Sveriges lantbruksuniversitet
Swedish University of Agricultural Sciences

WORKING PAPER
01/2018

Evaluation of Cost Efficiency in Hydropower-Related Biodiversity Restoration Projects in Sweden - *A stochastic frontier approach*

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Working Paper Series 2018:01
Uppsala 2018

ISSN 1401-4068
ISRN SLU-EJKON-WPS-1801

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Abstract

Various restoration projects intended to mitigate the adverse ecological effects of hydropower plants, e.g., restoration of fish habitats and spawning grounds, have been implemented in different parts of Sweden. However, it is unclear whether these projects are economically in line with least-cost principles. Therefore, we estimated the cost frontier function of the projects and predicted the corresponding efficiency level by a stochastic frontier analysis. The estimates are based on a survey data from 245 projects in Sweden that are carried out between 1986 and 2015. This dataset contains expert judgments on the effects of each projects in terms of different ecological indicators. The results indicated an evidence of cost inefficiency in the projects, which had an average efficiency score of 55%, suggesting potential to minimize cost efficiency loss by 45%. Factors such as project duration, project management class, and restoration measure type were statistically significant determinants of the cost inefficiency score. Notably, projects owned by private and non-government principals showed better performance than projects owned by municipalities and national authorities such as the Swedish Forestry Agency and the Swedish Transport Administration.

Key words: Hydropower, biodiversity restoration, cost efficiency, stochastic frontier analysis

JEL: D24, Q25, Q57

1. Introduction

Hydropower is a vital source of energy supply in Sweden. Official reports indicate that energy production from hydro sources supplied nearly 61 TWh in 2016 alone which corresponds to 41% of total electricity production (SEA, 2015). The hydropower energy source is well known for its minimal emissions of pollutants and low production costs, and is an effective mechanism for controlling the significant fluctuations in energy demand and supply (Sparrevik et al., 2011; Rudberg et al., 2014). Based on these advantages and substantial generation potential, in recent decades Sweden has adopted a policy to promote a clean, renewable energy supply from hydropower. However, there has been a growing criticism of power generating hydropower dams due to their distortion of ecological conditions in the riverine landscape. For instance, streams can be entirely or partly desiccated, thereby destroying habitats and migration pathways for fish species. In this regard, approximately 2000 water bodies in Sweden do not meet the requirement of sufficient ecological status (EU, 2000). Furthermore, fish species such as eel and salmon, which are protected under the EU Habitat Directive (Council Directive 92/43/EEC 1992), are affected by hydropower plants (Hav, 2014).

Restoration measures aimed at improving biodiversity, such as stabilization of channels and improvement of riparian and in-stream habitats and water quality around hydropower plants, usually require a considerable amount of investment. Before restoration projects are implemented, economic and ecological aspects of these projects need to be evaluated, in order to utilize the limited investment resources most effectively. In this regard, a number of studies have assessed the ecological effects of different restoration measures(Green and O'Connor, 2001; Pejchar and Warner, 2001; Polasky, 2009; Renflt et al., 2010; Miteva et al., 2012; Fooks et al., 2017), while few have estimated the costs of restoration projects(Bullock et al., 2011; Nebhver et al., 2011; Rudberg et al., 2014).

In the past decade, there has been a growing trend for the application of stochastic frontier analysis (SFA) and data envelope analysis (DEA) to examine economic and environment-related problems (Reinhard et al., 1999; Bravo-Ureta et al., 2012; Huang et al., 2016). For instance, Reinhard et al. (1999) used SFA to estimate technical and environmental efficiency associated with dairy farms in Netherlands, while Bravo-Ureta et al. (2012) used SFA as an impact evaluation tool in natural resource management to compare technical efficiency between a treatment and control group in the MARENA1 program in Honduras. Furthermore, eco-efficiency and environmental efficiency analysis has been used in economics literatures as a tool of evaluating ecological and environmental performances (Huppes and Ishikawa, 2005;

Kuosmanen and Kortelainen, 2005; Huang et al., 2016). To the best of our knowledge, no previous study has assessed the cost efficiency of biodiversity restoration projects around hydropower plants, measured as the targeted ecological effects in relation to restoration costs.

A common approach in environmental applications of SFA is to treat pollution as an additional input into firms' production of goods and services, or as an undesirable output together with the desirable outputs (for a review, see Tyteca, 1996; Lansink and Wall, 2014). Other environmental applications include evaluation of ecological status with respect to some economic performance and efficiency measure, e.g., ecological effect per unit cost or economic value per unit ecological effect (Kuosmanen and Kortelainen, 2005). In the present analysis, we employed the so-called frontier eco-efficiency (FEE) model, which relates the costs of different measures to their corresponding ecological effect (see Robaina-Alves et al., 2015). To this end, we evaluated whether costs associated with biodiversity restoration measures were technically cost efficient or not, considering the desired environmental targets. Where there was evidence of cost inefficiency, we estimated its connection with project- specific characteristics, restoration class, and institutional factors such as project ownership by private, public, or non-government organizations (NGO). Our analysis was based on micro-data collected from biodiversity restoration projects at hydropower plants in different parts of Sweden.

In our view, the novel contribution of this study is application of the FEE model to evaluate the cost efficiency of hydropower related biodiversity restoration projects, hence aiding policy design for cost-effective implementation of biodiversity restoration projects. The remainder of the paper is structured as follows: Section 2 presents a brief preliminary analysis of biodiversity restoration projects in Sweden. Section 3 presents descriptions of the methodological framework, including the theoretical foundations of stochastic frontier cost function, while the econometric specification, results, and a discussion are presented in Section 4. Some conclusions are drawn in section 5.

2. Preliminary analysis

Data for this study were taken from two main sources: the national database for restoration measures (CBJ, 2016) and a survey of 275 hydropower plants in Sweden (Sandin et al., 2017). The official database includes information on types of measures (construction of technical natural fishway, road culvert, instream and spawning area restoration, and dam removal), the timing of the project, the principal (county board, municipality, NGO, private firm, or others), and cost. The projects were implemented over a 30 year period, between 1985 and 2015, but all costs were adjusted using the 2016 consumer price index. The survey by Sandin et al. (2017) included questions on the ecological effects of different biodiversity restoration measures.

