9 Addressing Agricultural Pressures on Water Resources

A DEA Environmental Assessment in the Case of European Transboundary Basins

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9.1 INTRODUCTION

Agriculture is not only the main user of water resources (around 70 percent of all water abstraction worldwide) (FAO 2017) but is also the most intractable source of riverine and marine pollution (Boyle 2014; Mohaupt et al. 2007). Most European river basins (RBs) face significant pressures from agriculture, both in terms of quantity (e.g., abstraction stress) and quality (e.g., agrochemical diffuse pollution) (EEA 2015; EC 2012; Mohaupt et al. 2007; Özerol et al. 2012). The European Union (EU) water legislation aims to achieve an ecologically and chemically good status of water bodies through the reduction of demand pressures on water resources, a progressive reduction of pollution, and the preservation of ecosystems (van Rijswick et al. 2010). The European Environment Agency (EEA) (EEA 2012a, 2015) indicates that the environmental status of a large number of water bodies across Europe is unacceptable and is of greater concern in large transboundary river basins (TRBs). Furthermore, the EEA asserts that emissions of nutrient (e.g., phosphorus and nitrogen) compounds from agriculture represent a serious pollutant threat to European water bodies (EEA

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2012b; EU 2017). The EU Water Framework Directive (WFD) (EC/2000/60) explicitly includes transboundary rivers, for whose management international or interregional structures are envisaged (Art. 3, Preamble 35). However, this directive fails to specify instruments for coordination and responsibilities for riparian states so that the established environmental goals may be more easily achieved (van Rijswick et al. 2010; Wiering et al. 2010).

Transboundary rivers represent complex interdependent structures at a multidimensional level (e.g., socioeconomic, political, environmental) for riparian countries, whereby the river is a source of externalities (e.g., agrochemical pollution) and mutual vulnerabilities (Dimitriou et al. 2012; Dinar et al. 2013; Kauffman 2015). In this respect, the subtractability and nonexclusion characteristics of transboundary rivers have led them to become common pool resources shared by riparian countries where the common dilemma of overuse and mismanagement can easily occur (Ostrom 2005). Much of the recent research on the management of transboundary rivers has focused on the determinants of political and geographical conditions for effective cooperative management (e.g., Dinar et al. 2013; Zawahri and Mitchell 2011), legal aspects and conflict resolution (e.g., Jager 2016; Petersen-Perlman et al. 2017), and risks associated to climate change, such as droughts and floods (e.g., Bakker and Duncan 2017; Pulwarty and Maia 2015). Although environmental degradation is considered an incentive for international joint efforts (Dinar et al. 2013) and agricultural pressures on water resources represent a significant challenge for RB management (Kallioras et al. 2006), their analysis has attracted much less attention in academic and international cooperation spheres (Munia et al. 2016; UNECE 2011).

The present work aims to fill this gap and offers a comparative analysis for the identification of factors that may influence performance patterns in managing agricultural pressures. With this aim, the data envelopment analysis (DEA) methodology used in this study enables the assessment of the capacity of the analyzed RBs regarding the minimization of agricultural pressures on water resources.

The rest of the paper is organized as follows. Section 9.2 briefly describes our case study comprised of 20 European RBs. The following section (Section 9.3) presents the DEA methodology used and describes the data. Section 9.4 summarizes the results of our analysis. Finally, certain concluding remarks are offered in Section 9.5.

9.2 CASE STUDY

TRBs comprise about 47 percent of the world's continental land area and Europe has the largest number of these transnational rivers, that is, 68 out of the total 286 in the world (Giordano et al. 2014). The watershed or RB scale has often been considered a useful organizational unit for the evaluation of policy and research issues, especially regarding environmental concerns (EC 2012). Further, the WFD recognizes the RB as the main natural unit for the protection of the status of water bodies, and as the appropriate scale for integrated water resource management (IWRM) (Berbel and Expósito 2018). In this respect, the DEA method applied herein, which is explained in greater detail in the subsequent section, uses RBs as the decision units to be assessed. Specifically, the method applied in this study allows us to assess the management

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performance of these decision units (or RBs in our case study) at reducing agricultural pressures, such as nutrient pollution and abstraction stress, on water resources.

Over recent decades, cooperation efforts, involving all TRBs to solve environmental problems derived from human activities (including agriculture), have increased dramatically, as globally initiated by the Global Environment Facility (GEF) set up by the World Bank in 1991 (Gerlak 2004), and followed by the Convention on the Protection and Use of Transboundary Watercourses and International Lakes (UNECE, Helsinki, 1992; Libert 2015). To this end, the Convention established the principle of the joint management bodies, based on any multilateral institutional arrangement for cooperation between the riparian countries. Since then, most European TRBs have created international commissions to jointly manage water resources on a basin scale and to reduce negative externalities from human activities, such as agricultural pollution and abstraction stress. One of the first multilateral cooperation institutions was established in 1988 through the International Commission for the Protection of the Danube river (ICPDR), whose main objective is the implementation of the Danube River Protection Convention. Similarly, international commissions have been established for many other rivers in Europe, such as the Conventions of the International Commission for the Protection of the Elbe (1991), the Oder (1999), the Rhine (1999), and the Albufeira Convention for the Iberian rivers (1998). As further discussed in the "Discussion" section of this study, these transboundary commissions have been widely criticized for failing to have achieved effective joint management aimed at reducing pressures on water resources (including those of the agricultural sector), and hence for failing to have served as an effective instrument to improve the ecological and chemical status of TRBs (Bernauer and Kuhn 2010).

