scientific basis for effective and feasible solutions to eliminate the imbalance caused by today’s excessive consumption. To the best of our knowledge, however, in reviewing the literature, it is seen that there is no study in the framework of the EKC hypothesis that takes into account the ecological deficit variable as an indicator of environmental quality.

**Methodology**

In this study, the non-linear relationship between ecological balance and economic growth was investigated using the panel smooth transition regression. Such a relationship was examined based on the model used by Bagliani et al. (2008), Al-mulali et al. (2015), Uddin et al. (2017), Aydin et al. (2019a), and Ulucak et al. (2020). The model is explained in detail by Aydin et al. (2019a) and is presented in Eq. (1);

\[ EB_{it} = \beta_0 + \beta_1 Growth_{it} + \theta X_{it} + \varepsilon_{it} \]  

(1)

where \( EB \) represents ecological balance; \( growth \) is the GDP growth rate, \( X \) represents other macroeconomic variables that might have an impact on ecological balance; \( \varepsilon \) is the error term; \( i = 1, 2, \ldots, N \) countries.

The panel threshold regression (PTR) model was introduced by Hansen (1999) as the first regression model analyzing the non-linear relationship among the variables in panel data as part of regime-switching econometric models. In this method, the effect of the threshold variable on the dependent variable may vary in regimes below or above the threshold. This situation causes the slope parameters to differ according to the regime-switching mechanism that depends on the threshold variable. In the PTR approach, it is assumed that slope parameters vary suddenly and each regime differs with respect to the detected threshold value. However, it may not be possible to observe these sudden changes among the regimes in economic models (Güloğlu and Nazloğlu 2013).

This approach, in evaluating the relationship between ecological balance and per capita real GDP, classifies the countries included in the panel with respect to their per capita real GDP values and estimates a different set of parameters for each group. As a result, it assumes that there are certain differences between developed countries with high per capita real GDP and developing countries with low per capita real GDP. In this setup, it is possible for a developing country which is near the threshold to become suddenly a developed country. However, in reality, this switch does not happen in one period but happens over a longer period of time. Thus, estimated parameters change not instantly but smoothly. Therefore, this study employs the panel smooth transition regression (PTSR) approach which allows gradual transition of parameters from one regime to another. The method was developed by González et al. (2005).

To examine the non-linear relationship between ecological balance and economic growth, we transformed the model shown in Eq. (1) into a two-regime fixed PTSR model shown in Eq. (2):

\[ EB_{it} = \mu_i + \beta_0 LN GDP_{it} + \beta_1 LN GDP_{it} * g(q_{it}; \gamma, \theta) + \varepsilon_{it} \]  

(2)
where $EB$ represents log-transformed per capita ecological balance; $\text{LnGDP}$ is log-transformed per capita real GDP; $\varepsilon$ is the error term; $t = 1, 2, \ldots, T$ denotes the time period; and $i = 1, 2, 3, \ldots, N$ are countries. Coefficients $\mu_i$ allow for the possibility of unit-specific fixed effects and the variable $q_i$ is a potential threshold variable. In Eq. (2), $g(q_i, \cdot; \gamma, \theta)$ is used as a transition function and is shown in the form of the logistic function, as in Eq. (3):

$$
g(q_i; \gamma, \theta) = \left[1 + \exp\left(-\gamma(q_i - \theta)\right)\right]^{-1} \quad (3)
$$

In Eq. (3), while the parameter of $\theta$ is a threshold parameter between two regimes which are represented by $g(q_i, \cdot; \gamma, \theta) = 0$ and $g(q_i, \cdot; \gamma, \theta) = 1$, $\gamma$ (smoothness parameter) represents the smoothness measure of the change in the value of the transition function. In other words, it reflects the smooth transition from one regime to another. As the smoothness parameter approaches infinity ($\gamma \to \infty$), transition from 0 to 1 does not happen instantly, unlike the PTR approach in which $\theta$ is the threshold parameter and the switching from one regime to another happens abruptly. In this situation, the model is estimated using the PTR approach. On the other hand, as the smoothness parameter approaches to zero ($\gamma \to 0$), the transition function turns out to be a constant number and the estimation model is reduced to linear form. In this situation, the model is estimated by using a panel within the estimator (Fouquau et al. 2008; 287-288).

