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Economic Perspectives on Environmental Policies: The Costs and Benefits of Environmental Regulation in Estonia

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Declaration:

Hereby I declare that this doctoral thesis, my original investigation and achievement, submitted for the doctoral degree at Tallinn University of Technology has not been submitted for any academic degree.

/Sirje Pädam/

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MAJANDUS H26

Keskkonnapoliitika majanduslik aspekt: keskkonnareguleerimise kulud ja tulud Eestis

SIRJE PÄDAM

TTÜ KIRJASTUS

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INTRODUCTION

Environmental issues are an important field of economic analysis because missing markets and distorted prices fail to guide human decisions in a way that are desirable for society and long term sustainable development. Many problems with deteriorating environmental quality are unintended because they arise as side-effects of human activity. Since markets do not account for these side-effects, either because of missing property rights or because side-effects originate from diffuse sources, there is a need for collective action, i.e. policies to control the externalities. The other dilemma is that even when there is demand for improvements in environmental quality, markets will not provide this sufficiently. This happens because quality improvements are public goods, and these can be enjoyed, often for free, by a collective of individuals. The challenge for policy makers is to determine the demand for environmental improvements and to finance those desirable.

Although Estonia implemented environmental legislation and environmental charges soon after restoration of independence and introduced further regulations when becoming a member of the European Union, big challenges remain in the field of environmental policies, including the urgency to restructure Estonia's energy sector. Another is the limited use of economic analysis for guiding policy decisions. Effective policy making also includes consideration about how the general public value environmental improvements. The lack of studies in this field is an important reason for including research papers on environmental valuation in this thesis.

The aim of this doctoral thesis is to study recent environmental policies in Estonia from an economic perspective. This includes exploration of whether the allocation of EU funds towards the environment is sustainable, whether the use of supports to stimulate renewable energy production is efficient and how large the costs are to assure drinking water quality by the closure of a recreation area.

The thesis is made up of six independent research papers which are connected by a common theme, which is the **costs and benefits of environmental policies**. The research papers represent different case studies mainly based on Estonia and on a general level they focus on **three aspects** of environmental costs and benefits. The first aspect relates to the allocation of **environmental expenditure in the public budget**. The second aspect is about the economic efficiency of using **feed-in tariffs (FIT)** as a means to reduce the environmental impact of the energy sector. The third aspect is devoted to the **valuation of non-market goods** via stated preference methods.

The first two research papers examine whether **public environmental expenditure** is allocated in an environmentally sustainable way and how the financial crisis in 2008 affected environmental expenditures of the state and local government budgets. The purpose is to find out whether the goal of environmental sustainability is mirrored by actual public investment plans and to examine how an external shock affects public spending on the environment.

Taking the allocation of public environmental expenditure as given, the first two papers neither make an attempt to distinguish between externalities and public goods, nor explain why governments arrive at a specific budget allocation. Instead the first paper is devoted to an assessment of whether the allocation plans of EU cohesion funds of Estonia, Latvia and Lithuania during the budget period 2007–2013 are environmentally sustainable. The reason for choosing the three Baltic States is that they have comparable economic prerequisites, the countries are of similar size and subject to the same regulations concerning cohesion funding. The main input is made up of the ratified funding plans of each country. By using economic theory of sustainable development, the research paper suggests a two-step assessment procedure of environmental sustainability. The first step is based on the cost-benefit criterion and the second step uses ecological information as an additional constraint in order to detect potential issues of uncertainty and irreversibility not covered by the cost-benefit rule. In order to identify critical issues of sustainability, the second step uses the environmental performance index (EPI), which is an ecological indicator.

The second research paper explores how the Estonian public environmental expenditures were affected by the financial crisis in 2008. The hypothesis is that environmental expenditure is highly sensitive to budget cuts that were necessitated by the crisis. With its successful record of keeping government budgets under control, Estonia is likely to be more inclined than other countries to cut environmental expenditure and, therefore, we expect to find a significant influence from the crisis. We use data from Statistics Estonia in order to follow the developments during the time period 1995–2008 and preliminary budget data from the Ministry of Finance covering 2009.

The second aspect of the theme of the thesis concerns the use of **feed-in tariffs (FIT)** as a support scheme to promote production of renewable energy. This scheme comprises a guaranteed price to producers of renewable electricity and a purchase obligation of network operators. As the environmental pressure from the Estonian energy sector is significant, with about 90 per cent of electricity generated from oil shale, Estonia faces great challenges to meet the environmental requirements established by the EU and its national legislation. In order to promote the development of renewables in its energy sector, Estonia introduced FITs in May 2007. This decision can be understood as a policy that aims at reducing externalities from existing production of electricity by subsidising an environment friendly alternative. The purpose is to find out whether Estonia's FITs are effective, cost-effective and efficient, and what their implication is on income distribution.

The research paper makes an assessment that covers the first three and half years of Estonia's FITs. In addition to the use of sector-wide data for assessing effectiveness, efficiency and income distribution, the research includes a case study of the cost-effectiveness of two combined heat and power (CHP) plants. For the purpose of the case study, the analysis uses financial data from the annual reports of the CHP plants.

The third aspect of the theme of the thesis is **valuation of non-market goods.** Valuation in the fourth, fifth and sixth research paper is done with stated preferences methods, which relates to the measurement of quality changes according to how people themselves express the benefits they receive from an improvement or the disbenefit they experience from a loss in environmental quality. The survey of the fourth research paper is not a genuine preference survey as it asks for people’s attitude to make financial contributions and to accept cuts in their living standard for protecting the environment. The aim of the fourth paper is to find the main determinants of cross-country differences in public support to environmental protection. The research paper applies basic statistical analysis in order to make cross-country comparisons and for finding explanations to international differences in attitudes to environmental protection.

The fifth and the sixth research papers are based on surveys which apply the contingent valuation method (CVM). The aim of the fifth research paper is to find the demand for improving animal wellbeing at the Tallinn Zoological Gardens. Since animal wellbeing is just one aspect of the services that zoos provide, information about revealed preferences was not found to be sufficient for the purpose of the paper. The survey covered both zoo visitors and non-visitors in order to distinguish between use value and non-use value of improvements in animal wellbeing. The sixth research paper sets out to determine the cost of assuring drinking water quality by closure of a recreation area in the vicinity of the city of Tallinn. Since the area surrounding Lake Ülemiste is closed to public access, revealed preferences were not found to be possible for determining the value of the area in its alternative use. In the sixth paper the result of the survey is compared to the costs of additional investments in water purification technology. The two contingent valuation surveys apply open-ended willingness to pay questions and use regression analysis for interpreting and generalising survey results.

Table 1 below provides an overview of the connections between the aspects and papers, and the connecting features in terms of the analytical approach.

Table 1 Aspects of research papers and analytical approach

Aspect	Research paper	Cost benefit analysis (CBA)	Economic valuation	Non-economic assessment
Public environ-mental expenditure	1	X	CVM	EPI
	2			
FIT	3	X	Damage cost	
Non-market valuation	4			Attitude
	5	X	CVM	
	6	X	CVM	

Although the research papers cover several issues, cost-benefit analysis (CBA) connects the different aspects of the main theme of the thesis as CBA is used in research papers 1, 3, 5 and 6. Apart from the surveys reported in research papers

4, 5 and 6, environmental valuation originating from other sources is used as an input in research papers 1 and 3. Research paper 1 bases its assessment on environmental valuation on Baltic CVM surveys. In research paper 3, the value of the externalities is assessed by application of damage cost estimates. Non-economic assessment methods are used as a complementary input in research paper 1 and as the main input in research paper 4.

At the methodological level the empirical sources differ between the aspects of the theme of the thesis. The first aspect uses as its main source the ratified funding plans of EU cohesion funds and data about government expenditure. Energy sector statistics and the financial reports of two combined heat and power (CHP) plants are inputs of the second aspect. Surveys were carried out for the purpose of empirical data collection for the third aspect of the thesis.

The contribution of this doctoral thesis in theoretical and practical terms lies in the following.

1. The first paper assesses budget allocation plans of the EU cohesion policy funds for Estonia, Latvia and Lithuania, i.e. three countries with similar prerequisites. Analysing three countries of similar size and comparable level of economic development adds evidence for making generalisations. Previously the issue of fund allocation to the environment has only to a limited degree been studied as an academic topic. This paper takes one step further in terms of analysis as the focus is not only on economic efficiency, but primarily on environmental sustainability.
2. There is very little academic research devoted to the impacts of financial crises on public environmental expenditure. Based on the developments during the recent economic crisis, a case study is made for Estonia – a country with a record of stringent budget discipline. Additionally this paper discusses possible explanations to the observed developments.
3. According to the author's knowledge, no assessments have so far been carried out with respect to FITs granted to combined heat and power (CHP) plants – at least no relevant public information is available. In addition to general assessment criteria, the effects of FITs are studied on the basis of financial indicators of two CHP plants.
4. Conducting a survey in Estonia that included a selection of questions from the International Social Survey Program (ISSP) made for the first time possible a cross national comparison of Estonian attitudes to making financial sacrifices or accepting cuts to one's living standards for reasons of environmental protection.
5. Inferences of animal wellbeing are generally based on animal behaviour. The fifth research paper takes another perspective. The hypothesis is that humans have preferences for animal wellbeing and that these preferences can be measured by revealed or stated preferences. The reason for choosing a contingent valuation survey, rather revealed preferences, was to test whether human valuation of animal wellbeing also includes non-use value. To the author's knowledge non-use value of zoo animals has previously not been studied on academic level.

6. Improvement of environmental quality of city dwellers includes access to recreation areas. Lake Ülemiste and its surrounding are currently closed to public access. By its vicinity to the city of Tallinn, Lake Ülemiste provides a unique case study of an environmental quality improvement.

The **following articles** have been **published** in the course of research. All articles are co-authored.