The distribution of hydropower generation plants in Sweden varies across counties, with a dominance of small plants in southern and central Sweden, and large plants in the north (Widmark, 2002). Particularly, there is highest number of small hydropower plants in Jönköping County, which has at least twice as many as other counties. In connection, the survey by Sandin et al. (2017) showed that the majority of biodiversity restoration projects are located in southern Sweden, while few plants are located in the north-eastern parts of the country (see Fig. 1).

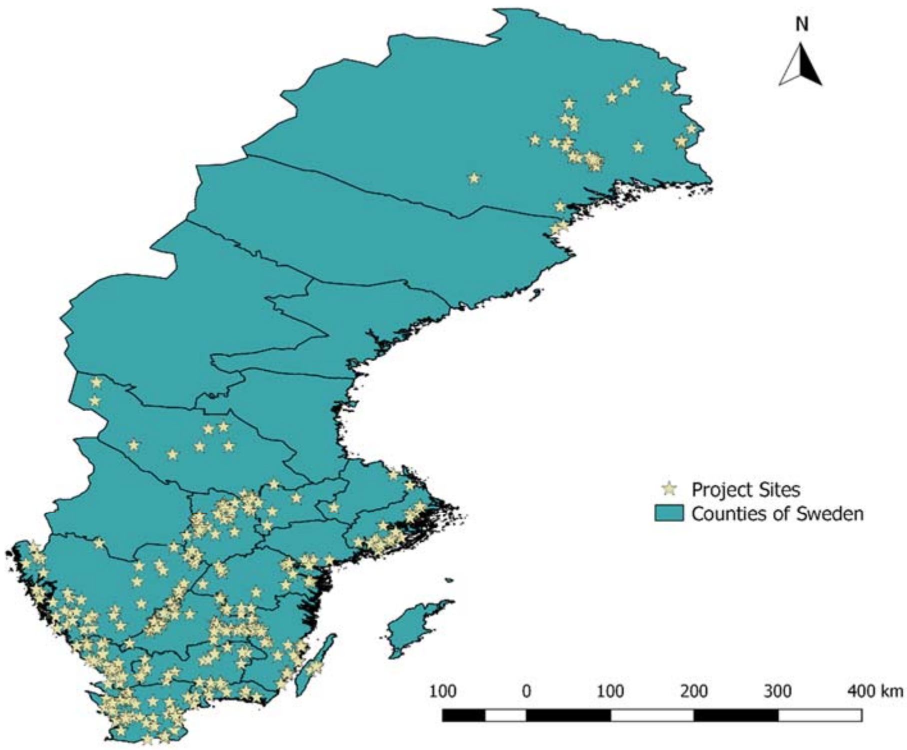

Figure 1: Location of biodiversity restoration measures at hydropower plants in Sweden.

Two main classes of restoration measures are included in the survey and the national database: i) connectivity improvements in the catchment and ii) restoration of habitat and spawning conditions in downstream regions. In the present study, we included four measures of connectivity improvement (technical fishways, natural fishways, dam removal, road culverts) and two measures linked to biotope improvement (habitat restoration, spawning restoration). These projects were implemented by five different categories of principals: county boards, municipalities, NGOs, private entities, and others. NGOs include local organizations for water and fish management, while private actors can be individuals but also firms such as hydropower producers and forest companies. Others consist of government authorities, such as the Swedish Forest Agency (Skogsstyrelsen) and the Swedish Transport Agency (Transportstyrelsen). In total, the dataset included 487 different hydropower plants disturbed throughout the country. The relative proportions of different restoration measures and principals are shown in Fig. 2.

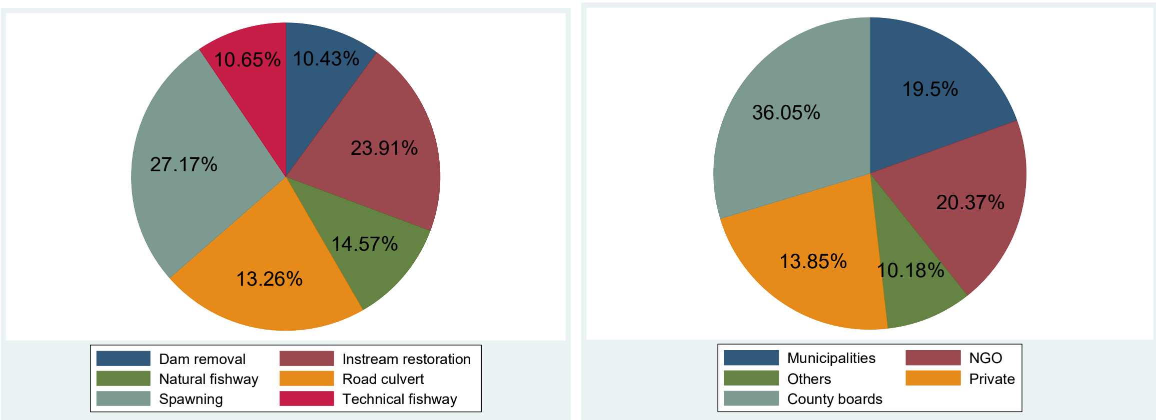

Figure 2: Relative proportions of (left) restoration measures and (right) principals.

Each of the two classes of biotype improvement measures (habitat and spawning restoration) accounted for approximately half the total number of measures. Municipalities, county boards, and NGOs were responsible for nearly half of the projects. Municipalities were the main principals of investments in technical fishways and dam removal and, together with NGOs, accounted for a significant proportion of projects targeting spawning improvements and instream restoration (Fig. 3).