As the transition function is a continuous function of the transition variable, it can take on values between 0 and 1. In Eq. (1), the regression coefficient becomes $\beta_0$ when the transition function is equal to zero ($g(q_i, \cdot; \gamma, \theta) = 0$) and $g(q_i, \cdot; \gamma, \theta) = \beta_0 + \beta_1$ when the transition function is equal to one ($g(q_i, \cdot; \gamma, \theta) = 1$). On the other hand, the regression coefficient becomes the weighted average of $\beta_0$ and $\beta_1$ when the value of the transition function is between zero and one ($0 < g(q_i, \cdot; \gamma, \theta) < 1$). Therefore, it is preferable to interpret the signs of the coefficients in the PSTR model rather than interpret the coefficients directly (Fouquau et al. 2008; 287-288). In other words, the positive or negative impact of the independent variable on the dependent one is estimated, hence allowing varying elasticities with respect to different time periods to be interpreted (Güloğlu and Nazlıoğlu 2013).

PSTR can be with two regimes as well as with more than two regimes. A version of the PSTR model with more than two regimes is given in Eq. (4). The transition function with more than two regimes in the PSTR model is given in Eq. (5) below.

\begin{equation}
EB_{i,t} = \mu_i + \beta_0 \text{LnGDP}_{i,t} + \sum_{j=1}^{r} \beta_j \text{LnGDP}_{i,t} * g_j(q_{i,t}; \gamma_j, \theta_j) + u_{i,t} \quad (4)
\end{equation}

\begin{equation}
g(q_{i,t}; \gamma, \theta) = \left[1 + \exp\left(-\gamma \Pi_{j=1}^{m}(q_{i,t} - \theta_j)\right)\right]^{-1}, \gamma > 0, c_1 \leq c_2 \leq \ldots \leq c_m \quad (5)
\end{equation}

Equation (6) provides the formula to calculate elasticity measure when the transition (threshold) variable ($q$) is different than the explanatory variable ($q = \text{LnGDP}_{i,t}$) in the PSTR model with more than two regimes.

\begin{equation}
e_{i,t} = \frac{\partial E_{i,t}}{\partial \text{LnGDP}_{i,t}} = \beta_0 + \sum_{j=1}^{r} \beta_j \cdot g_j(q_{i,t}; \gamma_j, \theta_j) \quad (6)
\end{equation}

Equation (7) provides the formula to calculate the elasticity measure when the transition (threshold) variable ($q$) is equal to the explanatory variable ($q = \text{LnGDP}_{i,t}$) in the PSTR model with more than two regimes.

\begin{equation}
e_{i,t} = \frac{\partial E_{i,t}}{\partial \text{LnGDP}_{i,t}} = \beta_0 + \sum_{j=1}^{r} \beta_j \cdot g_j(q_{i,t}; \gamma_j, \theta_j) + \sum_{j=1}^{r} \frac{\partial g_j(q_{i,t}; \gamma_j, \theta_j)}{\partial \text{LnGDP}_{i,t}} \cdot \text{LnGDP}_{i,t} \quad (7)
\end{equation}

PSTR analysis is performed in three stages: testing the linearity, determination of the appropriate number of regime ($r$), and estimation (Fouquau et al. 2008; 287-288). Testing for linearity is achieved under the hypotheses $\gamma = 0$ or $\beta_0 = \beta_1 = 0$. However, the test statistics is not standard as there are some undefined parameters under the null hypotheses in both cases. As a result, a first-degree Taylor expansion of $\gamma = 0$ is employed instead of the transition function itself. In testing for linearity, the standard $F$-test is used and non-rejection of the null hypothesis implies that the linear model is appropriate while the rejection of the null hypothesis implies that the PSTR approach is appropriate.