In this context, this study takes a representative sample of 20 European RBs to assess their management capability in terms of reducing agricultural pressures on water resources. It is worth noting that the area of some RBs, such as Ebro and Po, are mainly located in one country. This study aims to assess the managerial efficiency at RB scale, thus taking the RB as a management decision-making unit, regardless of the number of riparian countries or the existence of TRB agency. Nevertheless, the potential effect of these factors, among others, on the estimated efficiency patterns are also analyzed in this study.

Table 9.1 shows various characteristics regarding drainage area, population, per capita gross domestic product (GDP), renewable water sources (RWS), total water withdrawals, and the relative weight of agriculture on those total withdrawals. These selected indicators show the high heterogeneity in our sample of RBs. A further description of the riparian countries of each TRB is given in Table 9.2.

9.3 METHODOLOGY AND DATA

9.3.1 Method

The DEA method was initially proposed by Farrell (1957). Subsequently, it has been extended under various functional schemes, such as an input-oriented scheme with constant returns to scale (Charnes et al. 1978), output-oriented maximization (Charnes et al. 1981), variable returns to scale (Banker et al. 1984), and both radial

TABLE 9.1
Indicators of Selected European TRBs (2010)

River basin	Drainage area (10^3 km²)	Population (10^3)	GDP pc (USD)	RWS (km³/year)	Water withdrawals (km³/year)	Agric. withdrawals (% total)
Danube	796	80,185	18,478	221,762	53,822	36.0
Dnieper	511	29,457	5,889	66,635	14,486	35.5
Don	439	18,819	11,359	45,375	10,205	29.4
Douro/Duero	97	3,492	25,521	24,098	7,412	79.5
Ebro	85	2,805	28,024	19,082	9,865	72.3
Elbe	139	21,860	37,940	28,957	7,462	9.8
Guadiana	67	1,475	28,017	11,076	8,605	94.0
Kemi	54	105	44,504	17,900	30	2.7
Klarälven	50	901	60,573	20,564	602	12.7
Maritsa	53	3,476	8,965	11,970	6,404	52.6
Neman	93	4,789	12,144	20,754	880	12.1
Oder/Odra	119	15,718	15,163	20,997	4,720	4.1
Po	72	15,918	35,500	48,958	18,575	40.5
Rhine	164	48,831	49,543	74,972	28,834	4.5
Rhone	97	10,055	46,047	52,339	8,207	25.1
Seine	73	15,775	41,422	20,712	8,353	20.6
Tagus/Tejo	71	7,244	28,303	19,297	7,968	59.9
Vistula/Wista	192	23,148	12,753	34,604	7,699	3.4
Volga	1,412	58,621	14,612	274,165	25,004	11.4
Vuoksi	287	3,246	23,001	87,344	5,590	1.3

Source: Data from TWAP (2019).

and nonradial approaches (Sueyoshi and Sekitani 2009), among others. In the water management sector, DEA methods have been extensively used to assess efficiency among a group of management or decision-making units (DMUs) (e.g., water utilities) (Xiang et al. 2016; Romano et al. 2017, among others). Zhu (2016) offers a recent review of DEA literature and applications in environmental issues.

DEA methods are based on a nonparametric approach to estimate efficient frontiers in the sense that no assumption regarding the functional form is required. This enables relative efficiency estimates to be obtained for a group of DMUs (i.e., our sample of European TRBs) by using multiple inputs and outputs and alternative output–input specifications. Furthermore, DEA methods measure the efficiency of a DMU with the simple restriction that all sampled DMUs lie on or below the efficient frontier and obviate the need to assign prespecified weights to either inputs or outputs. Each DMU not on the frontier (thus, an inefficient DMU) is scaled against a convex combination of the DMUs on the frontier faced closest to it. Thus, efficiency mappings of a group of DMUs can be obtained and efficient (or benchmark) DMUs can be identified. Additionally, and conversely to other methodologies, such as qualitative analysis and multicriteria schemes, DEA methods are capable of identifying