Pädam, S., Ehrlich, Ü. and **Tenno, K.** 2010. The Impact of EU Cohesion Policy on Environmental Sector Sustainability in the Baltic States. Baltic Journal of Economics, 10 (1), pp. 23–41.

Ehrlich, Ü. and **Pädam, S.** 2010. Public Environmental Expenditure in Time of Crises in Estonia. In: Discussions on Estonian Economic Policy 18: Eesti majanduspoliitilised väitlused 18: Estnische Gespräche über Wirtschaftspolitik 18, Mäeltsmees, S. and Reiljan, J. (Editors). Berlin: Berliner Wissenchafts-Verlag, Mattimar, pp. 38−51.

Kleesmaa, J., Pädam, S. and **Ehrlich, Ü.** 2011. Subsidising Renewable Electricity in Estonia. In: Energy and Sustainability III: Villacampa Esteve, Y. Mammoli, A. A. and Brebbia, C. A. (Editors). Southampton, UK, WIT Press. WIT Transactions on Ecology and the Environment; 143, pp. 229–240.

Pädam, S. and **Ehrlich, Ü.,** 2011. Paying for Environmental Protection in Estonia in International Comparison. The Economy and Economics after Crisis. Sepp, J. and Frear D. (Editors) Berlin: Berliner Wissenchafts-Verlag, pp. 197 −209.

Pädam, S. and **Ehrlich, Ü.,** [forthcoming]. The Willingness to Pay for Improving Animal Wellbeing at Tallinn Zoological Gardens. The World Economy: Contemporary Challenges, Hejduk, I.K. and Grudzewski, W.M. (Editors) Difin, Warsaw.

Pädam, S. and **Ehrlich, Ü.** 2011. The Foregone Recreation Value of Lake Ülemiste. In: Discussions on Estonian Economic Policy 19: Eesti majanduspoliitilised väitlused 19: Estnische Gespräche über Wirtschaftspolitik 19, Mäeltsmees, S. and Reiljan, J. (Editors). Berlin: Berliner Wissenchafts-Verlag, Mattimar, pp. 135 −148.

Author's contribution

Paper 1. Based on theoretical literature in the economics of sustainable development the author of this thesis designed an assessment method of sustainability and carried out the assessment of funding plans of Estonia, Latvia and Lithuania. The author took active part in the final evaluation and was responsible for writing the manuscript of the paper.

Paper 2. The author of this thesis organised data collection and carried out the assessment of the impacts of the crisis on budget expenditures and participated in the preparation of the overview of recent Estonian environmental policies.

Paper 3. Based on scientific literature the author of this thesis defined the framework of the economic assessment and made the calculations concerning externalities. The author participated in the compilation of Estonian materials concerning legislation, financial data and emissions and in the overall assessment.

Paper 4. The author of this thesis took active part in the development of the survey questionnaire which was designed to evaluate public attitudes towards protected areas at the prospect of the celebration of the 100th anniversary of nature protection in Estonia in 2010. The author of this thesis prepared a framework for the international comparison, made basic statistical comparisons and participated in the cross-country comparative analysis.

Paper 5. The author of this thesis was involved in the preparation phase of the survey. The author carried out a literary survey, searched information about zoo finances and practices of zoo animal welfare and took active part in the analysis of the survey results.

Paper 6. The author of this thesis took part in the planning phase of the survey. Furthermore, the author collected information on water purification practices and costs in neighbouring countries, carried out a cost-benefit analysis and took active part in the analysis of survey results.

Overview of the approval of research results

1. An early version of the assessment of EU Cohesion Fund allocation to the environment was presented by the author at the Conference of the International Society for the Study of European Ideas (ISSEI) at Helsinki University in July 2008.
2. Results of the research about the impacts of the financial crisis on public environmental expenditure were presented by the author at the Scientific Conference on Economic Policy at Värska in June 2010.
3. The assessment of FITs on Estonian electricity production and the case study of CHP plants were presented by co-author Jüri Kleesmaa at the scientific conference “Energy and Sustainability 2011” in Alicante, Spain, in April 2011.
4. The international survey results in comparison to the Estonian survey were presented by the author: First at the international scientific conference “Nature Conservation Beyond 2010”, in Tallinn in May 2010 and then at the Congress of Political Economists conference in Honolulu in July 2010.
5. The paper on animal wellbeing was presented by the author at the conference of the Congress of Political Economists in Singapore in July 2011.
6. The results of the contingent valuation survey of the foregone recreation value of Lake Ülemiste were presented by the author at the Scientific Conference on Economic Policy at Värska in July 2011.

Abbreviations

CBA – Cost-Benefit Analysis
CEE – Central and Eastern Europe
CHP – Combined Heat and Power, Cogeneration of Heat and Power
CO_2 – Carbon Dioxide
CV – Contingent Valuation
CVM – Contingent Valuation Method
EEK – Eesti kroon, Currency of Estonia 1992-2010
EIC – Environmental Investment Centre
EKC – Environmental Kuznets Curve
EL – Euroopa Liit
EPI – Environmental Performance Index
EU – European Union
EU-ETS – European Union Emission Trading System
FIT – Feed in Tariff
GDP – Gross Domestic Product
GWh, MWh, kWh – energy content
ISSP – International Social Survey Program
MW – Mega Watt, Capacity
NGO – Non-Governmental Organization
NOAA –The National Oceanic and Atmospheric Administration
NO_x – Nitrogen Oxide
OLS – Ordinary Least Squares Regression
PM_{10} – Particulate matter
PPP – Purchasing Power Parity
PV – Photovoltaics
R&D – Research and Development
SMS – Safe Minimum Standard
SO_2 – Sulphur Dioxide
TSP – Total Suspended Particulates
WACC – Weighted Average Cost of Capital
WTA – Willingness to Accept
WTP – Willingness to Pay

Acknowledgements

This thesis marks the end of a journey that started almost 20 years ago. During many years I believed that the journey had ended with my decision in the mid-1990's *not* to write a doctoral thesis. Had it not been for the encouragement and support from people around me, I might have been right. Looking back, I can see how many people surrounding me have supported me in various ways and the words of one of my friends has taught me something essential: I have received more than I ever can give back.

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1. THEORY AND LITERARY OVERVIEW

The theory of resource allocation, also known as welfare economics provides the foundation of the analyses. In the basic model, prices reflect scarcity of resources and guide economic agents who act in their self-interest to allocate resources efficiently. This observation dates back to Adam Smith and his conclusion about the role of the invisible hand (Smith 1776). But it was not until the late 19th century and the discovery of marginal analysis, diminishing marginal productivity and consumer preferences in terms of utility that paved way for the development of welfare economics finally systematized in the 1950s. Arrow (1951) and Debreu (1959) derived the fundamental theorems of welfare economics. The first fundamental theorem says that all perfectly competitive equilibriums with complete markets are Pareto efficient and can be interpreted as a mathematical restatement of Adam Smith's invisible hand. The second fundamental theorem says that any Pareto efficient allocation can be attained via the price system by suitable choice of lump-sum transfers. The implication of the second theorem is that issues of efficiency and income distribution can be handled separately.

The strength of welfare economics lies in its systematic framework of evaluating the economic implications of alternative resource allocations. For this purpose the Pareto criterion (Pareto 1897) provides welfare implications of changes in resource allocation. Any change in the initial resource allocation that makes at least one individual better off without making any other individual worse off is defined as a Pareto improvement. An allocation of resources is efficient when no further Pareto improvements can be made. What the first fundamental theorem of welfare economics implies is that under certain conditions market economies have the potential to achieve Pareto efficient resource allocation and this occurs because economic agents will make agreements only if this improves the outcome to at least one of the parties without making the other worse off.

However, as a rule for practical policy guidance, the Pareto criterion is restrictive. A policy that makes one group of people better off but at the same time makes another group of people worse off, would require transfer payments to compensate those who lose. Assuming that the gain is larger than the loss, implies that a Pareto improvement cannot be attained until compensation has been made. In the late 1930s two British economists suggested a simplified criterion. Kaldor suggested that those who gain should be able to compensate the losers and still be better off (Kaldor 1939). The proposition of Hicks was that the losers should not be able to profitably bribe the gainers not to change (Hicks 1939). Their contribution is known as the Kaldor-Hicks criterion and, it states that a change enhances efficiency if gainers could in theory compensate the losers and still be better off. By being less stringent, the Kaldor-Hicks criterion is applicable to more situations than the Pareto criterion. The cost-benefit criterion rests on the Kaldor-Hicks criterion, and it states that if the benefits of a project are larger than the costs, the project is socially preferable, and if there are several alternative projects the one with the greatest net benefit should be chosen (Gramlich 1981).

1.1. Analysis of environmental problems

Environmental problems have special implications. Since resource scarcity is not reflected by prices, incentives will be distorted and markets will not achieve an efficient resource allocation. As a result, unregulated markets pay too little attention to environmental quality, implying that regulation can lead to potential Pareto improvements. The most important market failures related to the environment occur when negative externalities[1] are present and when there is demand for public goods. Pigou was the first one to analyse pollution externalities and to suggest a tax to internalise the externality (Pigou 1920). An important step was taken in the analysis of public goods by Samuelson's observation that some goods are jointly consumed in such a way that one individual's consumption "leads to no subtraction from any other individual's consumption of that good" (Samuelson 1954, p. 387 and Samuelson 1955). Samuelson divided private and public goods through a one-dimensional classification. In the 1960s several scholars contributed and added a second dimension to Samuelson's classification. Today it is generally held that private goods are excludable and rival, while pure public goods are defined by two basic characteristics: nonexcludability and nonrivalry of consumption (Buchanan 1968, Stigliz 2000). So called quasi-public goods lack one of the two characteristics. If it is possible to exclude consumers, by collecting fees for the right to consume the public good, these goods become club goods. This would be the case if a unique forest area is fenced and visitors can enter only after having paid a fee. Improvement in air quality is a pure public good since no one can be excluded from consumption (nonexcludability) and there will not be less air quality left to other consumers (nonrivalry). Common resources including ocean fishery and common lands are non-excludable, but rival. The distorted incentives of common use of rival public goods have been vividly illustrated by Garret Hardin stating that "the utility of a herdsman who adds one animal is individual since he receives all proceeds from the sales" while "the negative component of the additional overgrazing is shared by all herdsmen". …"Therein is the tragedy. Each man is locked into a system that compels him to increase his herd without limit – in a world that is limited" (Hardin 1968).