Once the linear model is rejected, the next stage is to determine the appropriate number of regimes. This is done by first testing the null hypothesis of $r=r^*=1$ against the alternative hypothesis of $r=r^*+1$. The stage is finalized if the null hypothesis cannot be rejected. If the null hypothesis is rejected, then the new null hypothesis of $r=r^*+1$ is tested against the alternative hypothesis of $r=r^*+2$. This process is continued until the first non-rejection of the null hypothesis (Fouquau et al. 2008; 287-288). In the final stage of PSTR analysis, the estimation stage, fixed effects of cross sections are subtracted from the time average of the variables, and then, the transformed model is estimated using the non-linear least-squares method (NLS) (González et al. 2005).
Model, data, empirical results, and implications

Model and data specifications

In this study, the non-linear relationship between ecological balance and economic growth as well as the non-linear relationship between economic growth and each of the equilibria that determine the ecological balance in areas of cropland, fishing grounds, forest products, and grazing land are investigated in five separate models by using PSTR approach for 15 EU countries for the period of 1995–2016. The following countries are included in the study: Austria, Belgium, Denmark, France, Germany, Greece, Italy, Luxembourg, Netherlands, Portugal, Spain, Sweden, and United Kingdom. Finland and Ireland are excluded from this study since their detailed sub-component data are not available.

Using Eq. (1), the non-linear relationship between ecological balance and economic growth in model 5 as well as the non-linear relationship between economic growth and each of the equilibria that determine the ecological balance in areas of cropland, fishing grounds, forest products, and grazing land are investigated in models 1, 2, 3, and 4 respectively. All models are presented in Eq. (8);

Model 1: \[ CBS_i = \beta_0 + \beta_1 \ln GDPPC_i + \beta_2 \text{Dependency}_i + \beta_3 \text{Urbanization}_i + \beta_4 \text{Energy}_i + \epsilon_i \]
Model 2: \[ FBS_i = \beta_0 + \beta_1 \ln GDPPC_i + \beta_2 \text{Dependency}_i + \beta_3 \text{Urbanization}_i + \beta_4 \text{Energy}_i + \epsilon_i \]
Model 3: \[ FOBS_i = \beta_0 + \beta_1 \ln GDPPC_i + \beta_2 \text{Dependency}_i + \beta_3 \text{Urbanization}_i + \beta_4 \text{Energy}_i + \epsilon_i \]
Model 4: \[ GBS_i = \beta_0 + \beta_1 \ln GDPPC_i + \beta_2 \text{Dependency}_i + \beta_3 \text{Urbanization}_i + \beta_4 \text{Energy}_i + \epsilon_i \]
Model 5: \[ EBS_i = \beta_0 + \beta_1 \ln GDPPC_i + \beta_2 \text{Dependency}_i + \beta_3 \text{Urbanization}_i + \beta_4 \text{Energy}_i + \epsilon_i \]

In the models employed, each of the following variables—cropland balance sheet (CBS), fishing grounds balance sheet (FBS), forest products balance sheet (FOBS), grazing land balance sheet (GBS), and ecological balance sheet (EBS)—is employed as the dependent variable while the logarithm of real GDP per capita (\( \ln GDPPC \)) is included as the main independent variable of interest as an indicator of economic growth as well as the threshold variable for the models. To account for human activities that may affect pressure on ecological footprint and bio-capacity, as well as to minimize the problem of neglected variable bias (see, for example, Atici 2009; Jalil and Mahmud 2009; Lee et al. 2010; Shahbaz et al. 2012; Li et al. 2016; Ansari et al. 2020; Dogan and Inglesi-Lotz 2020; Erdogan 2021), this study employs energy consumption (\( \ln Energy \)), urbanization rate (\( \ln Urbanization \)), and age dependency ratio (dependency) as control variables. Detailed information on the databases where the variables of the study are obtained is given in Table 1.

Since it is assumed that all carbon uptake is considered within the biological capacity of other productive areas such as forest area, cropland, grazing land, forest land, and fishing grounds, there is no bio-capacity corresponding to the carbon footprint. Here, the inclusion of carbon bio-capacity in addition to forest land bio-capacity, in particular, may lead to double counting (Kitzes et al. 2008; Lin et al. 2016). Therefore, carbon bio-capacity individually is not included among the sub-components of the bio-capacity accounts. Furthermore, the built-up land cannot practically produce an ecological balance, as construction available is physically built on former cropland. That is, it is assumed that the EF and BC of built-up land are equal (Toderoiu 2010; Kori 2013; Iha et al. 2015; Lin et al. 2016). Therefore, the built-up land balance accounts are not included in the model. Table 2 presents the descriptive statistics of the variables included in the models. Detailed information on the databases where the variables of the study are obtained is given in Table 2.