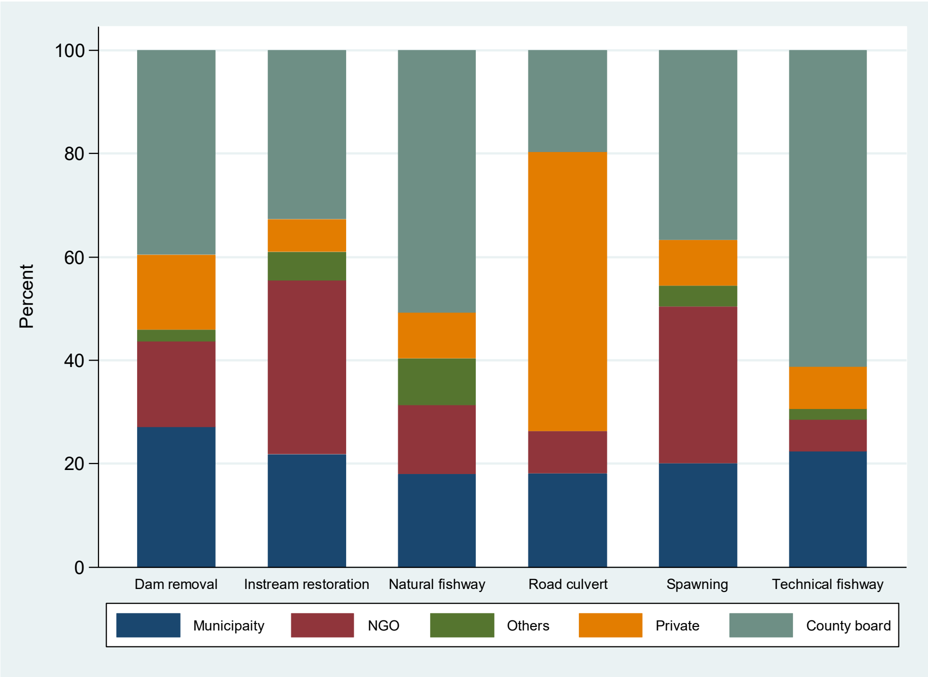

Figure 3: Types of restoration projects invested in by different classes of principals.

Data on costs were obtained from CBJ (2016) and included the principal's total operating costs for implementing and managing the measures. Costs in terms of impacts on hydropower plants' provision of energy were not included, which implies underestimation of the overall costs. This may be of particular importance for measures restoring connectivity in the landscape. Given this caveat, the costs per measure for different measures are displayed in Table 1. The average cost for all measures amounted to 298 000 SEK[1]. Construction of natural fishways was the most expensive measure, while improving spawning grounds was the least costly measure, representing around 17% of the cost of a natural fishway (Table 1).

It can be argued that it is not the cost per measure that is important, but rather the cost in relation to ecological effects. Data on ecological effects of the projects would require measurements of ecological status before and after the implementation of the restoration project. Biodiversity recovery of restoration project may take time, which would necessitate repeated measurements at the sites. Such data is not available, and we therefore used results from a survey of experts at county boards, which included questions on the perceived ecological impact of restoration projects (Sandin *et al.*, 2017). The responses were scaled from -10 to 10, where 10 is the best achievement. However, the survey did not contain any instructions on how to scale the effects, and the responses therefore rest on the experts' subjective evaluation. For each hydropower

[1] 1 Euro ≈ 9.47 SEK with the 2016 average exchange rate.

plant, the survey data contain responses on several indices of perceived ecological effect. One is the effect on the primary target of the project, such as improvements in trout, salmon, or eel, while others include five additional ecological effects. Since both targets and additional effects may impact the decision on project investment, we included both aspects. To reduce the number of variables, we constructed a weighted index of the five other ecological effects by employing principal component analysis (see e.g., OECD, 2008). Two different ecological effect variables were then constructed: *Targeffect*, which includes only the effect on the target for the restoration, and *Toteffect*, which adds the constructed index on other effects to the index on the targeted effect. Data on ecological effects are not available for all hydropower plants with cost information in the national database, but for 275 of these plants.

Table 1: Description of costs by restoration measures in SEK (2016 prices).

Measure type	Statistic	Cost per measure	Cost per *Targeffect*	Cost per *Toteffect*
Dam removal	Mean	563534	86927	72919
	Obs.	48	31	31
Instream restoration	Mean	254308	109384	87662
	Obs.	110	54	54
Natural fishway	Mean	588821	146320	114583
	Obs.	67	37	37
Road culvert	Mean	183965	74256	70854
	Obs.	60	21	22
Spawning	Mean	101698	26858	22937
	Obs.	124	72	72
Technical fishway	Mean	375682	116311	80013
	Obs.	49	32	32
Total	Mean	298104	85953	68566
	Obs.	458	247	248

Cost per ecological effect for the restoration measures was similar for both *Targeffect* and *Toteffect*; unit costs were highest for technical fishways and considerably lower for improvement of spawning conditions. On average, the cost per *Targeffect* was approximately 5.5-fold higher for construction of natural fishways than for spawning improvements. The unit costs were reduced for all measures when *Toteffect* was used instead of *Targeffect.* The reduction was relatively greater for the most costly measures, since other ecological effects were greater and the difference between the lowest and highest cost was then reduced. Comparison of cost per *Toteffect* among principals showed that NGOs had the lowest cost per effect for all measures (Fig. 4).

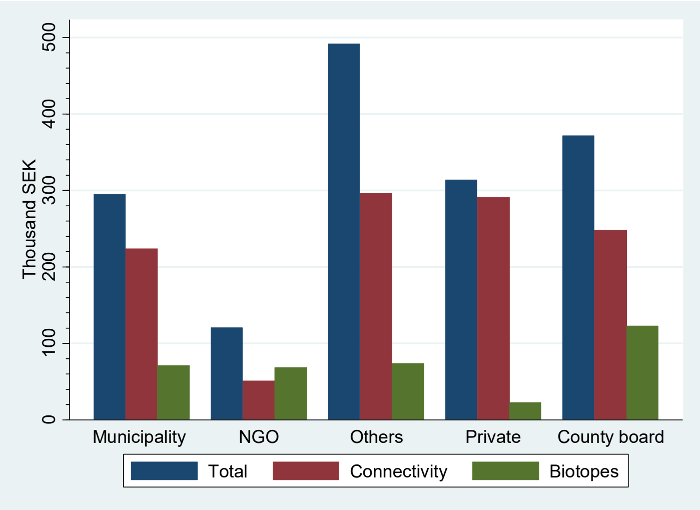

Figure 4: Cost per restoration effects across principals and measure types.