The efficiency criterion is useful for explaining environmental problems and for finding effective means for regulation, but is efficiency compatible with sustainability? Assuming that sustainability is a synonym to sustainable development we can apply the most widely quoted definition of the Brundtland report (World Commission 1987) i.e. "Sustainable development is development

[1] An externality arises when someone's production or consumption decision affects the utility or production of another economic agent without the responsible party considering the impact of the externality on other economic agents.

that meets the needs of the present without compromising the ability of future generations to meet their own needs" (ibid. p.43). Hence, there seems to be a connection between efficiency and sustainability – efficiency implies a resource allocation that considers people's preferences and sustainability presumes that people meet their needs. In addition, efficiency requires cost-effective use of resources and this implies non-wastefulness. However, efficiency is a non-dynamic concept. Without introduction of a time dimension, efficiency does not cover future generations. In order to move efficiency in the direction of sustainability, there is a need to introduce time perspective and to allow for intertemporal reallocation of resources so that human needs can be met at different points in time.

1.2. Public budget and the environment

The theory of the public budget is multidisciplinary, partly political, partly economic, partly accounting and partly administrative (Khan and Hidlreth 2002). One stream of research provides a budget theory resting on welfare economics and allocation of scarce resources. Hackbart and Ramsey (2002) present an outline of a budget theory based welfare economics. They connect public budgeting to the three functions of the government: allocation function, distribution function and stabilisation function. The allocation function of the budget refers to the case when the public sector allocates resources through the political process in areas where markets fail. Environmental expenditure can, therefore, be regarded as a part of the allocation function and according to welfare economic budgeting, as well as welfare economics in general, this is the reason why environmental funding generally is the responsibility of the public sector (ibid).

In addition to government functions, there is the theory of fiscal federalism, which deals with the level of government most suited for a certain type of expenditure. Optimal allocation assigns the responsibility to the territorial authority where beneficiaries correspond to that of taxpayers (Hackbart and Ramsey 2002, Pitlik 2007). If the benefits of public goods spill over to neighbouring territory there is reason to centralize responsibility, and similarly when disbenefits from environmental pollution cross administrative borders. Fiscal federalism would thus predict that EU cohesion funding to the environmental sector is devoted to environmental issues of cross border characteristics.

Due to the fact that environmental protection to a large degree is a public sector responsibility, allocation and effective management of public funding are key to achieving successful environmental policy making. However, Vincent and his co-authors (2002) note that despite strong reasons for analysing public expenditure and the environment, there is only limited academic literature within this field. Apart from two recently conducted analyses (Wang 2011, Lopez et al. 2011), Vincent's observation still seems to hold ten years later. Wang analyses local level environmental spending in 66 Florida counties during the time period 1999–2008. Lopez and his co-authors define a macro level model for studying fiscal spending and its impact on pollution of 30 countries.

Although EU budgeting has been studied from various angles, including reframing (Neheider and Santos 2011), factors affecting bargaining outcomes (Akzoy 2010), macro-economic effects of EU budgets (Asdrubali and Kim 2008) and EU responsibilities from the viewpoint of fiscal federalism (Pitlik 2007), there is hardly no literature related to EU budgets and the environment apart from reports published by the Czech Republic based NGO on EU funding and the environment (CEE Bankwatch Network 2007).

Pitlik (2007) who analyses the EU budget allocation 2007–2013, compares the various budget items to the theory of fiscal federalism. He finds that only 10 per cent of total budget resources can be attributed to tasks for which there is a shared EU responsibility and concludes that the existing budget structure rather can be explained by redistribution through agricultural and structural policy to compensate losers in the integration process. Vincent and his co-authors (2002) study the sector wise allocation of budget expenditures and carry out a trend analysis in relation to GDP-development. They find that during the Asian crisis public sector spending to the environment was more sensitive to the GDP development than other public funding. Wang (2011) compares the structure of local government budgets over time and finds that counties with higher level of manufacturing and agriculture spend more to protect the environment (e.g. pollution control, protection of natural areas) and counties with higher population densities spend more on services that utilize the environment (e.g. water supply, waste disposal). Another finding is that counties that devote more spending on public safety and economic development spend less on the environment. Based on these results Wang (2011) suggests that environmental spending is a result of combined forces of environmental pressure and budgetary politics.

The budget allocation of the first research paper regards the environmental sustainability of planned investment funds. Literature offers a wide variety of evaluation methods of environmental sustainability (Pädam 2003). A welfare theoretical notion is suggested by Stavins and his co-authors (2003). According to their interpretation, sustainability can be understood as dynamic efficiency along a feasible consumption path. In addition, sustainability entails non-wastefulness, implying that the choice of a consumption path is such that the economy is on the Pareto frontier where sustainability represents a non-declining path of welfare evaluated in an infinite time horizon. For the purpose of sustainability assessment Stavins et al. propose a decision rule similar to the Kaldor-Hicks criterion, i.e. that those who are made better off by a policy in theory can fully compensate those who are made worse off (ibid). A policy that fails the intertemporal Kaldor-Hicks test cannot pass the stricter Pareto test. However elegant their formulation, there is one serious shortcoming since natural capital and ecosystems are assumed to be convex sets in Stavin's model. By this they rule out irreversibility of environmental change and the possible events of an ecological collapse[2]. Dasgupta and Mäler (2004) point out that convexity is absent in ecology and for this reason competitive price equilibriums à la Stavins might not exist. This is further developed by Arrow and his co-authors (2004), who conclude that imperfect economies might be incapable of sustaining welfare over time due to scarcity of resources and limited substitution possibilities among capital assets. Despite these complications they show that the general cost-benefit rule holds for guiding sustainable investment decisions in an imperfect economy under the condition that proper accounting prices can be determined, and these can to a large degree differ from market prices (ibid). Non-convexities place significant challenges on the determination of proper accounting prices because such prices should increase rapidly when the use of a resource approaches a threshold level.

The Safe Minimum Standard (SMS) has been proposed as a policy rule in situations characterised by uncertainty and potential irreversibility, i.e. when non-convexities might become binding constraints. The SMS is not a price, but rather a quantity restriction. Randall and Farmer discuss the interrelation between cost-benefit analysis and SMS as decision rules for conservation policies (Randall and Farmer 1995). They suggest that when there is uncertainty and potential irreversibility, SMS should be applied as an additional constraint on the cost-benefit analysis. The cost of applying the SMS manifests itself in terms of a lower level of human consumption of natural resources. In order to make their suggestion applicable they recommend some kind of early warning system for policy decisions (ibid. p. 42).

During economic crises, governments have two general policy choices in relation to the public budget. One is to run a counter cyclic budgetary policy in order to stimulate domestic demand and the other one is to assure budget balance in order to avoid budget deficits that can become a threat to financial stability. It is of course possible that governments attempt to combine these approaches. This could be the case if governments have a chance to reallocate expenditure from fields with small multipliers into activities that have large multipliers by e.g. postponing procurements from abroad and using these funds for e.g. investments in domestic construction projects. Despite government efforts, certain fields of expenditure might experience larger cuts than others during an economic downturn. Since fiscal policies belong to the stabilisation function of the public budget and not targeted at correcting market failure there is a risk of significant cut-downs in allocative expenditures, including environmental spending. Observations during the Asian financial crisis in 1998 suggest that public environmental expenditure is more sensitive to crises than other public expenditure (Vincent 2002). During the Asian crisis as well as during the recent

[2] By assuming ecosystems are convex sets excludes the event of an ecological collapse, which refers to a catastrophic decline in the carrying capacity of an ecosystem, often resulting in mass extinction. Past experience shows that when anthropogenic pressure reaches a certain threshold level, marginal changes may result in irreversible effects. An example is the Atlantic cod fishery that collapsed in the 1990s due to overfishing.

financial crisis, the slowdown has been characterized by severe strains on financial markets. Bowen and Stern provide several arguments to why additional environmental policies should be introduced during economic downturns (Bowen and Stern 2010). Recessions induced by a fall in private demand characterized by involuntary underutilized resources will lower the opportunity cost of some environmental projects, without affecting their benefits significantly. This is further confirmed by references to evaluations, which have found that many environmental projects pass the effectiveness criteria of being appropriate for combating economic slowdown: i.e. they have an impact on domestic output and a significant employment multiplier (see Bowen and Stern 2010 for references). Although timeliness of implementing new environmental projects might be a problem, this feature affects most fiscal measures. The authors notice that timeliness is a proper argument only when comparing fiscal and monetary policy. Investments in environmental protection that mitigate unemployment and increase domestic demand seem thus to have similar positive impacts as traditional fiscal policy measures such as construction projects. In addition, environmental investments are important for sustaining development in the long run.

1.3. Feed-in tariffs to stimulate renewable energy

Feed-in tariffs (FIT) is the most widely used support scheme for renewable electricity: implemented in 23 EU countries and 60 countries worldwide in 2011 (Gipe 2011). Denmark and Germany were the pioneers to introduce FITs in late 1980s and early 1990s respectively. Success stories about countries that have implemented FITs and exceeded initially set goals seem to have spurred developments. The motives for FITs vary − one reason is that production costs of renewable electricity typically are higher than those of non-renewable − another is promotion of new technology. The EU has set a target to increase the share of renewable energy[3] to 20 per cent of energy consumption by the year 2020 (EC 2009). EU countries including Estonia, therefore, look for efficient ways to stimulate renewable energy capacity.