As can be seen from Table 2, mean values of cropland balance, fishing grounds balance, forest products balance, forest products balance

| Table 1 Basic information about variables |
|------------------------------------------|
| Variables      | Explanation             | Source                      |
| CBS            | Cropland balance sheet (gha per person) | Global Footprint Network     |
| FBS            | Fishing grounds balance sheet (gha per person) | Global Footprint Network     |
| FOBS           | Forest products balance sheet (gha per person) | Global Footprint Network     |
| GBS            | Grazing land balance sheet (gha per person) | Global Footprint Network     |
| EBS            | Ecological balance sheet (gha per person) | Global Footprint Network     |
| LnGDPPC        | Real GDP per capita at chained PPPs (2011 US$) (in Log) | Penn World Table           |
| LnEnergy       | Energy use (kg of oil equivalent per capita) (in Log) | World Data Bank            |
| Urbanization   | Urban population (% of total population) | World Data Bank            |
| Dependency     | Age dependency ratio (% of working age population) | World Data Bank            |
grazing land balance, and ecological balance respectively are -0.238, 0.224, 0.417, -0.257, and -4.009. In addition, Table 3 presents the correlation matrix of the dependent and independent variables and shows that there is a statistically significant negative correlation between real GDP per capita and cropland balance, grazing land balance, and ecological balance (-0.14, -0.47, and -0.50, respectively), and statistically significant positive correlation between real GDP per capita and fishing grounds balance (0.11). On the other hand, the correlation between real GDP per capita and forest products is very small and statistically insignificant.

**Empirical results**

In this study, in which both non-linear relationships between ecological balance and economic growth, and between economic growth and ecological equilibrium in the areas of cropland, fishing grounds, forest products, and grazing land are investigated, first of all, dependency among the cross sections (countries) are analyzed. It greatly affects the estimation results whether taking into consideration the dependency among the cross sections making up the panel data or not (Breusch and Pagan 1980; Pesaran 2004). Therefore, the existence of cross-sectional dependency in series and in the model should be tested prior to further analysis. This possibility should also be considered when selecting the unit roots in order to avoid misleading estimations. Thus, this study first analyzes the cross-sectional dependency using the \( \text{LMadj} \) (adjusted Lagrange multiplier) test developed by Breusch and Pagan (1980) and further its deviation corrected by Pesaran et al. (2008). The results of the \( \text{LMadj} \) test together with other comparable tests are provided in Table 4.

As can be seen in Table 4, the null hypothesis that there is no cross-sectional dependency is strongly rejected.
based on the test results of the series related to variables used in the models. Therefore, it is concluded that there exists cross-sectional dependency in the series. This conclusion suggests that a shock in one of the countries affects others as well. As explained above, this result requires choosing appropriate test methods that take cross-sectional dependency into consideration when the methods are chosen for further stages of the analysis. As a result, in the later stages of this study, Moon and Perron’s (2004) second generation panel unit root test, which takes the cross-sectional dependency and stationarity of series into consideration is used. The results of the Moon and Perron test are provided in Table 5.

According to the results in Table 5, the null hypothesis claiming that the series has unit root is rejected for all series. This indicates that the series are stationary at level (I(0)).

Having established the stationarity of variables used in the models at level (I(0)), the next step is to proceed with the first stage of PSTR analysis, that is testing the linear model against the non-linear model. The results of the Wald test (LM), fisher tests (LMF), and LRT tests (LRT) which are used to test linearity in the models and also to determine the number of transition function are shown in Table 6.