TABLE 9.2
Riparian Countries

TRB	Riparian countries
Danube	Albania, Austria, Bosnia And Herzegovina, Bulgaria, Croatia, Czech Republic, Germany, Hungary, Italy, (The former Yugoslav Republic of) Macedonia, (Republic of) Moldova, Montenegro, Poland, Romania, Serbia, Slovakia, Slovenia, Switzerland, Ukraine
Dnieper	Belarus, Russian Federation, Ukraine
Don	Russian Federation, Ukraine
Douro/Duero	Portugal, Spain
Ebro	Andorra, France, Spain
Elbe	Austria, Czech Republic, Germany, Poland
Guadiana	Portugal, Spain
Kemi	Finland, Norway, Russian Federation
Klarälven	Norway, Sweden
Maritsa	Bulgaria, Greece, Turkey
Neman	Belarus, Latvia, Lithuania, Poland, Russian Federation
Oder/Odra	Czech Republic, Germany, Poland, Slovakia
Po	France, Italy, Switzerland
Rhine	Austria, Belgium, France, Germany, Italy, Liechtenstein, Luxembourg, Netherlands, Switzerland
Rhone	France, Italy, Switzerland
Seine	Belgium, France
Tagus/Tejo	Portugal, Spain
Vistula/Wista	Belarus, Czech Republic, Poland, Slovakia, Ukraine
Volga	Kazakhstan, Russian Federation
Vuoksi	Belarus, Finland, Russian Federation

input–output relationships that remain unobserved for other methods. DEA methods therefore constitute a powerful instrument in obtaining additional strategic information not provided by alternative methodologies.

DEA methods allow for two types of model orientations: input or output (Avkiran and Rowlands 2008). Since our objective is to estimate the relative efficiency of each DMU at generating selected outputs (i.e., reduction of agricultural pressures), we believe the output specification to be the most appropriate. The specific approach used in this study also accounts for the role of undesirable outputs, such as agricultural pressures (quantitative and qualitative) on water resources, in order to assess efficiency among a group of DMUs (e.g., TRBs). Recent DEA applications have shown the relevance of including undesirable outputs (e.g., pollution) as a more realistic specification of the optimization model (Sueyoshi and Goto 2011). Furthermore, recent DEA developments have revealed the importance of using different output specifications (i.e., natural and managerial) in order to obtain additional strategic information for the assessment of DMUs (Expósito and Velasco 2018; Sueyoshi and Goto 2011). With the aim to assess the managerial capacity of RBs at minimizing agricultural pressures, this study uses a managerial output specification.

The managerial specification implies a high methodological sophistication in the sense that a DMU may decrease the directional vector of undesirable outputs by increasing inputs. This type of specification is referred to as "managerial disposability" and is usually related to the implementation of innovative management initiatives (e.g., effective multilateral arrangements to reduce agricultural pressures on water resources at TRB scale). In our specific case, it would reflect the TRB capacity to minimize agricultural pressures on water resources despite a potential increase of agricultural water withdrawals. The optimization model produces an autonomous indicator of relative efficiency referred to herein as "managerial efficiency" scheme. Subsequently, a simplified mathematical description of the applied DEA model is offered. Nevertheless, a detailed description of the DEA model applied can be found in Expósito and Velasco (2018, 2020).

In our DEA model, each j-th DMU $j = 1,\ldots n$, uses inputs $X_j = \left(x_{1j},\ldots,x_{mj}\right)^{\mathrm{T}}$ and generates desirable outputs, represented by $G_j = \left(g_{1j},\ldots,g_{sj}\right)^{\mathrm{T}}$, and undesirable outputs, represented by $B_j = \left(b_{1j},\ldots,b_{hj}\right)^{\mathrm{T}}$. Furthermore, $\mathrm{d}_i^x, i = 1,\ldots m$, $\mathrm{d}_r^g, r = 1,\ldots,s$, and $\mathrm{d}_f^b, f = 1,\ldots,h$ represent slack variables related to inputs, and desirable and undesirable outputs, respectively. $\lambda = \left(\lambda_1,\ldots,\lambda_n\right)^{\mathrm{T}}$ are unknown structural or intensity variables, which are used for connecting the input and output vectors via a convex combination. R is the range resolute throughout the upper and lower bounds of inputs, desirable outputs, and undesirable outputs and is expressed by following expressions:

$$R_{\mathrm{i}}^{x} = (m+s+h)^{-1}\left(\max\left\{x_{\mathrm{ij}} \mid j = 1,\ldots,n\right\} - \min\left\{x_{\mathrm{ij}} \mid j = 1,\ldots,n\right\}\right)^{-1}$$

$$R_{\mathrm{r}}^{g} = (m+s+h)^{-1}\left(\max\left\{g_{\mathrm{rj}} \mid j = 1,\ldots,n\right\} - \min\left\{g_{\mathrm{rj}} \mid j = 1,\ldots,n\right\}\right)^{-1}$$

$$R_{\mathrm{f}}^{b} = (m+s+h)^{-1}\left(\max\left\{b_{\mathrm{fj}} \mid j = 1,\ldots,n\right\} - \min\left\{b_{\mathrm{fj}} \mid j = 1,\ldots,n\right\}\right)^{-1}$$