FITs entail a guaranteed price for those undertakings that produce electricity from renewable sources whereas the network operator is obliged to purchase their production (del Rio and Gual 2007). There are two possibilities for covering the costs of FITs, either on the account of consumer's electricity bill or through the public sector budget. Denmark covered its feed-in tariffs via the public budget. However, the rapid increase in wind power capacity came at a high cost to the government − the tax refunds and output subsidies surpassed Euro 75 million in 1998 (Sijm 2002).

In an assessment of FITs Sijm (2002) studies the German, Danish and Spanish experiences according to five evaluation criteria, which are based on

[3] Renewable energy embraces water, wind, solar energy, waves, tides, geothermal heat, landfill gas, biogas and energy from biomass.

welfare economics. The criteria are: **Investment Certainty**, **Effectiveness**, **Efficiency**, **Market Compatibility and Competition** and **Administrative Demands**. By investment certainty, Sijm denotes the time the producer receives a guaranteed price. Effectiveness signifies the extent and speed of renewable capacity increase. Efficiency covers both static efficiency i.e. that electricity is generated and sold at minimum cost and dynamic efficiency in terms of promoting innovations and therefore reducing costs. Market compatibility examines whether there are distortionary impacts of FITs on competition. Administrative demands relate to the complexity of the system. del Rio and Gual (2007) carry out an evaluation of the Spanish FITs using the first three criteria. Their evaluation goes more into detail concerning different types of renewable energy sources and they also carry out cost-benefit analyses of different renewables: wind, solar, small hydro and biomass.

Both evaluations find that FITs have been effective in supporting the development of wind energy capacity, but not equally successful concerning other renewable energy sources. The schemes are assessed as successful in reducing investor uncertainty. Since tariffs may not be compatible with a liberalised, competitive market and a system of harmonised renewable energy policies within the EU, Sijm (2002) points out that the goal of investor certainty should rather be set to reflect short to medium term perspective. Static efficiency is not found by Sijm (2002). Similar results are provided by del Rio and Gual (2007) whose cost-benefit analyses show that the net social benefits are negative for all renewable energy sources if total energy production costs are compared to social benefits. When comparing social benefits only to generation costs, wind and small hydro pass the cost-benefit criterion while photovoltaics (PV), primary and secondary biomass do not. In addition they find that although consumer costs were relatively low, increasing from eurocents 0.14 to 0.26 per kilowatt-hour between 1999 and 2003, the costs are high compared to avoided externalities. The authors agree though that there may be arguments of dynamic efficiency for supporting wind and PV because FIT will increase R&D investments and lead to learning effects from increased diffusion. del Rio and Gual find scattered evidence in support of dynamic efficiency. In his assessment, Sijm (2002) concludes that FITs are effective in promoting electricity generation from renewable sources, but costly, inefficient and distortive.

Frondel et al. (2010) use slightly different assessment criteria in their impact evaluation of the amended German system of FITs introduced in 2000. In their overview of developments they note that between 2000 and 2008 wind energy increased its share in Germany's electricity production from 1.3 to 6.3 per cent and biomass from 0.4 to 3.6 per cent. Photovoltaics (PV) grew from almost nothing to 0.6 per cent in 2008. Irrespective of its small share of energy production PV subsidies comprise about 25 per cent of feed-in tariff expenditures (Frondel 2010). The authors divide their evaluation into impacts to **Costs** and **Benefits.** On the cost side they calculate the net present costs of commitments from wind and PV installed in Germany during the time period

2000–2009, they estimate the per kWh cost and calculate the abatement cost of one tonne of carbon dioxide (CO_2), see Table 2.

Table 2 Feed-in tariff expenditure and the per kWh cost of the German Renewable Energy Resources Act, 2001–2008 Sources: Table 2 in Frondel et al (2010) and AGEB (2011)

	2001	2002	2003	2004	2005	2006	2007	2008
FIT expenditures in € billions	1.58	2.23	2.61	3.61	4.40	5.61	7.59	9.02
Total electricity cons. Germany, billion kWh	585.1	587.4	598.6	608.0	612.1	617.2	618.1	614.6
Cost eurocents per kWh	0.3	0.4	0.4	0.6	0.7	0.9	1.2	1.5

The costs are then compared to climate impacts and employment effects. They find that the carbon abatement cost of PV is Euro 716 per tonne[4] and that of wind is Euro 54 per tonne CO_2, which in comparison to emission allowance prices on EU-ETS (European Emissions Trading System) are very high.[5] However, they note that the climate impact is zero due to the coexistence of FITs and the EU-ETS. The same conclusion was also made by e.g. Sijm (2005) who noted that "once the EU ETS becomes operational, the effectiveness of all other policies to reduce CO_2 emissions of the participating sectors becomes zero". This happens because EU-ETS sets a binding carbon emissions cap. When CO_2 emissions are reduced by substitution of fossil fuels with renewable energy, the producer of fossil electricity can sell its emission allowances to another sector, which then can increase emissions (Frondel et al. 2010). Employment effects are assessed small, and possibly non-existent, since if more workers produce the same output, the employment effect is counterproductive to net job creation. Neither do they find any significant benefits concerning the development of new technology.

There are other FIT evaluations, an overview is e.g. provided by del Rio and Gual (2007) and it seems as existing assessments have concentrated on wind power and PV, which is understandable due to their dominance in the share of capacity and costs among renewable energy sources. For this reason, recent developments in Estonia provide a slightly different picture. Estonia introduced FITs in May 2007 and has besides an increase in wind energy capacity seen a very rapid growth in renewable electricity production from biomass, see Figure 1.

[4]They assume that PV is replaced by a mixture of natural gas and hard coal with an emission factor of 0.584 kg CO_2 per kWh. The same emission factor of replaced energy is used for wind.

[5]They point out that the price of emissioon allowances have never exceeded Euro 30 per tonne CO_2.

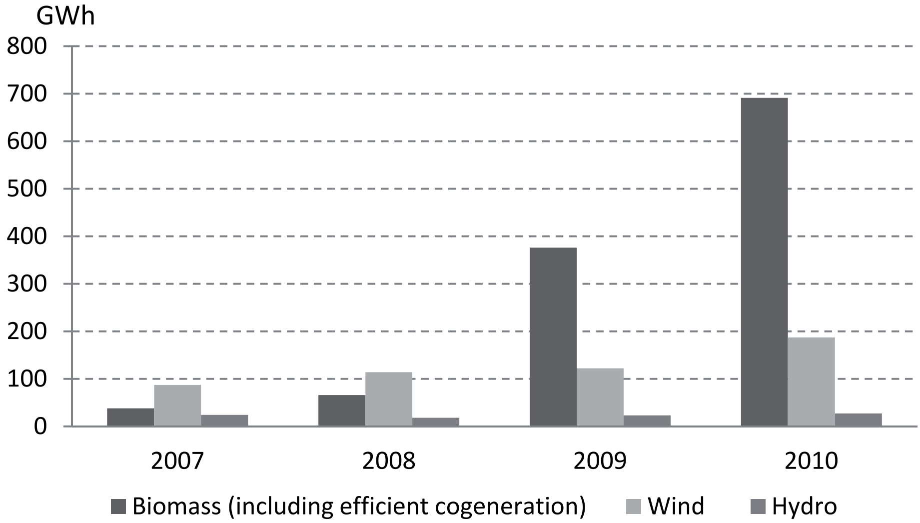

Figure 1. Production of electricity from renewable sources in Estonia 2007–2010. Sources: Development plan (2009) 2007 and Arus (2011) 2008–2010.

Between 2007 and 2010, biomass increased its share in Estonia's electricity production from 0.3 to about 8.2 per cent and wind from 0.7 to 2.2 per cent.[6] The fast increase in biomass relates to a rapid development of combined heat and power plant capacity. In order to fill the gap in FIT-evaluation literature, the assessment of Estonia's FITs, which is reported in research paper 3 covers a case study of two CHP plants that started operations in 2009.

1.4. The value of environmental improvements

Irrespective of what kind of consumption choices individuals make, consumption affects utility, and the impact on utility represents some value to the individual. When individuals make a choice, either in relation to what to buy or how to spend their time, they appraise the value they will receive from a particular choice. Many goods are not subject to market transactions and they can be enjoyed for free, e.g. bird watching and swimming in a lake. Through human choices the value of these activities can be assessed. In his seminal paper, Krutilla (1967) went even further by suggesting that people receive utility from natural assets just because they exist. Thus, utility may originate from the pure knowledge of conservation of a species of a certain wilderness area. Figure 2 below shows an overview of different values of the environment.

[6] Note that the development plan (2009) is the source of shares in 2007. Production by renewable source is based on Arus (2011) and divided by gross final consumption which is found as gross electricity production minus exports and internal use by power plants plus imports according to electricity balance table FE03 statistics Estonia.

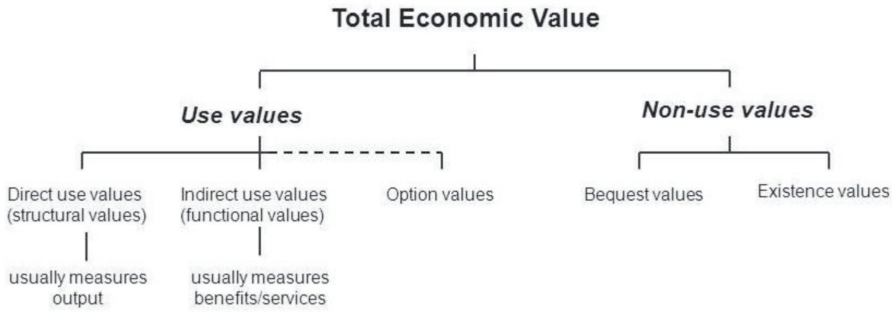

Figure 2. Total economic value. Source: based on Barbier (2000)

Stemming from Krutilla's work, it is now widely held that the value of the environment can be divided into use value and non-use value. While use value refers to the value people put on using the environment directly, either for exploitation purposes (logging, housing development), consumption for recreational intentions (bathing, bird watching, camping), or indirectly via the use of ecosystem services (forests for erosion protection or as carbon sinks), non-use value relates to no use at all. It is commonly recognized that non-use values include existence value, i.e. benefits derived from knowing that a resource exists, and intrinsic value, which relates to the willingness to pay for maintenance of natural areas and biodiversity, independent of its usefulness to humans. While use values might be private or public goods, non-use values are pure public goods.