As can be seen in Table 6, the null hypothesis is rejected at the 1% significance level in models 1, 4, and 5, and rejected at the 5% significance level in models 2 and 3 according to the LM, LMF, and LRT test results. Thus, the alternative hypothesis which suggests that there is at least one non-linear threshold effect in each model is accepted. Hence, it can be concluded that it is not appropriate to use linear models in investigating the impact of per capita real GDP on cropland, fishing grounds, forest products, and grazing land equilibriums and ecological balance. Once it is determined that the linear model is not appropriate for all models for further analysis, the next step is to determine the appropriate number of

| Table 5 | Results of Moon and Perron’s (2004) panel unit root tests |
|---|---|---|---|---|
| CBS | 3 | −14.449* | −5.978** | 0.594 |
| FBS | 1 | −6.575* | −2.444* | 0.685 |
| FOBS | 7 | −23.674* | −8.476** | 0.283 |
| GBS | 1 | −7.116* | −3.747*** | 0.809 |
| EBS | 9 | −12.695* | −8.610* | 0.666 |
| LnGDPper | 2 | −6.708* | −2.837* | 0.678 |
| LnEnergy | 1 | −35.827* | −7.626** | 0.660 |
| Urbanization | 3 | −28.170* | −6.485* | 0.272 |
| Dependency | 2 | −33.918* | −8.113* | 0.145 |

Note: \( \hat{r} \) is the estimated number of common factors, \( t^*_{a} \) and \( t^*_{b} \) are the unit root test statistics based on de-factored panel data. Corresponding \( p \)-values are in parentheses. \( \hat{\rho}_{pool}^* \) is the corrected pooled estimates of the autoregressive parameter. *, **, *** indicate significance at 1%, 5%, and 10% levels, respectively.

| Table 6 | Tests for the linearity |
|---|---|---|---|---|---|
| Threshold variables (LnGDPper) | H₀: r = 0 vs H₁: r = 1 | Model 1 | Model 2 | Model 3 | Model 4 | Model 5 |
| LM | 36.119* (0.000) | 11.636** (0.020) | 11.241** (0.024) | 29.666* (0.000) | 41.121* (0.000) |
| LMF | 3.144* (0.000) | 2.751** (0.029) | 7.873* (0.000) | 11.293* (0.000) |
| LR | 38.612* (0.000) | 11.880* (0.000) | 11.468* (0.000) | 31.320* (0.000) | 44.395* (0.000) |

Note: Under \( H₀ \), the LM and LR statistics have an asymptotic \( \chi^2 \) (mK) distribution, whereas LMF has an asymptotic \( F \) (mK, TN-n-m(K + 1)) distribution. Moreover, \( r \) is the number of transition functions. \( P \)-values are in parentheses. * and ** indicates the 1% and 5% significance level respectively.
regimes. For that purpose, LM, LMF, and LRT tests are repeated for all models and the results are shown in Table 7.

As can be seen in Table 7, the null hypothesis that the model has one threshold effect cannot be rejected for all models. Thus, it is concluded that all models have one threshold effect and the model can be estimated using the PSTR approach with two regimes. In the next step, both non-linear relationships between ecological balance and economic growth, and non-linear relationship between economic growth and the equilibrium in each of the areas of cropland, fishing grounds, forest products, and grazing land which determine the ecological balance are estimated using the PSRT method with two regimes. The results appear in Table 8.

As it can be seen in Table 8, the smoothing parameter (γ) turns out to be very small for each model (3.570, 16.186, 8.596, 3.809, and 26.594 respectively). The relatively small value for γ indicates that switching from one regime to another is not sudden but rather smooth in the relation between per capita real GDP and each equilibrium situation. This situation is demonstrated in Figure 1 in the Appendix for each model.

Also, as can be seen in Table 8, the threshold values for per capita real GDP for model 1 is found to be $32,565.22 (θ = 10.391); for model 2, it is $33,389.61 (θ = 10.416); for model 3, it is $53,103.60 (θ = 10.880); for model 4, it is $89,859.26 (θ = 11.406); and for model 5, it is $61,697.58 (θ = 11.030). The coefficient estimated for per capita real GDP (β0) in the first regime where per capita real GDP is below the threshold value is statistically significant and negative (−0.454) at the 5% significance level for model 1; it is statistically significant and negative (−0.013) at the 10% significance level for model 3; it is statistically significant and negative (−0.004) at the 5% significance level for model 4 and is statistically significant at the 1% level for model 5. The coefficient estimated for per capita real GDP (β0) is found to be not statistically significant for model 2.