3. Theoretical framework

3.1. Formulation of stochastic frontier cost function

Following the seminal works of Aigner *et al*. (1977), Battese and Corra (1977), Meeusen and van Den Broeck (1977) and, later, Førsund *et al*. (1980), Schmidt and Lin (1984), and Greene (2008), applications of SFA have become popular in measuring firms' technical efficiency or productivity level. The economic reasoning behind technical efficiency is directly linked to how a firm utilizes an existing limited resource to produce a maximum level of output. Thus, a producer is said to be technically efficient if a firm is producing maximum output from a given input level combination. Specifically, SFA distinguishes the actual and potential value of output or cost in the production process.

In this study, we specified a twice-differentiable Cobb-Douglas frontier cost function corresponding to biodiversity restoration projects. The cost function is based on duality theory in microeconomics, where available technology can be indirectly represented by the conditional input demand function for given exogenous prices and optimal behavior of the producer (Diewert, 1982; Varian, 1992; Resti, 1997; Shepherd, 2015). The theoretical representation of the stochastic frontier cost function is written as:

$$C_i = f\left(q_i, w_i, r_i\right)\exp\left\{\varepsilon_i\right\} \tag{1}$$

where C is the cost of biodiversity restoration, q represents ecological output, w is wage rate for labor, r is interest rate, and $i=1,..,N$ are restoration projects. The term ε represents the error term, which is divided into inefficiency, u, and statistical noise, v, *i.e.*:

$$\varepsilon_i = u_i + v_i \tag{2}$$

In equation (2), v is assumed to be normally distributed as $N(0,\sigma^2)$ and u is a positive random *i.i.d* error term, which can follow a half-normal, truncated, exponential, or gamma distributions (Wang and Schmidt, 2009). The term u denotes a non-negative deviation from the frontier cost function, i.e., minimum cost estimated for a given level of output and input prices.

The economic reasoning behind equation (2) is directly linked to the existence of two distinguishable stochastic random error components in the specified cost function. The first part, u, represents cost inefficiency that arises due to a number of project-specific factors in the restoration process. The second component, v, represents stochastic noise that cannot be controlled by a firm, such as climate and any accidental disaster. Hence, the existence of cost inefficiency can be reflected by an upward deviation from the minimum possible cost for a given ecological production. Consequently, the level of cost efficiency, *CE*, associated with each project is predicted by taking the ratio of the frontier (or possible minimum cost), C_i*, and the corresponding observed cost level, C_i, calculated as:

$$CE_i = \frac{C_i^*}{C_i} = \frac{f(X;\beta)\exp(v_i)}{f(X;\beta)\exp(u_i + v_i)} = \exp(-u_i) \tag{3}$$

where β denotes a vector of parameters to be estimated. $u_i = 0$ implies full efficiency and the amount by which equation (3) is less than one represents the degree of cost inefficiency.

3.2. Estimation technique

A maximum likelihood estimator (MLE) is commonly applied to estimate the stochastic frontier cost function given by equation (1). It provides estimates of a linear cost function with a disturbance that is assumed to be a mixture of two components, which have a strictly non-negative and symmetric distribution, respectively. Accordingly, a likelihood function is represented by the variance parameters, $\sigma^2 \equiv \sigma_u^2 + \sigma_v^2$ and $\lambda = \sigma_u / \sigma_v$ (see Aigner *et al.*, 1977). The term σ^2 denotes a total error variance in equation (3), while λ is the ratio of the standard deviations of the inefficiency and idiosyncratic components. This shows the contributions of each components in the total variation.

The parameter $\gamma = \sigma_u / (\sigma_u + \sigma_v)$ is used for testing the existence of cost inefficiency (Battese and Corra, 1977). Rejecting the null hypothesis of $\gamma = 0$ confirms the existence of cost inefficiency when the model fits half-normal distribution. However, in the case of more complicated models (such as truncated normal), a log-likelihood-based test for inefficiency is recommended, as the gamma parameter does not provide essential information on the existence of a one-sided error term (Kumbhakar *et al.*, 2015). In general, accepting the null hypothesis implies absence of cost inefficiency that the variation in the total error term, ε is attributable to the statistical noise component, and thus equation (1) can be estimated using linear least square regression method.

4. Econometric specification, data retrial and descriptions

4.1. Regression model

Based on the theoretical model given in equation (1), the cost frontier model to be estimated can be specified considering ecological output, input prices, i.e., wage, and interest rate as covariates explaining biodiversity restoration cost. Assuming each project chooses allocation of measures that minimize total cost, the frontier cost function is given as:

$$\log C_i = \beta_0 + \beta_1 q_i + \beta_2 \log w_t + \beta_3 \log r_t + \beta_4 D_j + u_i + v_i \tag{4}$$

where, C_i is an expenditure on each biodiversity restoration projects indicted by an index of $i= 1, 2, ..., N$ and q_i is ecological output measured as an expert judgment on the performance of biodiversity restoration projects. We further classified ecological output as *Targeffect* and *Toteffect*. Input prices, i.e., wages and interest rate, are represented by w_t and r_t, respectively which are indexed by the project implementation year, $t \in [1985, 2015]$. The variable D_j for $j = 1, 2, ..., 15,$ represents a dummy that controls any potential unobserved heterogeneity across counties. The term u_i and v_i denotes cost inefficiency and random noise components, respectively, while β_i is a parameters to be estimated.

Identifying the determinants of cost inefficiency is important for effective resource utilization and policy formulation with respect to biodiversity restoration measures. Hence, we obtained policy relevant variables by regressing the predicted cost inefficiency score on institutional and project-related specific variables such as project management, class of restoration measures, and project duration[2]. Project management, such as county board, private, municipalities, NGOs, and other agencies, may explain cost inefficiency associated with biodiversity restoration projects. In relative terms, we expected lower cost inefficiency for projects operated by private principals than those run by public principals. This is directly linked to the profit-maximizing behaviour of private agents, whereby they are expected to be more efficient in resource management than public principals.