The managerial efficiency of the k_{th} DMU is evaluated by the following radial model:

$$\mathrm{Max}\,\xi + \varepsilon\left[\sum_{i=1}^{m} R_i^x\, d_i^x + \sum_{r=1}^{s} R_r^g\, d_r^g + \sum_{f=1}^{h} R_f^b\, d_f^b\right]$$

$$s.t.\sum_{j=1}^{n} x_{ij}\,\lambda_j + (-1)^o\, d_i^x = x_{ik} \quad \left(i = 1,\ldots,m\right),$$

$$\sum_{j=1}^{n} g_{rj}\,\lambda_j - d_r^g - \xi g_{rk} = g_{rk} \quad (r = 1,\ldots,s),$$

$$\sum_{j=1}^{n} b_{fj}\,\lambda_j + d_f^b + \xi b_{fk} = b_{fk} \quad (f = 1,\ldots,h),$$

$$\sum_{j=1}^{n} \lambda_j = 1,$$

$$\lambda_j \geq 0 (j = 1,\ldots,n),\, d_i^x \geq 0 (i = 1,\ldots,m),$$

$$d_r^g \geq 0 (r = 1,\ldots,s), d_f^b \ \geq\ 0 (f = 1,\ \ldots,\ h) \text{ and,}$$

$$\xi\text{: unrestricted} \tag{9.1}$$

Its solution provides the necessary efficiency scores, measured by:

$$\theta^* = 1 - \left[\xi^* + \varepsilon\left(\sum_{i=1}^{m} R_i^x\ \mathrm{d}_\mathrm{i}^{x*} + \sum_{r=1}^{s} R_r^g\ d_r^{g*} + \sum_{f=1}^{h} R_f^b\ d_f^{b*}\right)\right], \tag{9.2}$$

being $o = 1$.

Once the efficiency mapping of DMUs is estimated, this study tests the impact of certain determinants on the estimated efficiency scores for our sample of European TRBs. The most commonly included determinants (or explanatory factors) in the literature regarding the comparative analysis of the environmental performance of TRBs (see, for example, Bernauer and Kuhn (2010) and Knieper and Pahl-Wostl (2016), among others) are those related with population (e.g., population density), economic development (e.g., per capita income or GDP), overall human pressure on water resources (e.g., water withdrawals on renewable water resources on a RB scale), and geo-location variables. In order to test whether these determinants carry a sufficiently large influence to explain efficiency scores in our case study, we have applied both an ordinary least squares (OLS) model, which controls for heteroscedasticity with robust standard errors, and a Tobit model, which controls for upper limit values (i.e., 1 = full efficiency). The use of two alternative regression models contribute to the robustness of the obtained results.

9.3.2 Data

In 2012, the GEF approved the Transboundary Water Assessment Programme (TWAP) of the United Nations Environment Programme (UNEP) following an earlier programme (Development of the Methodology and Arrangements for the GEF Transboundary Waters Assessment Programme in 2011). TWAP offers a unique homogeneous dataset for TRBs that enables international comparison and assessment studies (UNEP 2016). TWAP constitutes a global assessment platform of 286 TRBs distributed worldwide and offers a wide variety of indicators regarding water quantity, water quality, ecosystems, governance, and socioeconomic fields. This study uses the latest available data (year 2010) of the TWAP dataset (TWAP, 2019). In order to apply our DEA assessment to the 20 European TRBs selected, the following indicators have been employed:

1. Agricultural withdrawals (input indicator): This indicator has been selected as the necessary input for agricultural activities (i.e., irrigation and livestock production). This input is withdrawn from the RB system (both surface and groundwater sources) for production objectives and given back (partially) in the form of returns which usually contain pollutants, such as nitrogen and phosphorus compounds. Values are measured in km^3/year (Table 9.3).
2. Nutrient pollution (qualitative output indicator): This indicator considers river pollution from nitrogen and phosphorus compounds, which are mainly caused by agricultural activities (Bernauer and Kuhn 2010; Dimitriou et al. 2012; EEA 2012b). Urban wastewater and atmospheric deposition of nitrogen can also generate nutrient pollution, but to a much lesser extent (Boyle 2014). The construction of the indicator is based on the methodology developed by Mayorga et al. (2010) and Seitzinger et al. (2010). In our specific DEA optimization model, this indicator needs to be minimized. Values range from 0 to 1 (from less to more highly polluted) (Table 9.3).
3. Agricultural abstraction stress (quantitative output indicator): This indicator identifies agricultural water stress, and is constructed based on the consumption-to-availability ratio (mean annual water consumption divided by the sum of mean annual runoff on a river basin scale). A reduction of the agricultural stress indicator implies a decrease in the pressure of agriculture on water resources of the RB (e.g., through more efficient practices of water use). Its introduction as an additional output indicator is justified since the presence of agricultural nutrient pollution in RBs is strongly related to quantity factors, as measured by consumption-to-availability ratios (EEA 2012b;

TABLE 9.3
Descriptive Statistics

	Agricultural withdrawals	Nutrient pollution	Abstraction stress
Average	3,688	0.69	0.06
St. Deviation	4,589	0.21	0.07
Maximum	19,397	1	0.30
Minimum	0.83	0.25	0.00

Munia et al. 2016). In our specific case, this output has been introduced in terms of its inverse value, therefore this indicator needs to be maximized in our optimization problem. Values range from 0 to 1 (Table 9.3).