The coefficient estimated for per capita real GDP (β0 + β1) in the second regime, where per capita real GDP is below the threshold value, is statistically

| Table 7 | Tests for the remaining non-linearity of the PSTR model |
|---------|-----------------------------------------------------|
| Threshold variables (LnGDPper) | H0: r = 1 vs H1: r = 2 | Model 1 | Model 2 | Model 3 | Model 4 | Model 5 |
| LM | 14.584 (0.265) | 6.858 (0.144) | 5.305 (0.257) | 5.385 (0.250) | 6.959 (0.138) |
| LMF | 1.133 (0.0.334) | 1.603 (0.174) | 1.233 (0.297) | 1.252 (0.289) | 1.627 (0.168) |
| LR | 14.969 (0.243) | 6.942 (0.139) | 5.354 (0.253) | 5.436 (0.245) | 7.045 (0.134) |

Note: Under H0, the LM and LR statistics have an asymptotic χ² (mK) distribution, whereas LMF has an asymptotic F (mK, TN-N-m(K + 1)) distribution. Moreover, r is the number of transition functions. P-values are in parentheses. * and ** indicates the 1% and 5% significance level respectively.

| Table 8 | Estimated results of the PSTR model |
|---------|-----------------------------------|
| Threshold variables (LnGDPper) | Model 1 | Model 2 | Model 3 | Model 4 | Model 5 |
| LnGDPper1 | −0.454** (0.130) | −0.055 (0.048) | −0.013*** (0.006) | −0.004** (0.001) | −1.259* (0.214) |
| LnGDPper2 | 0.507*** (0.221) | 0.019 (0.017) | 0.032 (0.029) | 0.027** (0.008) | 6.174*** (3.326) |
| Dependency1 | 0.004 (0.011) | 0.011** (0.003) | 0.020*** (0.010) | 0.015* (0.003) | 0.146* (0.018) |
| Dependency2 | −0.003 (0.020) | −0.010** (0.003) | −0.038 (0.024) | −0.058* (0.010) | −0.629* (0.156) |
| Urbanization1 | 0.038*** (0.013) | −0.006*** (0.003) | 0.002 (0.006) | −0.011* (0.002) | 0.009 (0.015) |
| Urbanization2 | −0.067*** (0.023) | 0.005** (0.001) | 0.015 (0.013) | 0.024* (0.005) | −0.461 (0.354) |
| LnEnergy1 | 0.019** (0.006) | 0.004*** (0.002) | 0.005 (0.011) | −0.001 (0.003) | −0.021 (0.023) |
| LnEnergy2 | −0.035* (0.009) | 0.0002 (0.004) | −0.019 (0.024) | −0.009 (0.006) | 0.004 (0.046) |
| Location parameters, θ | 10.391 | 10.416 | 10.880 | 11.406 | 11.030 |
| Slope parameters, γ | 3.570 | 16.186 | 8.596 | 3.809 | 26.594 |

Note: Standard errors are corrected for heteroskedasticity in parentheses. *, **, *** indicate significance at 1%, 5%, and 10% levels, respectively.
significant and positive (0.053) at the 10% significance level for model 1, it is statistically significant and positive (0.023) at the 5% level for model 4 and statistically significant and positive (4.915) at the 10% level for model 5. The coefficient estimated for per capita real GDP ($\beta_0 + \beta_1$) is found to be not statistically significant for models 2 and 3.