A third value category is usually added: option value, which presupposes a wish to preserve the environment for future use. Originally proposed by Weisbrod (1964) and later clarified by Lindsay (1969) to be the insurance premium, i.e. willingness to pay for opportunities of future use.

According to Johansson (1993), the changes in utility that will result from environmental policies, alter either the quantity or quality of a nonmarket environmental good. For the individual, the environmental good is available in fixed quantities. From this follows that compensating surplus (or the equivalent surplus) is the appropriate welfare measure. Since demand is conditional on the environmental good, q, this will enter the demand function. For an individual who maximizes her utility subject to the budget constraint, the indirect utility function, v, can be found by inserting the demand function into the utility function:

$$v = v(p, y, q) \tag{1}$$

where y is income, p is a vector of prices for market goods and q denotes environmental quality or provision of environmental commodities. The change in an individual's utility from an improvement of environmental quality from q^0 to q^1 can be defined as:

$$\Delta v = v(p, y, q^1) - v(p, y, q^0) \quad (2)$$

Rearranging, it is possible to define the change in welfare as the maximum amount of income that she is willing to give up in order to receive the higher environmental quality, q^1, while keeping her utility constant.[7] This, in turn, corresponds to a movement along the Hicksian demand curve.

$$v(p, q^0, y) = v(p, q^1, y - WTP) \quad (3)$$

The values of environmental improvements are public goods, not traded in private markets, making the determination of their demand a challenging exercise. Valuation methods are usually divided into two approaches: stated preference and revealed preferences (Freeman 2003, Haab and McConnell 2002). Stated preference methods seek to infer individuals' preferences of environmental quality directly, by asking them to state their willingness to pay for a change in environmental quality or quantity. In contingent valuation surveys, for example, this might consist of asking people either their maximum willingness to pay (WTP) for an increase in environmental quality, or their minimum willingness to accept compensation (WTA) to forgo such an increase. Respondents might instead be asked about their maximum WTP to avoid a decrease in environmental quality, or their minimum WTA to accept this reduction.

Revealed preference methods estimate the value by studying human behaviour in complementary markets, i.e. money spent on travelling to a natural park (travel cost method) or how the local environment affects housing prices in urban areas (hedonic model). Use values can be estimated by direct and indirect methods. However, since human behaviour is a prerequisite for the travel cost and hedonic approaches they cannot elicit non-use values. Non-use values can only be estimated by using the direct approach of stated preference methods (Barbier 2000, Smith 1993).

1.5. Contingent valuation methodology

The contingent valuation methodology uses surveys or experiments to find the monetary measure of welfare associated with a discrete change in the provision of an environmental good. In the field of environmental economics, the contingent valuation methodology (CVM) has become an important tool to determine the value of changes in environmental quality. In a recent overview of the history of contingent valuation, Carson (2011) provides references to more than 7,500 papers and studies worldwide. Still CV studies are very rare in

[7] If there is environmental deterioration, then the measure is the minimum amount of money that must be given to an individual to compensate for the loss in utility due to lower environmental quality, i.e. the willingness to accept (WTA).

Estonia. Prior to 2010 less than five studies had been carried out (Ehrlich and Habicht 2001, Oisalu and Strosser 2007, see also Annex III of Söderqvist and Hasselström 2008). The first empirical application of the contingent valuation method (CVM) was made by Davis (1963) in his study of hunters in Maine. But it was not until the mid-1970s that the method started to develop in earnest (Brookshire et al., 1976; Randall et al., 1974). Since then, the method has become the most widely used, but also a controversial technique of environmental valuation. Comprehensive accounts of the method are found in Mitchell and Carson (1989), Champ et al. (2003) and Alberini and Kahn (2009).

Several authors have expressed doubts about some aspects of the contingent valuation method. Some of them question data collection methods, as data do not originate from market transactions (Diamond and Hausman 1994). Another concern is that in the survey situation, respondents could reveal something else than their true WTP, either because of the hypothetical nature of questions, or because respondents answer strategically (Boyle 2003). For these reasons research has been conducted for the purpose of testing validity and reliability. Validity refers to the extent to which CVM measures the theoretical construct under investigation (Carson and Mitchell 1989). Reliability, on the other hand, refers to the replicability of the obtained results.

The research on reliability and validity suggests careful design as the core of the contingent valuation study (Mitchell and Carson 1989, Arrow et al. 1993, Smith 1993, Carson et al. 2001, Boyle 2003). Important for reliability are sample size and "the degree to which respondents find [the description of the questionnaire] credible and realistic" (Mitchell and Carson 1989). In Boyle's overview about research on the validity of the contingent valuation method, he notes that CV provides plausible estimates of use values when compared to cash transactions and indirect methods such as travel cost surveys, i.e. there is criterion and convergent validity. However, he points out that for non-use value, criterion validity has not been found (Boyle 2003, p. 153). The difficulty to compare estimates of non-use values to behavioural data and survey results has increased suspicion about non-use value. This is a reason to be cautious about estimates of non-use values. However, as pointed out by Carson et al. (2001) ignoring non-use value because of the difficulties to determine its validity may lead to overlooking significant welfare gains or losses.

Another discussion concerns the observation that CVM potentially gives rise to larger WTP measures than the respondents' true WTP. Boyle (2003, p. 155) notes that "(f)urther validity research needs to focus on design features that would reduce the overestimation bias." The divergence between real and hypothetical WTP statements, so called hypothetical bias, has been measured in two meta analyses of existing CV studies, which have compared real and hypothetical payments (List and Gallet 2001, Murphy et al. 2005). Both studies report about significant divergence also irrespective of the format of the question. However, as Loomis (2011) observes, the meta analyses have not considered survey quality. Another reason to hypothetical bias is that many tests have concerned private goods. Observations based on selling real deliverable market goods for cash may well result in WTP which is lower than respondents' true value. When respondents are buying perfect substitutes of goods they find for a known price in a retail store, they have no incentive to tell their maximum WTP (Loomis 2011, Harrison 2006, and Harrison et al. 2004).

There are relatively few studies that test the divergence between open-ended questions for public goods with and without payment. In a survey of red kite conservation in Wales, Christie (2007) finds that although the mean of hypothetical donations is larger than the mean of actual donations, there is equality of hypothetical and actual WTP among those who provide a positive bid. The explanation to the deviation between the hypothetical and real WTP amount may thus be that contingent valuation surveys overstate the intention to pay rather than the willingness to pay (ibid). Another survey that studied real and hypothetical donations for a public good (purchase of a remote Scottish island for nature conservation purposes) did not find any significant difference between real and hypothetical amounts (Macmillan et al. 1999).

Mitani and Flores (2009) report that recent results from laboratory experiments[8] using one-shot[9] binary choice questions, generally find no statistically significant hypothetical bias. In their own experiment where they apply one shot open-ended elicitation format they find no hypothetical bias between the real and hypothetical payment treatments.

Empirical issues

There is a large range of practical issues related to a CV study: creation of hypothetical markets, carrying out a survey and analysing the results (see e.g. Mitchell and Carson 1989, Arrow et al. 1993, Haab and McConnell 2002, Boyle 2003, Whitehead 2006). However, there is no standard approach how to design a contingent valuation survey. Nevertheless, virtually every application consists of several well-defined elements. The first is to identify what change the survey sets out to value, i.e. the difference between the baseline utility and the utility with the new environmental condition (Boyle 2003). This step serves as the basis for the scenario or description of the (hypothetical, i.e. contingent) policy or program the respondent is being asked to value or vote on. In order to follow welfare economics, the physical change needs to be described clearly in the survey to give the individual an opportunity to perceive how the survey scenario affects her utility (ibid).

For this reason information about the physical change of the policy is of great importance (Mitchell and Carson 1989, Carson et al. 2001, Boyle 2003). It has been shown empirically that WTP will be biased, unless people are told what they are being asked to value (Boyle 2003, p. 125).

The second step is to identify the survey population, whether to survey households or individual citizens and the geographical delimitation. In CV studies that include non-use value, the population should expand outside those

[8] Laboratory experiments denote experimental studies with few subjects where researchers induce values and let individuals bid under controlled forms.

[9] One shot implies that experiments do not apply bidding games.

who actually use the resource. However, Carson et al. (2001) point out that limitations are relevant in some settings because of the payment mechanism, if "state policy makers would only be interested in comparing the values of state residents to the state tax payments"….. "Thus, even if some residents of another state valued the park, the state providing the park can choose not to 'care' about their values. In this case, the population that should be surveyed is that of the state which is considering providing the park." (p. 180). Having defined the survey population and the geographic scope the third step is to choose the sample size. Mitchell and Carson (1989) note that CV surveys require large sample sizes because of the large variance in WTP. Based on earlier CV surveys they determine that sample sizes should be in the range of 200 to 2,500 observations (p. 225).