Classes of restoration measures could also be linked to the level of cost inefficiency. Furthermore, project duration could have a positive or negative effect on the level of cost inefficiency. If a project runs for a long period, this could provide potential for learning where project owners can decrease cost inefficiency. However, the positive effect of project duration could reflect additional spending by principals in order to maintain the planned amount of operation. Therefore, the model for explaining the determinants of cost inefficiency is given in equation (5) considering institutional and project-specific characteristics as explanatory variables:

$$CI_i = \phi_0 + \sum_{k=1}^{K} \phi_i z_i + v_i, \quad \mathrm{i} = 1, 2, ..., N \tag{5}$$

where CI_i represents the predicted cost inefficiency and z_i denotes institutional and project-specific characteristics that affect cost inefficiency. These include project ownership, restoration class, and duration of project. The term v_i is an idiosyncratic error component. The terms ϕ_i are parameters to be estimated.

4.2. Data retrieval and description

In addition to the data on costs and ecological effects presented in section 2 of this paper, we extracted data on wages and capital costs from official sources. We used data on average monthly salary from Swedish Statistics and we used the return on a relatively risk-free asset, short-term government bonds, as a measure of the opportunity cost of capital (SCB, 2016). We chose the return on short-term government bonds due to the condition that the opportunity cost of investing in risky capital is proportional to the potential return on risk-free investment such as government bonds and treasury bills. Descriptive statistics are displayed in Table 2.

[2] This follows one-step estimation of stochastic frontier cost function where parameters for cost frontier and inefficiency determinists are obtained from simultaneous estimation of the model.

The data indicated substantial variation in the distribution of biodiversity restoration projects across counties, as a substantial number of small hydropower dams are located in Jönköping County (see Fig. A1 in the Appendix). This could have an implication for the estimates, for instance concentration of small hydropower plants in a given county could result in higher investment costs for the individual principals than in other regions. To account for such heterogeneity, we introduced a county-level dummy in our regression model (Table 2).

Table 2: Summary of descriptive statistics

Variable	Obs.	Mean	Std. Dev.	Min	Max
Total cost (SEK)	458	298104	675320	2060	5994082
Wage rate (SEK/month)	455	164	16	115	184
Interest rate (%)	487	0.986	1.236	-1.116	6.710
Targeffect	275	5.204	3.437	-10	10
Toteffect	275	6.287	4.251	-12.09	14.24
Project duration(Years)	460	0.964	1.655	0	24.07
Restoration class	491	1.521	0.500	1	2
Management	491	3.265	1.582	1	5
Measure type	460	3.548	1.594	1	6
County	460	8.263	4.037	1	15
Year	466	2006	5.567	1985	2015

5. Results

Following the empirical specification in equations (4) and (5), we estimated the cost frontier function classified into *Toteffect* and *Targeffect* models. Columns (1) – (4) in Table 3 show estimates for the *Targeffect* model, while columns (5) – (8) show estimates for the *Toteffect* model. The estimation procedure followed the one-step maximum likelihood estimation of the stochastic frontier model suggested by Wang and Schmidt (2002). This approach was chosen since it addresses the potential bias in parameters due to possible correlation between regressors of the cost frontier function and inefficiency determinants.

Table 3: One-step maximum likelihood estimates of cost frontier function.

Variables	The dependent variable is Log(Cost)							
	Targeffect model				Toteffect model			
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
Cost frontier variables								
Log (wage)	2.521***	2.598***	2.564***	2.499***	2.514***	2.592***	2.558***	2.494***
	(0.300)	(0.234)	(0.223)	(0.194)	(0.294)	(0.231)	(0.223)	(0.196)
Log (interest rate)	0.167***	0.058	0.054	0.047	0.164***	0.054	0.050	0.042
	(0.043)	(0.142)	(0.135)	(0.168)	(0.040)	(0.141)	(0.134)	(0.169)
Targeffect	0.059*	0.064***	0.065***	0.067**				
	(0.030)	(0.020)	(0.021)	(0.029)				
Toteffect					0.053***	0.059***	0.060***	0.061***
					(0.018)	(0.012)	(0.014)	(0.021)
County dummy	YES	YES	YES	YES	YES	YES	YES	YES
Inefficiency determinant variables								
Management								
NGO	-14.81***	-3.096***	-2.708***	-2.999***	-14.286***	-3.022***	-2.624***	-2.892***
	(1.074)	(0.576)	(0.570)	(0.460)	(1.079)	(0.604)	(0.631)	(0.540)
Others	1.049***	0.582***	0.600***	0.230	1.070***	0.607***	0.628***	0.257
	(0.159)	(0.153)	(0.048)	(0.155)	(0.163)	(0.155)	(0.044)	(0.156)
Private	-0.488***	-1.600***	-1.321***	-1.036***	-0.461***	-1.579***	-1.299***	-1.015***
	(0.008)	(0.112)	(0.080)	(0.120)	(0.032)	(0.139)	(0.114)	(0.129)
County board	0.588***	-0.240***	-0.171***	-0.149	0.620***	-0.210**	-0.140***	-0.121
	(0.056)	(0.079)	(0.019)	(0.103)	(0.088)	(0.104)	(0.048)	(0.083)
Project duration		1.019***	0.783***	0.716***		1.020***	0.781***	0.715***
		(0.013)	(0.023)	(0.167)		(0.022)	(0.005)	(0.177)
Restoration class								
Connectivity			0.896				0.907*	
			(0.550)				(0.538)	
Measure Type								
Instream restoration				-0.034				-0.045
				(0.360)				(0.328)
Natural fishway				0.749***				0.731***
				(0.003)				(0.013)
Road culvert				0.193				0.186
				(0.624)				(0.656)
Spawning				-1.740***				-1.736***
				(0.467)				(0.494)
Technical fishway				-0.092				-0.054
				(0.246)				(0.225)
Constants	0.265	0.838***	0.326	0.966***	0.241	0.814***	0.295	0.951***
	(0.182)	(0.186)	(0.277)	(0.042)	(0.199)	(0.191)	(0.277)	(0.071)
Vsigma	0.424*	0.337	0.307	0.270	0.418*	0.333	0.303	0.266
	(0.240)	(0.240)	(0.246)	(0.206)	(0.238)	(0.240)	(0.242)	(0.203)
Mean efficiency	0.375	0.534	0.527	0.547	0.369	0.535	0.528	0.546
Cluster	Municipal	Municipal	Municipal	Municipal	Municipal	Municipal	Municipal	Municipal
Loglikelihood	-495.93	-421.74	-419.66	-415.79	-495.77	-421.41	-419.28	-415.54
AIC	995.86	847.48	843.32	833.58	995.55	844.82	840.57	835.08
BIC	1003.09	854.48	850.33	837.08	1002.78	848.32	844.07	842.08
Observation	275	245	245	245	275	245	245	245