In summary, our optimization problem aims to minimize not only the agricultural nutrient pollution (qualitative output indicator) but also the agricultural abstraction stress (quantitative output indicator). Therefore, the DEA method applied enables both output indicators to be simultaneously optimized. Descriptive statistics of our input and output variables are shown in Table 9.3. It is worth noting that the contrast of output/input ratios of each RB with the mean values shows that any potential problems of sensitivity caused by the existence of outliers and statistical noise are ruled out. DEA methods do not require data to be normalized (e.g., by agricultural activity or withdrawals in the basin) to obtain efficiency scores. Additionally, the condition regarding the minimum number of observations per variable established by Banker et al. (1996) is met (20 DMUs and 3 input–output variables). Hence no misspecification problems are observed.

9.4 RESULTS

As discussed in the previous section, relative efficiency relies on effective management initiatives to minimize undesirable outputs. In our managerial output specification, the imposition of the reduction of undesirable outputs can be achieved despite an input increase (i.e., higher water withdrawals). Under this assumption, the obtained efficiency mapping describes the capacity to reduce agricultural pressures on water resources (despite potential higher withdrawals of water for agriculture) and helps to identify those benchmark RBs from which to learn.

The results of our efficiency assessment for our sample of European RBs are presented in a step-by-step manner. The first step focuses on the efficiency mapping obtained from our model specification (Table 9.4). The results identify four efficient (benchmark) RBs: the Danube, Volga, Kemi, and Vuoksi RBs. In these RBs, an increase in agricultural withdrawals would not necessarily imply greater pressures on water resources from agriculture. This can be explained by the implementation of effective management initiatives that positively impact on the input–output intrinsic relationship. Conversely, inefficient RBs would therefore need to increase their management efforts toward reducing agricultural pressures, since the increase of agricultural withdrawals would be counterproductive. Within the heterogeneous range of inefficient RBs, the Elbe and Rhine RBs register the lowest efficiency scores (0.25), thus showing the urgent need to implement agricultural management initiatives at basin scale. Interestingly, the Kemi and Vuoksi RBs, two efficient DMUs, present a very low proportion of agricultural water withdrawals on total withdrawals on a basin scale, which could be argued as a possible explanatory factor for its high efficiency scores (since lower agricultural water withdrawals would lead to less agricultural nutrient pollution and less abstraction stress). Nevertheless, the Volga and Danube RBs register greater agricultural water withdrawals (Table 9.1) and are also fully efficient DMUs in our managerial specification. Therefore, as widely argued in the existing literature (Zhu 2016), DEA outcomes do not depend on input data (or on its

TABLE 9.4
Mapping of Efficiency Scores

	Efficiency score
Danube	**1**
Dnieper	0.64
Don	0.34
Douro	0.45
Ebro	0.50
Elbe	0.25
Guadiana	0.55
Kemi	**1**
Klarälven	0.57
Maritsa	0.35
Neman	0.42
Oder	0.33
Po	0.52
Rhine	0.25
Rhone	0.33
Seine	0.52
Tagus	0.25
Vistula	0.33
Volga	**1**
Vuoksi	**1**
Average	*0.54*
St. Dev.	*0.26*
EU[2]	*0.38*
Non-EU[3]	*0.82*

Note: [1]Weighted by drainage area. [2]EU group with more than 50 percent of the TRB drainage area within EU countries. [3]Conversely, more than 50 percent of the TRB area within non-EU countries.

heterogeneity among DMUs), but instead on the intrinsic input–output relationships optimized for the selected group of DMUs. Furthermore, the less efficient DMUs, the Elbe and Rhine RBs, show a low proportion of agricultural withdrawals of the total, thus showing that a low input value does not necessary lead to a high efficiency score. Additionally, it is worth noting that average values show a relatively low level of efficiency, 0.54 (Table 9.4), and there is high heterogeneity in our sample (as measured by the estimated standard deviations).

In a final step, the analysis focuses on the determinants of our estimated efficiency scores. To this end, an OLS model is estimated, which controls for heteroscedasticity with robust standard errors (Table 9.5), together with a Tobit model, which controls for upper limit values (i.e., 1 = full efficiency) (Table 9.6). The results show whether certain factors (as described in Section 3.1) determine the efficiency scores achieved by our group of RBs. Both estimation techniques show similar estimated parameters in terms of the signs of the relationship and significance levels, thus

TABLE 9.5
Analysis of Determinants of Efficiency (OLS Robust)

Variable	Coeff.	p-value
GDP per capita[1]	0.108	0.094
Non-EU[2]	0.029	0.816
No. of countries	-0.024**	0.014
Population density[1]	-0.151***	0.002
RB Area[1]	0.147**	0.026
TRBMIs [2]	-0.027	0.121
Withdrawals (over total renewable resources)[1]	0.001	0.988
R^2	*0.721*	

Note: [1]Variables in logs. [2]Binary variables. *** Denotes significance at 1 percent. ** Denotes significance at 5 p.