The fourth step involves the design of the survey instrument including the valuation question. In order to design a credible CV study, Carson notes that "…survey designers need to ensure that prospective consumers understand what they are being asked to value, how it will be provided, and how it will be paid for" (Carson, p. 178) . The response format of the valuation question can be of several different kinds. Boyle (2003) identifies three primary formats: the open ended, payment card and binary choice. Following the recommendations of the NOAA panel (Arrow 1993), the binary choice has become the most popular among CV surveys. One advantage of this elicitation format is that it resembles most situations of consumer choice. Another is that stating a yes or a no is a simpler task for the respondent than it is to specify how a change affects his utility in monetary terms (Whitehead 2006). There are, however, disadvantages with the binary choice. Several studies have observed the so called problem of "yea saying", which implies that some respondents say yes to any bid. Boyle reports about a study that had as much as 30 per cent "yea saying" in a sample. (Boyle 2003, p. 140). Furthermore, binary questions do not induce truthful reporting for provision of a new private or quasi-public good because individuals can decide later whether they will consume or not (Carson et al. 2001). Important advantages of the open-ended question format are the continuous distribution of responses and that there is no need for pre-selection of bids (Boyle 2003). Finally, survey instrument should collect information on the socio-economic characteristics of the respondents (age, gender, income, education, etc.).

The final step concerns data analysis. Boyle (2003) proposes the use of Tobit regression for open ended surveys, because negative values are not allowed and the probable spike at zero. Similarly Haab and McConnell (2002) recommend censored models. Recent surveys that have analysed open ended WTP questions e.g. Christie (2007) and Garcia et al. (2009) apply Tobit and Heckman regressions, while MacMillan (1999) carry out two regression analyses: one binary logit to find the determinants to pay something rather than pay nothing and one stepwise logistic regression covering only positive amounts for analysing the influence of different attributes.

2. PUBLIC BUDGET AND THE ENVIRONMENT

The first aspect of the thesis covers public budgeting and the environment. The two research papers devoted to this aspect examine whether public environmental expenditure is allocated in an environmentally sustainable way and how the financial crisis in 2008 affected environmental expenditures of the state and local government budgets.

2.1. Sustainability of EU fund allocation

The research question of the first research paper is: ”Does EU funding to the environment represent an environmentally sustainable allocation?” Based on the ratified funding plans of Estonia, Latvia and Lithuania the paper sets out to determine whether the choices of the three countries are environmentally sustainable. The paper defines investments environmentally sustainable if they pass the cost-benefit criterion. In case they do not pass, investments can be sustainable only if judged to be critical to sustainability.

Furthermore, investments are qualitatively ranked by their relevance to sustainability. Investments into biodiversity and resource productivity, and into preventive measures are categorised as having high relevance. Investments into pollution control are graded as having medium to high relevance for sustainability while incidental environmental expenditures are ranked as having low relevance.

Following the results of Arrow et al. (2004), who validate that the general cost-benefit rule holds for guiding sustainable investment decisions in an imperfect economy, under the condition that proper accounting prices can be determined, the first step of the evaluation applies the cost-benefit rule. Since existing cost-benefit analyses might not include proper accounting prices, the paper introduces a second qualitative evaluation step. The paper uses the environmental performance index (EPI), which is an ecological indicator to identify fields critical for sustainability.

The conclusions from both steps point in the same direction. Available evidence from cost-benefit analyses suggests that investments into drinking water infrastructure and waste management do not pass the cost-benefit criterion. Investments into sewerage services and waste water treatment pass the cost-benefit criterion, but only when the positive environmental impacts of water bodies are accounted for. Investments into biodiversity protection pass the cost-benefit criterion. The complementary assessment of the second step supports the findings based on the cost-benefit rule. Since EPI does not indicate that there are health considerations due to deficiencies in the existing drinking water infrastructure, these investments are not assessed critical to sustainability. Similarly, there is no indication from EPI that current levels of waste management lead to critical impacts. In addition, the ecological information of EPI suggests that more attention should be paid to biodiversity (e.g. conservation of habitats) and productivity of natural resources (e.g. fishery and cropland intensity), which are receiving only small shares of investments from EU funds in the Baltic States.

Due to the fact that it was not possible to evaluate all investments, the twelve fields were characterised according to their relevance to sustainability. In this classification, it was found that the Baltic States have chosen to allocate least investments into those fields that were found to be most relevant to sustainability, i.e. preventive measures, and biodiversity and resource productivity, which on average receive about 25 per cent of investments.

The result is somewhat surprising since EU fund allocation to the environment does not consider sustainability as priority. Neither do the allocation plans follow the cost-benefit rule. Referring to the three government functions: allocation function, distribution function and stabilisation function, the results suggest that the observed allocation of EU funds to the environment have a significant share of distribution purpose. Rather than reducing externalities and providing public goods, taxpayers in the old EU countries might be interested in raising the standard of water and waste management for reasons of improving living standards of their neighbours. This finding is similar to Pitlik's (2007) observation about the EU budget and fiscal federalism.

It was expected that the three Baltic States would show relatively similar budget allocations. However, country wise comparisons show significant differences in priorities. Estonia has the highest per capita contribution to the environment and also larger investments into those fields with relevance to sustainability. Lithuania ranks lowest according to its per capita funding, but shows a better position than Latvia concerning the highly relevant fields. Determinants of the differences in the choices of countries deserve further research.

2.2. The impact of the financial crisis on environmental expenditures

Since independence Estonian governments have prioritised budget balance and according to Aristovnik and Bercic (2007), Estonia has succeeded much better than most other CEE countries in keeping government budgets under control. Continuation of this policy was evident also during the recent economic crisis. In 2008 the government declared that the transition to the Euro was their highest priority, which led to strict focus on budget balance in accordance with the Maastricht criteria. With this background, the second research paper hypothesises that Estonia's budget balance priority implies that environmental expenditure is highly sensitive in times of crisis.

The paper poses two research questions. "How were public environmental expenditures affected by the recent financial crisis in Estonia?" And the second is: "What are the reasons for the observed developments?".

The method of this research paper follows Vincent (2002). Based on statistics for the period 1995–2008 and preliminary budget data concerning 2009, it is found that during the time period 2007–2009, local government spending on environmental protection fell, while central government environmental protection expenditure increased. It thus seems as local government spending on the environment is sensitive to a decreasing GDP while this is not the case of central government spending on the environment. The paper also shows that public expenditures on environmental protection during the previous financial crisis in 1998–1999 were significantly more sensitive to declining GDP than during the crisis in 2008.

The second research paper offers two possible explanations to the finding that central government expenditure to the environment increased during the observed time period. The first is that by accession to the EU, additional funding became available for environmental protection. The second explanation suggests that the green tax reform, which increased public revenues from environmental charges earmarked for environmental purposes, expanded environmental expenditure. At the time of writing the paper, data were not available for quantifying the impact from these two sources, so it was hypothesised that the main reason was the access to structural funds for financing environmental activities that became available for Estonia in 2007 according to the EU budget for the time period 2007–2013. Future research should follow up more recent developments and aim at quantifying these two explanations. Another interesting issue that deserves further research is to follow up Bowen and Stern's (2010) reasoning about the counter cyclical impacts of investments in environmental protection. Estonia's determination to minimize budget deficits during the critical crisis years combined with a significant inflow of EU funds offers an interesting case study of unintended fiscal policy impacts from environmental projects.

3. FEED-IN TARIFFS

The second aspect of the thesis focuses on the use of feed-in tariffs (FIT) as a tool to promote the development of renewable electricity. The research question is: “Are Estonian FITs effective, cost-effective and efficient and what are their implications on income distribution?”

In research paper 3 the main assessment criteria are: **effectiveness**, **cost-effectiveness** and **efficiency** as well as the implications on **income distribution**. In this respect, the research paper uses an approach similar to Sijm (2002) and del Rio and Gual (2007). The assessment was finalised in December 2010 and considered developments during three and a half years.

3.1. Case study of Estonia

Estonia introduced FITs in May 2007 and according to the Estonian Electricity Market Act, production of electricity from wind, small hydropower and biomass receive a uniform FIT of Euro 54 per MWh (megawatt-hour). The FIT for combined heat and power (CHP) plants differs according to fuel: efficient cogeneration using biomass (wood chips) provides the producer a support at the rate of Euro 54 per MWh for selling electricity to the network, while efficient cogeneration using waste or peat as a fuel is supported by Euro 32 per MWh. The plants receive FIT during the first twelve years of operation and the costs are funded via network charges paid by consumers.

Effectiveness is evaluated by following the capacity development and production of renewable electricity and comparing this to Estonia’s renewable energy targets. According to the National electricity development plan 2005–2015, the goal is to increase the share of renewable electricity to 5.1 per cent in 2010 and to extend the share of electricity from renewable resources to 15 per cent by 2015. For Estonia, these goals imply significant changes. In 2007, the share of renewable fuels in electricity production was less than 2 per cent of gross production[10] while the main supply originated from oil shale electricity, which made up close to 94 per cent. In 2010, Estonia had outperformed the goal of 5.1 per cent as the share of renewable sources contributed to about 10 per cent of electricity production. Data show further that Estonia’s FIT system has effectively contributed to the construction of cogeneration and wind capacity.

For purposes of assessing cost-effectiveness and efficiency, the financial records of two 25 MW_{el} cogeneration plants that started operation in 2009 serve as case studies. Two different approaches were applied to assess profitability without FIT. If found profitable without FIT, the conclusion is that the plants could have operated commercially without subsidies − hence FIT cannot be cost-effective. Based on financial and operating data, the revenue per MWh

[10] The share of renewables from gross inland consumption was 1.5 percent according to Table FE36 of Statistics Estonia.

without FIT were determined to be Euro 40 and Euro 36 respectively, in 2009. Financial data of the annual reports of both plants show that these revenues are sufficient to cover the costs. The second approach applies the rules of the Estonian Competition Authority in order to determine which prices, i.e. unit revenues, the plants need in order to be profitable in the long run when taking debts into account. The results show that required revenues are Euro 26 and Euro 50 per MWh. Hence, one of the plants would have been profitable without FIT. For the second plant the conclusion is uncertain: while the operating revenues without FIT are positive, the second method shows an opposite result. Since it was found that the debt coefficient is surprisingly weak for the second plant, it cannot be excluded that market regulation has provided incentives to the owners to adjust their financial accounts. For this reason, the overall conclusion is that FITs are not cost-effective.