Clustered standard errors in parentheses. *** p<0.01, ** p<0.05, * p<0.1. The reference for management, measure type and restoration class variables are municipalities, dam removal and biotopes, respectively. The inefficiency component follows the assumption of truncated normal distribution.

We also compared the distributional assumptions associated with the inefficiency component, u_i in equation (2). The implemented log-likelihood ratio test for model selection favored truncated normal assumption instead of half-normal and exponential assumptions[3].

Checking the existence of inefficiency in the estimation of stochastic frontier models is essential. Thus, we implemented a generalized log-likelihood ratio test to check whether there is cost inefficiency in all specifications. This procedure is preferred in the case of a truncated

[3]The log-likelihood ratio test statistics and results of detailed robustness tests are given in Section 4.6.

normal assumption, as the one-sided error term, u_i follows a mixed chi-square distribution (Kodde and Palm, 1986). The test has two degrees of freedom, since the null hypothesis has two restrictions: $\sigma_u^2 = 0$ and $\mu = 0$, where μ denotes mean of the one-sided error term. Accordingly, the null hypothesis of no cost inefficiency was rejected (at P<0.1) in all specifications, suggesting presence of cost inefficiency across all the biodiversity restoration projects studied. Overall, the *Targeffect* and *Toteffect* models produced similar results with respect to the estimated coefficients and significance level. Our main variables of interest that represented ecological outputs, i.e., *Targeffect* and *Toteffect*, were both positive and statistically significant (at least at P<0.1) in all specifications.

Comparing the reported information criteria, i.e., Akaike information criterion (AIC) and Bayesian information criterion (BIC), we found that the column (4) and (8) specifications in Table 3, corresponding to the *Targeffect* and *Toteffect* models, respectively, were preferable. These specifications had the lowest information criteria values and thus the corresponding estimates were robust in representing the data we used. Considering selected specifications, a 1% increase in *Targeffect* and *Toteffect* resulted in a 7% and 6% increase in restoration costs, respectively. Similarly, a 1% SEK increase in the average wage and interest rates led to a 2.5% and 0.05% increase, respectively, in biodiversity restoration costs in *Targeffect* specifications. On the other hand, a 1% rise in wage and interest rates led to a 2.5% and 0.04% rise, respectively, in *Toteffect*.

Results for the determinants of cost inefficiency, estimated jointly with the frontier cost function, are presented in Table 3. In both the *Targeffect* and *Toteffect* models, estimates for project management by NGOs and private principals were negative and highly statistically significant (*P<0.01*). This suggests that biodiversity restoration projects owned and managed by NGOs, private parties, and county boards have a higher likelihood of being more cost-efficient than those projects operated by municipalities. However, the results also indicated that projects implemented by others, such as the Swedish Forestry Agency and Swedish Transport Agency, had a higher probability of being cost-inefficient relative to those projects operated by municipalities. When we controlled for the effect of project duration on the technical inefficiency level associated with biodiversity restoration projects, we found that the estimates became positive and statistically significant (*P<0.01*), indicating that lengthy project duration results in higher cost inefficiency.

We also controlled for the potential effect of restoration within specific measure types. The corresponding estimates for natural fishway projects were found to be positive and statistically significant (*P<0.01*), suggesting there is a higher probability of these projects being cost-inefficient than projects linked to dam removal. However, projects linked to spawning had a lower probability (*P<0.01*) of being cost-inefficient than dam removal projects. Overall, implementation of the project by NGOs and private principals induced lower cost inefficiency associated with biodiversity restoration projects. However, projects implemented by others, such as the Swedish Forestry Agency and Swedish Transport Administration, had a higher chance of being cost-inefficient than projects implemented by municipalities.

6. Predicted cost efficiency score and robustness checks

6.1. Cost efficiency score

Based on the evidence of presence of cost inefficiency, we predicted the magnitude of efficiency score associated with individual biodiversity restoration projects. On average, restoration projects had an efficiency score of 55% in both the *Toteffect* and *Targeffect* models. This suggests a substantial variation in efficiency score between the most and least efficient projects. We also investigated the distribution of cost efficiency scores across different measure types and management categories, to examine whether there was substantial variation.