TABLE 9.6
Analysis of Efficiency's Determinants (Tobit Model)

Variable	Coeff.	*p*-value
GDP per capita[1]	0.134	0.141
Non-EU[2]	0.076	0.687
No. of countries	–0.034*	0.084
Population density[1]	–0.176***	0.005
RB Area[1]	0.174***	0.009
TRBMIs [2]	–0.032	0.278
Withdrawals (over total renewable resources)[1]	–0.009	0.835
LR Chi^2	*26.11*	

Note: [1]Variables in logs. [2]Binary variables. *** Denotes significance at 1 percent. * Denotes significance at 10 percent.

showing sufficient robustness of our estimates. No multicollinearity problems have been detected.

Estimates show that variables, such GDP per capita (measured in US dollars), non-EU versus EU (binary variable that takes value 1 when more than 50 percent of the RB area is located in non-EU countries), the existence of transboundary river basin management institutions (TRBMIs) (binary variable that takes value 1 if a management transboundary organization or commission exists) and total water withdrawals are revealed to be non-significant and therefore fail to explain efficiency scores. On the other hand, as expected, factors, such as an increasing number of countries, population density (inhabitants per square kilometer) and RB area, exert a negative effect on efficiency scores.

Based on the obtained results, it seems that EU RBs (understood as those mostly located in EU territory) fail to perform better as a result of the significant efforts derived from intensive water and agricultural regulation, compared to RBs mainly located in non-EU countries (Table 9.4). In this respect, and following Bernauer and Kuhn (2010), it is argued that the absence of any effect of EU membership on better management of agricultural pressures may indicate that agriculture and water policies are not sufficiently coordinated, and that the EU Common Agricultural Policy (CAP) might not being providing the results expected in terms of reducing intensive farming based on high use of water and fertilizers. This might also be understood as EU RBs had more intensive agriculture than non-EU RBs. Another interesting result shows that the existence of TRBMIs fails to exert a positive effect on the estimated efficiency scores, and therefore the mere creation of transboundary multilateral institutions does not necessarily lead to a better performance in terms of reducing agricultural pressures (both quantitative and qualitative) on water resources at RB scale. Again, the lack of effective cooperation and of strategic policy coordination could lie behind these results. Finally, the nonexistence of an income effect (as measured by GDP per capita) on efficiency mappings would show that agricultural pressures are less sensitive to capital-intensive factors (since richer countries can more easily implement measures such as treatment plants, irrigation modernization programs, and agricultural compensation schemes) and therefore require greater political and cooperation efforts (Bernauer and Kuhn 2010; Expósito et al. 2019). These findings are also in line with those of Dinar (2008) and of Knieper and Pahl-Wostl (2016). Additionally, a high number of riparian countries seem to interfere negatively on the RB performance at reducing undesirable pressures of agricultural activity on water resources, thereby achieving lower efficiency scores. In this respect, Dinar (2008) argues that the larger the number of states involved in transboundary river cooperation becomes, the more difficult it is to achieve effective cooperation for agricultural pollution abatement. Similarly, higher population densities constitute increasing pressures on the RB that would explain the observed negative effect on efficiency scores. Again, it is worth noting the case of the Danube RB, since it registers a high number of riparian countries and population density, as well as relatively high agricultural water withdrawals. Despite these determinants, the Danube RB is flagged as a benchmark unit due to its exemplary management initiatives. Finally, the RB area seems to play a positive role in explaining efficiency scores, thereby indicating that RBs of a more extensive nature would achieve better performance results. This result could be related to the consumption-to-availability ratio on a RB scale, since greater (in area) RBs would register greater renewable resources and would therefore register lower pressure of water withdrawals.

The findings offer highly relevant information to policy and decision makers, since they clearly suggest that current managerial initiatives are insufficient to achieve a sustainable agriculture, as well as to reduce the negative agricultural pressures on water resources. Therefore, a more effective transboundary cooperation is required and innovative initiatives focused on reducing agricultural pollution and efficient agricultural water use (thus reducing abstraction pressure) need to be implemented. Managerial initiatives implemented by the benchmark basins identified in this study, as well as the identification of the factors affecting the managerial efficiency to reduce agricultural pressures, might offer valuable information for decision makers.