The per MWh revenues were then compared with the marginal cost and the cost price[11] of electricity generation from a CHP plant. If the price (i.e. unit revenue) equals marginal cost, production is assessed efficient. The short run marginal cost was determined to be in the interval of Euro 5–7 per MWh. Both plants receive a market price for their output substantially above the short run marginal costs. Therefore, production is not found to be efficient. Based on Estonian cost data of new cogeneration capacity, Latosov et al. (2011) found that the cost price is Euro 54 per MWh of a 25 MW_{el} CHP plant. As mentioned above the revenues per MWh without FIT were determined to be Euro 40 and 36 respectively. Adding FIT, results in revenues of Euro 94 and 68 Euro per MWh. This comparison further confirms that the plants are overcompensated by the design of the FIT system.

Another aspect of efficiency concerns whether avoided external costs imply a larger benefit than cost when renewables replace fossil fuel electricity. The evaluation of external costs shows that every MWh of oil shale electricity that can be substituted by electricity produced from wood chip in CHP plants reduces the external costs by almost Euro 60 per MWh and if replaced by peat, the avoided cost would be about Euro 43 per MWh. Comparing these values with the Estonian FIT of Euro 54 per MWh and Euro 32 per MWh, show that the estimated environmental benefits are larger than the feed-in tariff. [12] If, however, considering that the climate change impact is zero due to the coexistence of FITs and the EU-ETS (see e.g. Fondler 2010, Sijm 2005) this conclusion changes. When the value of CO_2 is equal to zero, the benefits of replacing oil shale with wood chip is Euro 43 per MWh and with peat Euro 29 per MWh, and if accounting for the addition of non-fossil CO_2 emissions from CHP electricity, there is a further decline in benefits.

The impact on income distribution was assessed by observing the mark-up on electricity consumers. The cost of Estonian FITs has increased at a fast pace from Euro cents 0.1 per kilowatt-hour to Euro cents 0.8 per kilowatt-hour from

[11] The cost price is the price that exactly balances production costs not adding profits.

[12] Estonia's composition of electricity production has changed, but the production of oil shale electricity has not decreased between 2007 and 2010.

2007 to 2010, and consumers have collectively covered these costs through their electricity bill. Beneficiaries are the owners of large cogeneration plants, implying significant distributional impacts.

Table 3 Comparison of German, Spanish and Estonian FIT assessments

	Frondel et al. (2010) and Sijm (2002)	del Rio and Gual (2007)	Kleesmaa, Pädam and Ehrlich (2011)
Assessment period	2000–2008	1999–2003	mid 2007–2010
Effectiveness[a]	Installed total renewable capacity (wind) from 12,000 (6,100) to 45,598 (23,895) MW Growth pa 9 (19)%	Installed renewable capacity (wind) from 3,000 (1,609) to 7,800 (5,976) MW Growth pa 27(39)%	Installed renewable capacity (wind) from 75 (58) to 215 (132) MW Growth pa 42 (32)%
Cost-effectiveness	No	(No)	No
Efficiency (benefits > costs)	No	No (see externalities)	No (see externalities)
Administrative costs	Low in until 2000 High after revisions	Low (easy to implement, no bureaucracy)	Low (only two tariff levels)
Externalities	High CO_2 abatement cost	If only generation costs considered, then benefits of avoided externalities exceed costs	If oil shale is replaced, the avoided externalities exceed the level of FIT (not so if CO_2 value is zero)
Consumer costs Consumer index[a,b]	€ cents 0.3–1.5 /kWh (400%) $HCIP_{\text{all items}}$ (116)	€ cents 0.14–0.26 kWh (185%) $HCIP_{\text{all items}}$ (114)	€ cents 0.1–0.8 /kWh (800%) $HCIP_{\text{all items}}$ (114)

N.A – not available, *a* additional data from Eurostat and Statistics Estonia, *b* base year index=100

Table 3 gives an overview of the results of Estonia in comparison to the assessment of Germany by Sijm (2002), Frondel et al. (2010) and that of Spain by del Rio and Gual (2007). Since the assessments use slightly different assessment criteria and the evaluation period does not overlap, additional data and benchmarks have been added. Effectiveness relates to the quantitative increase. All evaluations show impressive growth rates of capacity. The annual growth rates of Estonia and Spain are very high. Low capacity in the base year and relatively short evaluation periods have an impact. If Germany is evaluated during the time period 2000–2004, total capacity increase is 23 per cent per year and wind 28 per cent, which is relatively close to the observed growth rates of Spain. Interesting to note is that while Germany and Spain have experienced a more rapid increase in wind capacity growth than total growth, this is not the case in Estonia.

The evaluations of Estonia and Germany conclude that FITs are not cost-effective. The Spanish evaluation does not calculate cost-effectiveness, but since the authors conclude that FIT is not efficient (in most cases), it is possible to infer that FIT is not cost-effective. In most cases the evaluations rule out efficiency. However, when generation costs (i.e. excluding fixed costs) are compared to avoided externalities for wind and hydro in Spain and when oil shale is replaced by electricity generated in CHP in Estonia the FIT schemes pass the cost-benefit criterion. Administrative costs are reported to be low in Estonia and Spain. The German FITs were until 2000 based on a percentage of earlier consumer prices of electricity and varied by the source of energy. The new FITs are based on the production costs of various renewable energy resources with digressive payments during 20 years, and therefore assessed as high (Sijm 2002).

One major concern when comparing the assessment of Estonia's FITs with that of the Spanish and German FITs, is the significantly higher increase in consumer costs in Estonia, also when the level of the general price increase has been accounted for.

The greatest drawback of subsidies and taxes is that the extent of their impact is not clear, ex ante. In Estonia, as in many other countries, FITs are used for achieving quantitative goals. It is not easy to select a monetary support matching the goal; therefore the regulation by FIT needs continuous revision. In addition, there is a need to revise the renewable energy support schemes, to make them compatible with the EU-ETS.

4. NON-MARKET VALUATION

The third aspect of the theme of the thesis is valuation of non-market goods. Valuation in the fourth, fifth and sixth research paper is related to how people themselves express the benefits they receive from an improvement in environmental quality. While research paper 4 reports the outcome of a survey that measures attitudes towards the willingness to pay for environmental protection, papers 5 and 6 cover contingent valuation surveys.

In the attitude survey respondents are inquired to express their attitude to questions about their willingness to pay: a) much higher prices, b) much higher taxes and c) accept cuts in their living standard in order to protect the environment. The responses are given on a six point scale: “very willing”, “fairly willing”, “neither nor”, “fairly unwilling”, “very willing” and “don’t know”. In the contingent valuation surveys, on the other hand, the respondents are asked to express monetary amounts that represent their maximum willingness to pay (WTP) as a voluntary contribution for an improvement in environmental quality. In paper 5 the improvement is an enhancement of animal wellbeing defined as provision of adequate enclosures to zoo animals according to the investment strategy of Tallinn Zoological Gardens. The good is a quality aspect of a quasi-public good. At the same time the paper explores whether there is non-use value of animal wellbeing, i.e. pure public good. In research paper 6, the quality improvement is defined as provision of public access to Lake Ülemiste and its surroundings. Hence, paper 6 covers quasi-public good that does not yet exist.

In order to follow survey recommendations, the physical change of the survey needs to be described clearly in order to give the individual an opportunity to perceive how the survey scenario affects her utility. The survey also needs to ask the individual to expresses his monetary WTP amount in a manner that captures his maximum willingness to pay. Table 4 below summarises the physical change of each survey and describes how this is expected to affect utility.

Table 4 Information scenario, physical change and change in utility of the surveys in paper 4, paper 5 and paper 6

Information scenario	Physical change in environmental quality	Change in utility
Paper 4. Environmental protection	Not specified, only reference to environmental protection in general	Depends on individual perception of environmental protection
Paper 5. Provide adequate enclosures to all zoo animals	More habitat-like enclosures for polar bears, giraffes, leopards, wolves, vultures etc.	Increases enjoyment of zoo visits and provides knowledge zoo animals are in good care
Paper 6. Open Lake Ülemiste to the public	New recreation area, lake for bathing and fishing. Potential minor negative impact on drinking water.	People’s valuation of recreation & potential concern about drinking water quality

Coming back to the attitude survey and comparing the theoretical definition of the WTP measure to that is reported in paper 4, it is obvious that the change of environmental quality is only described in general terms of "environmental protection". For this reason, it might be rather difficult for an individual to perceive what kind of change he has been asked to evaluate. In addition, different individuals interpret environmental protection quite differently. Since there is no exact monetary amount, the WTP of the attitude survey is not for this reason in correspondence to the requirements placed by economic theory.

It is important to remember that the attitude survey reported in paper 4 is based on psychological research, which relies on results that show correlation between people's attitudes and their behaviour (for an overview see e.g. Mitchell and Carson 1998) In the model of e.g. Fishbein and Ajzen (ibid.) attitudes are based on individuals' beliefs about attributes related to an object or a policy. In this respect there is a parallel to economic theory which assumes that people have preferences concerning goods and services, as well as policies. Although attitudes and preferences do not correspond they still provide a point of connection between the predictions of psychological research and economic theory about people's choices.