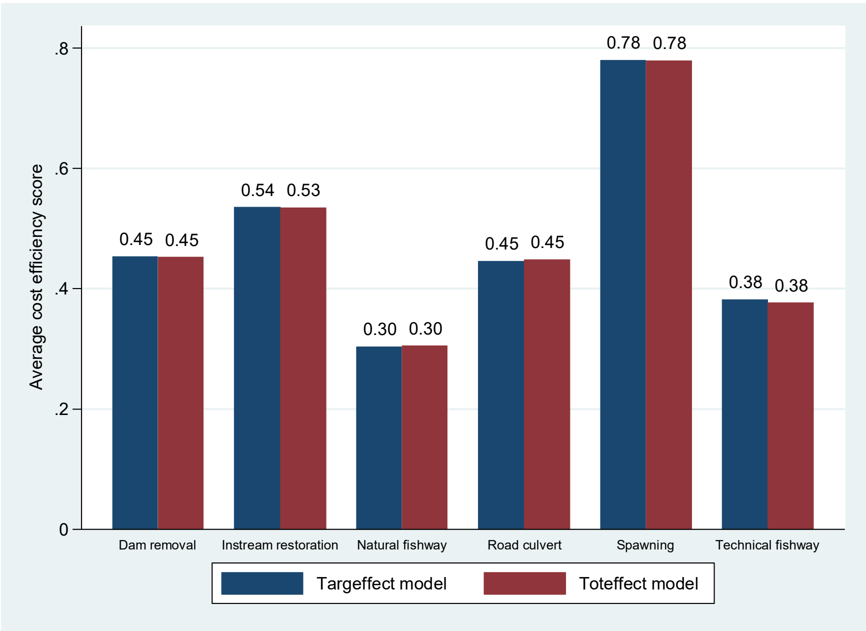

Figure 6: Cost efficiency score of different restoration measure types.

In this regard, the *Targeffect* and *Toteffect* models gave a similar score for both measure type and management classification (see Fig. 6). Specifically, there was a significant disparity between different restoration measure types, with the efficiency score for spawning and natural fishway projects being 78% and 30%, respectively. The cost efficiency scores also showed substantial variation between principals and management categories (Fig. 7). For instance, projects managed by NGOs and private sectors had an average cost efficiency score of 87% and 59%, respectively, while projects operated by others, such as the Swedish Forestry Agency and Transport Administration, had an average cost efficiency score of 31% (Fig. 7).

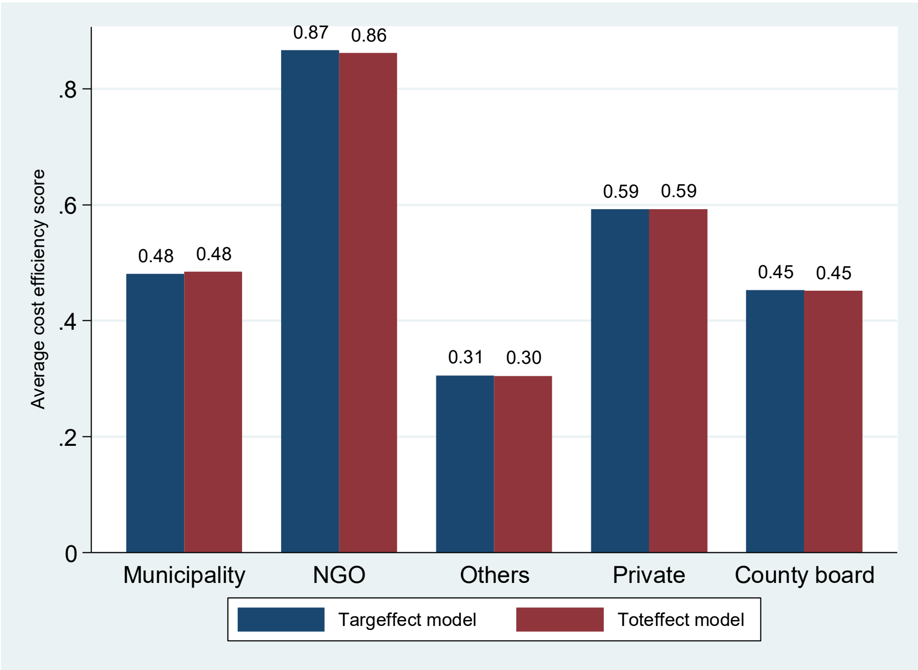

Figure 7: Cost efficiency score of different project management types.

The indicated disparity of efficiency distribution between principals could be attributable to a number of factors. For instance, better performance by privates might be an indication of efficient utilization of resources due to their profit-maximizing behavior. Meanwhile, there is low efficiency score by municipalities, county boards, and others. These projects are relatively expensive compared to those implemented by NGOs (see figure 4), suggesting these principals are not enjoying economies of scale with respect to the large amount of investment.

6.2. Robustness and hypothesis testing

Prior to maximum likelihood estimation, it is essential to test the OLS residuals skewness in order to endorse whether the specification of stochastic cost frontier function is valid (see Olson *et al.*, 1980; Waldman, 1982; Schmidt and Lin, 1984; Kumbhakar *et al.*, 2015). The maximum likelihood estimates are consistent if the distribution of OLS residuals are skewed to the right, i.e., positive skewness. The corresponding skewness test statistics showed a positive sign in both the *Toteffect* and *Targeffect* models (*skewness=0.112 and skewness=0.115*, respectively), as was expected. Thus, the maximum likelihood estimation was in line with a stochastic cost frontier specification.

Choosing an appropriate functional form is also essential before estimating the cost frontier function. In this regard, we implemented the log-likelihood ratio (LR)-based test and found that the corresponding null hypothesis supporting the trans-log functional form was rejected ($P<0.1$) in both the *Toteffect* and *Targeffect* models, favoring the Cobb-Douglas functional form of the cost frontier speciation. We also employed an LR test to compare three distributional assumptions (half-normal, truncated normal, exponential distribution) on the efficiency score and found that the truncated normal assumption fit the data well. The density plots of each distributional assumption confirmed absence of outlier effect in the estimates of cost frontier function (Fig. 8).