9.5 CONCLUSION

Transboundary rivers are under constant strain to meet the demands from agriculture and other uses. In this context, the joint efforts of all stakeholders implied in RB management, as required by the WFD and the Aarhus Convention, remain necessary for the development and support of the implementation of effective management measures to address agricultural pressures on water resources and achieve a more sustainable agriculture. Despite existing limitations, the Danube RB and its ICPDR constitute a good example to learn from, since our DEA analysis shows that it is an efficient managerial unit. Additionally, our results suggest that, at least in the case of the selected European RBs, the capacity to reduce agricultural negative pressures on water resources is more closely related to RB characteristics (e.g., area, population density, number of riparian countries) than to policy or institutional factors (e.g., EU membership, existence of TRBMIs). Findings offer highly relevant information to policy and decision makers, since they clearly suggest that current managerial initiatives are insufficient to achieve a sustainable agriculture and reduce the negative agricultural pressures on water resources. A more effective transboundary cooperation is required and initiatives focused on reducing agricultural pollution and efficient agricultural water use (thus reducing abstraction pressure) need to be implemented. In this sense, the Danube river basin offers some good learning examples of effective multi-country and multi-agent cooperation to address agricultural pressures, such as the Environmental Programme for the Danube River Basin (EPDRB) and the "Friends of the Danube" programme.

Finally, it is worth noting that the analysis carried out in this paper is of a static nature. Future research will focus on carrying out a dynamic analysis, which could offer additional significant information for the adaptation of RB management initiatives to a changing environment. Nevertheless, the TWAP data base has not been up-dated, being the latest data referred to year 2010, what constitutes a limitation to carry out a dynamic analysis. As soon as more data becomes available, we aim to evaluate dynamic changes in efficiency mappings and the factors that may explain these changes.

REFERENCES

Avkiran N, Rowlands T (2008) How to better identify the true managerial performance: state of the art using DEA. *Omega-Int J Manage S* 36(2):317–324.

Bakker MHN, Duncan JA (2017) Future bottlenecks in international river basins: where transboundary institutions, population growth and hydrological variability intersect. *Water Int* 42(4):400–424.

Banker RD, Chang H, Cooper W (1996) Simulation studies of efficiency, returns to scale and misspecification with nonlinear functions in DEA. *Ann Op Res* 66(4):231–253.

Banker RD, Charnes A, Cooper W (1984) Some models for estimating technical and scale inefficiencies in data envelopment analysis. *Manag Sci* 30(9):1078–1092.

Berbel J, Expósito A (2018) Economic challenges for the EU Water Framework Directive reform and implementation. *Eur Plan Stud* 26(1):20–34.

Bernauer T, Kuhn PM (2010) Is there an environmental version of the Kantian peace? Insights form water pollution in Europe. *Eur J Int Relat* 16:77–102.

Boyle S (2014) The case for regulation of agricultural pollution. *Environ Let Rev* 16:4–20.

Charnes A, Cooper W, Rhodes E (1978) Measuring the efficiency of decision making units. *Eur J Oper Res* 2(6):429–444.

Charnes A, Cooper W, Rhodes E (1981) Evaluating programme and managerial efficiency: an application of data envelopment analysis to Programme Follow Through. *Manag Sci* 27(6):668–697.

Dimitriou E, Mentzafou A, Zogaris S, Tzortziou M, Gritzalis K, Karaouzas I, Nikolaidis C (2012) Assessing the environmental status and identifying the dominant pressures of a trans-boundary river catchment, to facilitate efficient management and mitigation practices. *Environ Earth Sci* 66(7):1839–1852.

Dinar A (2008) *International Water Treaties. Negotiation and Cooperation along Transboundary Rivers*. New York: Routledge.

Dinar A, Dinar S, McCaffey S, McKinney D (2013) *Bridges over Water. Understanding Transboundary Water Conflict, Negotiation and Cooperation*. New Jersey: World Scientific.

EC (2012) A blueprint to safeguard Europe's water resources. Communication from the Commission to the European Parliament, the European Economic and Social Committee and the Committee of the Regions (COM (2012) 673). European Commission, Brussels.

EEA (2012a) European waters. Assessment of status and pressures. Report no 8/2012. Environment European Agency, Copenhagen.

EEA (2012b) European waters. Current status and future challenges. A synthesis. Report no 9/2012. Environment European Agency, Copenhagen.

EEA (2015) The European Environment. State and Outlook 2015. Synthesis Report. Environment European Agency, Copenhagen.

EU (2017) Agri-environmental schemes: how to enhance the agriculture-environment relationship. Science for Environment Policy. Thematic Issue 57. Issue produced for the European Commission DG Environment by the Science Communication Unit, UWE, Bristol.

Expósito, A, Pablo-Romero, MP, Sánchez-Braza, A (2019). Testing EKC for urban water use: Empirical evidence at river basin scale from the Guadalquivir River, Spain. *J Water Resour Plan Manag* 145: 4.

Expósito A, Velasco F (2018) Municipal solid-waste recycling market and the European 2020 Horizon Strategy: a regional efficiency analysis in Spain. *J Clean Prod* 172:938–948.

Expósito A, Velasco F (2020) Exploring environmental efficiency of the European agricultural sector in the use of mineral fertilizers. *J Clean Prod* 253:119971.

FAO (2017) *FAOSTAT Food and Agricultural Data*. Food and Agriculture Organization of the United Nations. Rome, Italy.

Farrell, M. J. (1957). The measurement of productive efficiency of production. *J R Stat Soc* A 120(III), 253–281.