Further interpretation of the PSTR model is as follows: In the first model, where the relationship between cropland balance and economic growth is estimated, it is found that an increase in economic growth has a negative impact on cropland balance when the per capita real GDP is below $32,565.22 while an increase in economic growth has a positive impact on cropland balance when the per capita real GDP is above $32,565.22. In model 3 where the relationship between the forest products balance and economic growth is investigated, it is found that an increase in economic growth has a negative impact on forest products balance when the per capita real GDP is below $53,103.60 while there is no statistically significant relation detected between economic growth and forest products balance when the per capita real GDP is above $53,103.60. In model 4 where the relationship between the grazing land balance and economic growth is estimated, it is found that an increase in economic growth has a negative impact on grazing land balance when the per capita real GDP is above $89,859.26 while an increase in economic growth has a positive impact on grazing land balance when the per capita real GDP is below $89,859.26. In model 5 where the relationship between total ecological balance and economic growth is estimated, it is found that an increase in economic growth has a negative impact on ecological balance when the per capita real GDP is below $33,389.61 while an increase in economic growth has a positive impact on ecological balance when the per capita real GDP is above $33,389.61. The age dependency ratio is found to be not significant on cropland balance.

In model 2, in which the relationship between fishing grounds balance and economic growth is estimated, it is found that an increase in age dependency ratio and energy consumption has a positive impact on fishing grounds balance (coefficients are 0.011 and 0.004 respectively) while an increase in urbanization ratio has a negative impact on fishing grounds balance ($-0.006$) when the per capita real GDP is below $33,389.61 while it has a negative impact on fishing grounds balance (coefficients are $-0.029$ and $-0.016$ respectively) when the per capita real GDP is above $33,389.61$. The age dependency ratio is found to be not significant on cropland balance. In model 2, the estimated coefficients for control variables that may possibly affect the equilibrium situation in each model such as age dependency ratio, urbanization ratio, and energy consumption are given in Table 8. In model 1 where the relationship between the cropland balance and economic growth is estimated, it is found that an increase in the urbanization ratio and energy consumption has a positive impact on cropland balance (coefficients are 0.038 and 0.019, respectively) when the per capita real GDP is below $32,565.22 while it has a negative impact on cropland balance (coefficients are $-0.029$ and $-0.016$ respectively) when the per capita real GDP is above $32,565.22$. The age dependency ratio is found to be not significant on cropland balance.

In model 3 where the relationship between forest products balance and economic growth is estimated, it is found that an increase in age dependency ratio has a positive impact on forest products balance (coefficients are 0.038 and 0.019, respectively) while there is no statistically significant relationship detected between energy consumption and forest products balance or between urbanization ratio and forest products balance when the per capita real GDP is above $32,565.22$. The age dependency ratio is found to be not significant on forest products balance.

In model 4 where the relationship between grazing land balance and economic growth is estimated, it is found that an increase in the urbanization ratio has a negative impact on grazing land balance when the per capita real GDP is above $33,389.61$ while an increase in energy consumption has a positive impact on grazing land balance ($0.011$ and $0.004$ respectively) while an increase in urbanization ratio has a negative impact on grazing land balance ($-0.006$) when the per capita real GDP is below $33,389.61$. The age dependency ratio is found to be not significant on grazing land balance.

In model 5 where the relationship between total ecological balance and economic growth is estimated, it is found that an increase in age dependency ratio has a positive impact on total ecological balance when the per capita real GDP is above $33,389.61$ while an increase in urbanization ratio has a negative impact on total ecological balance ($0.001$) while an increase in energy consumption has a positive impact on total ecological balance ($0.016$) while an increase in urbanization ratio has a negative impact on total ecological balance ($-0.001$). Energy consumption is found to be not significant on total ecological balance when the per capita real GDP is above $33,389.61$.

In model 2, in which the relationship between fishing grounds balance and economic growth is estimated, it is found that an increase in age dependency ratio has a positive impact on fishing grounds balance (0.029) while there is no statistically significant relationship detected between energy consumption and fishing grounds balance when the per capita real GDP is above $33,389.61$ while an increase in urbanization ratio has a negative impact on fishing grounds balance ($-0.001$) while an increase in energy consumption has a positive impact on fishing grounds balance ($0.016$) while an increase in urbanization ratio has a negative impact on fishing grounds balance ($-0.001$). Energy consumption is found to be not significant on fishing grounds balance when the per capita real GDP is above $33,389.61$.
found that an increase in age dependency ratio has a positive impact on grazing land balance (0.015) while the increase in urbanization ratio has a negative impact on grazing land balance (−0.011) when the per capita real GDP is below $89,859.26. On the other hand, when the per capita real GDP is above $89,859.26, an increase in age dependency ratio has a negative impact on grazing land balance (0.013). Energy consumption is found to be not significant on grazing land balance in both regimes.