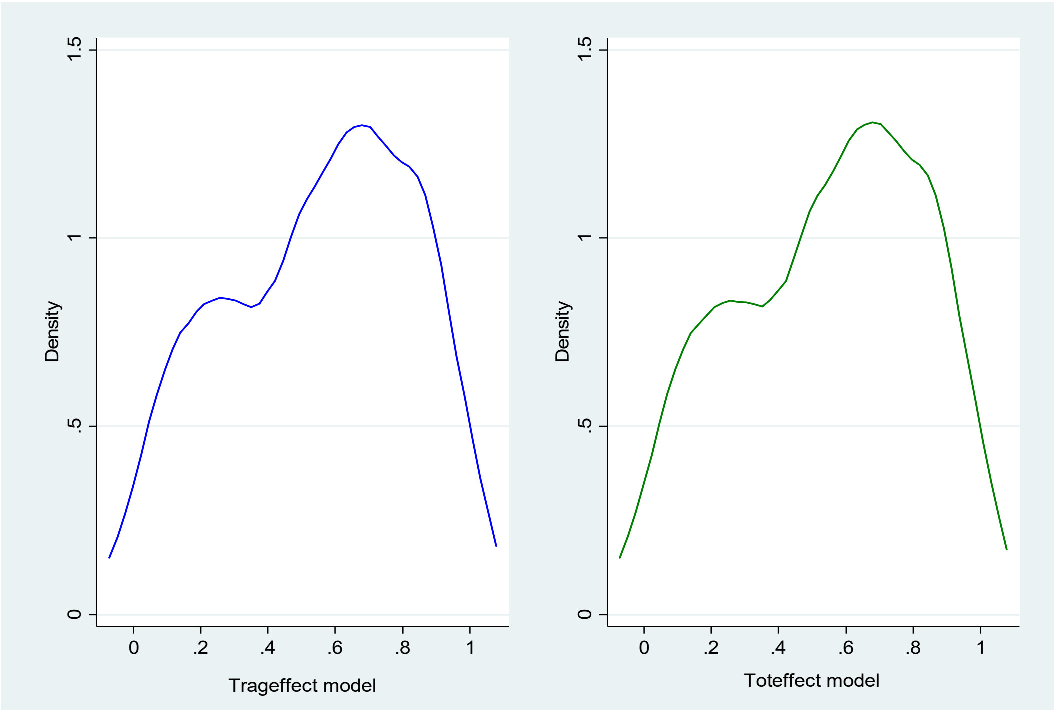

Figure 8: Kernel density plot of the predicted technical efficiency scores.

7. Conclusions

This study evaluated cost efficiency in biodiversity restoration projects at hydropower plants in Sweden using stochastic frontier analysis. To this end, we used data on costs and ecological effects of 275 different restoration measures obtained from official statistics and from a survey of hydropower plants. The data showed differences in average costs for different biodiversity improvement measures and project principals. Measures improving connectivity in the catchment were more expensive than measures improving biodiversity in degraded waters. The data also showed that the average cost of projects run by NGOs was lower than that of projects run by other principals (county board, municipality, private entities, others).

Two measures of ecological outputs were constructed: an index of targeted effect (*Targeffect*) and an index of total effects (*Toteffect*). Econometric analysis of all restoration measures indicated that elasticity of costs with respect to the *Toteffect* and *Targeffect* specifications was relatively similar, ranging in magnitude from 0.07-0.11 and 0.07-0.08 for *Targeffect* and *Toteffect*, respectively. Presence of inefficiency was tested for all projects by applying stochastic frontier analysis. A major finding was that the null hypothesis of no cost inefficiency was rejected in both the *Targeffect* and *Toteffect* models. The estimated average cost efficiency score for individual biodiversity restoration projects was around 55% in both models, suggesting substantial variation between the most and least efficient projects.

Estimates of the determinants of cost inefficiency showed that institutional factors such as project ownership significantly contributed to variation in inefficiency level. In particular, biodiversity restoration projects owned and managed by NGOs, private entities, and county boards had a higher likelihood of being cost-efficient than projects operated by municipalities. A potential conclusion based on the results in this study is that the total cost of biodiversity restoration at hydropower plants in Sweden could be reduced by reallocation of measures and project owners. However, our data on ecological effect rest on experts' subjective evaluation on a Likert scale, without any instructions on how to assess the effect. It is therefore unclear if and how differences in spatial and dynamic scales of the ecological effects between restoration measures are considered. This point out the need for data based on measurements and assessments of ecological status at the sites before and after implementation of the restoration projects.

Appendix

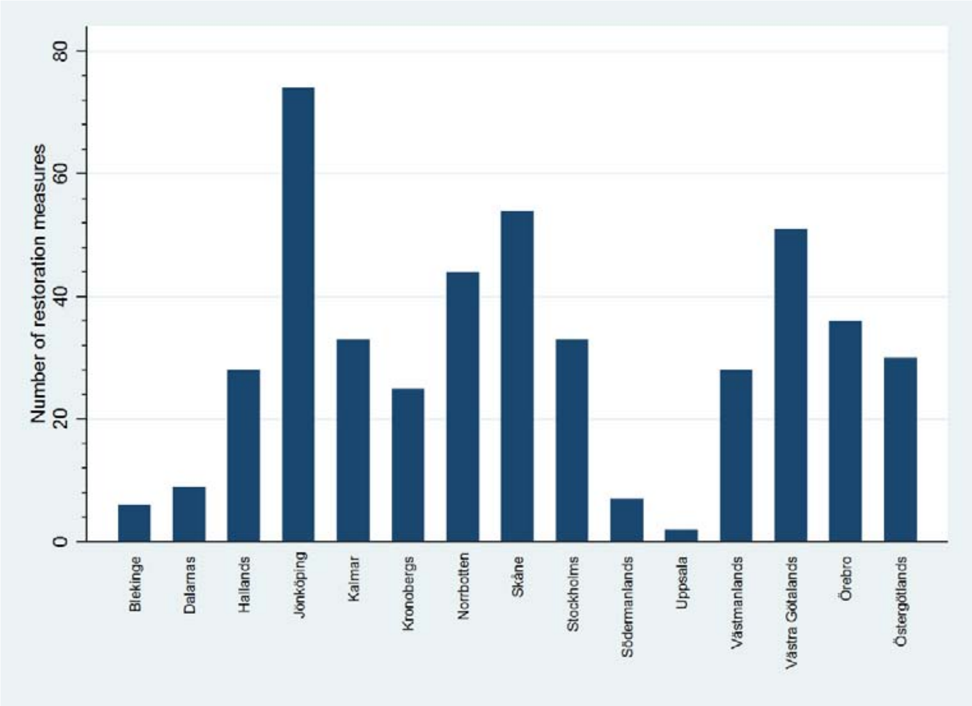

Figure A1: Number of restoration measures performed in different counties of Sweden.

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