Gerlak AK (2004) The global environment facility and transboundary water resource management: new institutional arrangements in the Danube river and Black Sea region. *J Environ Dev* 23(4):400–424.

Giordano M, Drieschova A, Duncan JA, Sayama Y, De Stefano L, Wolf AT (2014) A review of the evolution and state of transboundary freshwater treaties. *Int Environ Agreem-P* 14(3):245–264.

Jager NW (2016) Transboundary cooperation in European water governance. A set-theoretic analysis of international river basins. *Environ Policy Gov* 26:278–291.

Kallioras A, Pliakas F, Diamantis I (2006) The legislative framework and policy for the water resources management of transboundary rivers in Europe: the case of Nestos/Mesta River, between Greece and Bulgaria. *Environ Sci Policy* 9:291–301.

Kauffman GJ (2015) Governance, policy, and economics of intergovernmental river basin management. *Water Resour Manag* 29:5689–5712.

Knieper C, Pahl-Wostl C (2016) A comparative analysis of water governance, water management, and environmental performance in river basins. *Water Resour Manag* 30:2161–2177.

Libert B (2015) The UNECE Water Convention and the development of transboundary cooperation in the Chu-Talas, Kura, Drin and Dniester river basins. *Water Int* 40(1):168–182.

Mayorga E, Seitzinger SP, Harrison JA, Dumont E, Beusen AHW, Bouwman AF, Fekete BM, Kroeze C, Van Drecht G (2010) Global nutrient export from WaterSheds 2 (NEWS 2): Model development and implementation. *Environ Modell Soft* 25:837–853.

Mohaupt V, Crosnier G, Todd R, Petersen P, Dworak T (2007) WFD and agriculture activity of the EU: first linkages between the CAP and the WFD at EU level. *Water Sci Technol* 56(1):163–170.

Munia H, Guillaume JHA, Mirumachi N, Porkka M, Wada Y, Kummu M (2016) Water stress in global transboundary river basins: significance of upstream water use on downstream stress. *Environ Res Lett* 11:014002.

Ostrom E (2005) *Understanding Institutional Diversity*. Princeton, PA: Princeton University Press.

Özerol G, Bressers H, Coenen F (2012) Irrigated agriculture and environmental sustainability: an alignment perspective. *Environ Sci Policy* 23:57–67.

Petersen-Perlman JD, Veilleux JC, Wolf AT (2017) International water conflict and cooperation: challenges and opportunities. *Water Int* 42(2):105–120.

Pulwarty RS, Maria R (2015) Adaptation challenges in complex rivers around the world: the Guadiana and the Colorado basins. *Water Resour Manag* 29:273–293.

Romano G, Guerrini A, Marques RC (2017) European water utility management: Promoting efficiency, innovation and knowledge in the water industry. *Water Resour Manag* 31:2349–2353.

Seitzinger SP, Mayorga E, Bouwman AF, Kroeze C, Beusen AHW, Billen G, et al. (2010) Global river nutrient export: A scenario analysis of past and future trends. Global Biogeochem Cy 24:GB0A08.

Sueyoshi T, Goto M (2011) Measurement of returns to scale and damages to scale for DEA-based operational and environmental assessment: how to manage desirable (good) and undesirable (bad) outputs? *Eur J Oper Res* 211(1):76–89.

Sueyoshi T, Sekitani K (2009) An occurrence of multiple projections in DEA-based measurement of technical efficiency: theoretical comparison among DEA models from desirable properties. *Eur J Oper Res* 196(2):764–794.

TWAP (2019) Transboundary Water Assessment Programme. River Basins Component. UNEP and GEF. Available at http://twap-rivers.org

UNECE (1992) Convention on the protection and use of transboundary watercourses and international lakes. Helsinki, March 17, 1992, UN Economic Commission for Europe, United Nations.

UNECE (2011) Second assessment of transboundary rivers, lakes and groundwaters. EDE/MP.WAT/33, UN Economic Commission for Europe, United Nations.

UNEP (2016) Transboundary water systems – status and trends: crosscutting Analysis. United Nations Environment Programme (UNEP), Nairobi.

Van Rijswick M, Gilissen HK, Van Kempen J (2010) The need for international and regional transboundary cooperation in European river basin management as a result of new approaches in EC water law. *ERA Forum* 11: 129–157.

Wiering M, Verwijmeren J, Lulofs K, Feld C (2010) Experiences in regional cross border co-operation in river management. Comparing three cases at the Dutch–German border. *Water Resour Manag* 24(11):2647–2672.

Xiang Z, Chen X, Lian Y (2016) Quantifying the vulnerability of surface water environment in humid areas based on DEA method. *Water Resour Manag* 30:5101–5112.

Zawahri NA, Mitchell SM (2011) Fragmented governance of international rivers: Negotiating bilateral versus multilateral treaties. *Int Stud Quart* 55(3):835–858.
Zhu J (2016) *Data Envelopment Analysis: A Handbook of Empirical Studies and Applications*. New York: Springer.