In model 5 where the relationship between ecological balance and economic growth is estimated, it is found that an increase in age dependency ratio has a positive impact on ecological balance (0.146) when the per capita real GDP is below $61,697.58. On the other hand, when the per capita real GDP is above $61,697.58, an increase in age dependency ratio has a negative impact on ecological balance (−0.629) energy consumption and urbanization rate are found to be not significant on ecological balance in both regimes.

**Conclusion**

This paper provides new empirical evidence on the environmental effects of economic activities in terms of ecological balance calculations, taking into account the main criticisms of the EKC hypothesis and policy interpretations in the literature. Using the dataset of 13 EU member economies for the period 1995–2016, the paper empirically examines effects of per capita income on the ecological balance sheet—namely, total ecological balance, cropland balance, grazing land balance, forest area balance, and fishing grounds balance—and analyzes whether the relationship can exhibit the inverted U-curve of the EKC path. The sample consists of a data set of 13 EU countries for the period 1995–2016. To examine the impact of real GDP per capita on ecological balance sheets, this paper uses the panel STR model, which can endogenously estimate the threshold level while also allowing parameters to change smoothly from one regime to another.

The empirical findings confirm that there is a nonlinear link between per capita income and total ecological balance and its main components: cropland balance, grazing land balance, forest area balance, and fishing grounds balance. However, this study found no evidence to support the existence of the inverted U-shape EKC pattern for both the total ecological balance sheet and its sub-components.

The results from the PSTR model indicate that as the amount of goods and services produced per capita increases, the process initially improves the total ecological balance, cropland balance, and grazing land balance and after above per capita income levels of approximately $61,697, $32,565, and $89,859, respectively, environmental degradation occurs smoothly. These findings indicate that above a certain threshold level, nature is strongly polluted by human activities and cannot absorb this pollution at this rate, which exceeds its regenerative capacity. In other words, a higher production of goods and services results in a higher ecological erosion for grazing land, cropland, and total land. Moreover, these results are in line with Caviglia-Harris et al. (2009) for 146 countries, Liu et al. (2018) for China, Aydin et al. (2019a) for 26 EU countries, Esen et al. (2020) for 9 European countries, and Dogan et al. (2020) for BRICST.

Secondly, the findings obtained for forest area balance and fishing grounds balance differ. According to these results, there is a sustainable structure for the forest area up to the per capita income threshold of $53,103 but this statistical relationship deteriorates once this threshold level is surpassed, that is, the results become statistically insignificant. In addition, the effects of per capita income on fishing grounds balance are statistically insignificant both below and above the threshold of $33,389. The insignificance of the relationship between per capita real GDP and fishing grounds balance may be due to the expansion of aquaculture alongside traditional fishing activities in the last few decades. It is known that the importance of aquaculture increases in meeting the protein needs of the rapidly increasing population worldwide, as natural stocks gradually decrease. Based on FAO’s (2020) data, as of 2018, aquaculture supplies about 46% (82.1 million tons) of total production and about 52% of fish for human consumption. Despite these findings, it is not clear that aquaculture is beneficial on balance. It is argued that aquaculture, which has made critical contributions to the protection of biodiversity and food security as well as limiting overfishing, has a number of detrimental effects on the environment. Therefore, changing a negative activity such as overfishing with a solution like an aquaculture that could potentially cause a different range of problems might not be the right step. The transition from traditional fishing to aquaculture may have an unexpected impact on the relationship between per capita real GDP and fishing grounds variables.

Finally, the results of this study suggest that, for countries with per capita real GDP levels higher than the critical threshold, governments should consider limiting the ecologically damaging economic activities or imposing regulations to modify how they are conducted in order to reduce the ecological damage.
Appendix

Fig. 1 Estimated transition function of the PSTR model against real GDP per capita for all models
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Data availability  The data sets supporting the results of this article are included within the article and its additional files.

Declarations

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