4.1. Public attitudes towards environmental protection

Since contingent valuation survey is context specific, it is not easy to compare surveys carried out in different countries. The advantage of the data set of research paper 4 is that it allows cross-country comparisons. The research questions of paper 4 are: "How do public attitudes in Estonia compare to those in other countries?" and "What are the main determinants of cross-country differences in public support to environmental protection measured as the willingness of individuals to make financial sacrifices or accept cuts to one's standard of living to protect the environment?" The paper offers three hypotheses about the main determinants. The first is that the higher the level of income is in a country, the higher is the share of respondents who are willing to pay higher prices and taxes, as well as accepting cuts in the living standard. The second is that an already high tax burden tends to negatively (positively) affect the share of those willing (unwilling) to pay higher taxes. The third hypothesis is that poor quality of the environment increases the willingness to pay for environmental protection.

The three attitude questions towards the willingness to pay of the Estonian survey were compared with the attitudes of the 24 countries covered by the International Social Science Program (ISSP). In the cross-country comparison it was found that Estonia places itself in the middle position with respect to the two questions about paying higher prices and higher taxes for environmental protection. However, Estonian respondents seem to be more sensitive to cuts in the standard of living than internationally, as Estonia is found in the least willing group of countries in terms of acceptance to cuts in living standard for the sake of environmental protection. This result is in parallel to Bean's (1998) analysis of ISSP data. He found that the willingness to accept lower living standards to protect the environment is higher in Western countries than in former planned economies. It is also interesting to note that the other former planned economies included in the sample differ somewhat in their positions. Slovenia places itself in the top and in the middle, while most of them included in the comparison are found in the lower end of the ranking list for all three questions.

The paper finds evidence of positive correlation between GDP per capita and country level responses to the questions about paying higher prices and accepting cuts in the living standard. Instead of correlating to GDP per capita, the willingness to pay much higher taxes shows some correlation to a country's tax burden. The hypothesis that the share of willingness to pay increases with environmental deterioration does not find support.

In the cross-country comparison, the paper notices a dilemma as some countries found among those least willing to accept higher taxes are among those most willing to accept cuts in the living standard. This is the case of Austria, Sweden and Finland. In addition to Estonia, which moves in the opposite direction, there are also other countries showing a similar shift in position. Table 5 takes a closer look at the rankings with respect to the lower and higher end willingness to accept cuts in living standards.

Table 5 Lower end and higher end countries in terms of willingness to accept cuts in living standard in order to protect the environment, country and ranking

Position	Country ranking (prices, taxes, standard of living)
Least willing countries to accept cuts in standard of living in order to protect the environment Lower end position 25 to 18	Latvia (25,22,25) Bulgaria (22,24,24) Portugal (24,23,23) Estonia* (14,14,22) Czech Republic (20,21,21) Chile* (13,11,20) United Kingdom* (7,8,19) United States* (6,6,18)
Most willing countries to accept cuts in standard of living in order to protect the environment Higher end position 1 to 8	Germany** (17, 20, 8) The Netherlands (1,1,7) Japan (3,2,6) Finland** (23,25,5) Sweden** (15,18,4) Mexico** (18,3,3) Austria** (8,19,2) Switzerland (2,4,1)

Note: rank 1 is the country with the highest share of willingness, rank 25 the lowest
* high share (price/tax) low share regarding cuts, ** low share (price/tax) high share cuts

The numbers in the brackets show the country's ranking in terms of willingness to accept higher prices and taxes, and cuts in the standard of living. The rank goes from 1 to 25, where 1 is the country with the highest and 25 with the lowest percentage of willing answers.

The countries showing a shift similar to that of Estonia are the UK, Chile and the United States. Apart from Mexico, the countries that have a high willingness to accept cuts in the living standard in order to protect the environment, are high income countries with generous social security systems. By symmetry, a not so generous social security system could be the key to why countries shift from high willingness to pay higher prices and taxes to low willingness to accept cuts in the living standard in order to protect the environment.

4.2. Human valuation of zoo animal wellbeing

According to an American survey, people visit the zoo for reasons of social interaction and enjoyable experience interacting with wildlife (Tomas et al. 2003). The same survey finds that zoo visitors rank the health of the animals and viewing them in natural-like habitats as the two most important quality aspects of their visit. This suggests that animal wellbeing is included as one quality aspect of the use value of a zoo.

Since animal wellbeing is one aspect of the services that zoos provide, behavioural data from a revealed preference study would not be sufficient to extract the demand for improvements in animal wellbeing. A contingent valuation survey was therefore carried out. The survey covered both zoo visitors and non-visitors for the purpose of distinguishing between use value and non-use value. By reasons of municipal ownership and funding, the survey population consisted of the adult population of the City of Tallinn. While Tallinn Zoological Gardens is owned and funded by the City of Tallinn, the zoo attracts visitors from all over Estonia and from abroad.

Because of lack of funding the zoo might not achieve its goals of providing adequate accommodation to all zoo inhabitants. The research paper poses two research questions: “Does human valuation of improved animal wellbeing match the postponed investments in provision of adequate accommodation to all zoo animals?” and “Is there non-use value in human valuation of zoo animal wellbeing?”

The questionnaire described the baseline scenario as being a very slow realisation of the zoo development strategy of 2008–2012, declaring that with the current level of funding it may take several decades to provide appropriate enclosures to all animals. The information scenario indicated that those animals currently residing in non-suitable buildings will be provided new enclosures meeting their needs, see Table 4. Further it was stated that according to the development strategy this would require investments of Euro 40 million during a five year period. The WTP question was open-ended with a reminder about the budget constraint.

Probably the most important result of the research paper is that the survey shows that people do express non-use value for improving zoo animal wellbeing.

The analysis of the questionnaire showed that Tallinn inhabitants value an improvement of animal wellbeing at Tallinn Zoo to be on average Euro 17.3 annually, during a five year period. Those who stated that they never visit Tallinn Zoo, expressed non-use value of Euro 7.7, which is approximately 45 per cent of the total value. The survey results were used for estimating the consumer surplus of improving animal wellbeing. This showed that the adult population of Tallinn would receive a welfare improvement of Euro 24 million during a five year period from enhanced animal wellbeing. However, this sum is not sufficient to cover the investments of Euro 40 million that is foreseen by the zoo development plan.

There are two possible policy implications. One is that the zoo is too large for Tallinn and should be reduced in size. This policy implication rests on the argument that the benefits from the scale of the development plan are too small to cover the costs. The second policy implication is based on the theory of fiscal federalism and suggests that responsibility is centralised. This is because the existing funding system disregards the fact that there are beneficiaries of the improvements outside the territory of the City of Tallinn, while tax payers who cover the public funding of the zoo are Tallinn residents. Expanding the survey results to the adult population of Estonia as a whole suggests benefits could reach Euro 80 million.

4.3. The foregone recreation value of lake closure

The sixth research paper is also based on a contingent valuation survey and explores the willingness to pay for recreational visits to Lake Ülemiste. The lake is situated only two kilometres from the centre of Tallinn, but has been closed to public access since Soviet times. Although extensive investments have been made and Ülemiste water purification plant now uses up-to-date technology for drinking water production, the lake is still closed as a precaution. Practices elsewhere suggest that opening the lake to the public would not threaten drinking water quality. The City of Gothenburg, which is of similar size as the City of Tallinn, allows e.g. public use of its surface water reservoirs. For this reason restrictions imposed on Lake Ülemiste imply a potential loss in welfare to the population of Tallinn. The research question of the sixth research paper is: "How large is the foregone recreation value of the closure of Lake Ülemiste?".

Since the area around Lake Ülemiste is fenced the baseline scenario is clear as it assumes continuation of the current status. In the information scenario, the questionnaire provided a description about the recreation potential of the lake and its surroundings, see Table 4. It was also pointed out that there might be a small risk that the raw water that goes to the intake of the Ülemiste purification plant is affected. Balancing people's possible concerns, the text states that Stockholm and other cities using surface water for drinking water purposes allow recreation use of their drinking water reservoir. The respondents were asked to state their maximum willingness to pay in terms of covering potential cost for additional water purification equipment if the lake was opened. The WTP question was open-ended with a reminder about the budget constraint.

According to the responses to the attitude questions of the CVM survey, about one in three supports the idea of allowing recreational access to Lake Ülemiste, while more than one half oppose to the idea. The remaining respondents did not express any opinion. The survey showed that 47 per cent were willing to pay for making Lake Ülemiste accessible to the public. The binary logit regression shows that the only determinant to pay something rather than pay nothing is age and that young people have a higher probability to state a positive rather than zero WTP.

However, the relatively small share of supporters of the idea to open the lake is mirrored by a large share of zero responses: almost 52 per cent. Several authors report that it is not uncommon that open-ended CVM produce a high percentage of zero responses in empirical settings (e.g. Mitchell and Carson 1998 and Boyle 2003). In the case of Lake Ülemiste the large share of zeros could either be an expression of being against the idea or as suggested by Boyle (2003) – unfamiliarity. Probably both explanations are valid. On the other hand, the share of those who said they were opposed but still stated they were prepared to pay for water protection measures was only 2 per cent, which could imply that being very opposed against the suggestion is of minor importance.

In the analysis of the determinants about how much people are willing to pay it was found that age and income are significantly affecting the WTP amount. However, the large share of zeros could affect the regression results. The second step of the estimation which uses Ordinary Least Square (OLS) might for this reason be biased. Since this bias remains in large samples, new regressions have been made by using Tobit estimation in order to determine the possible range of the bias, see Appendix A. The outcome suggests that the bias of the OLS only influences the size of the coefficients, not significances, implying that the results presented in research paper 6 are reasonable.

According to the welfare estimate, the foregone recreation value of lake closure is about Euro1.8 million annually. In comparison to the revenues from water supply services to private clients in 2010, which was Euro 13.2 million (Tallinna Vesi, 2011), this loss might not seem very large. Another way to express this is that the costs of drinking water production are Euro 1.8 million higher than measured by the annual costs of water treatment and its distribution to households. Considering that the loss in welfare accrues every year the foregone value becomes significant. The paper shows that the sum of the net present value is about Euro 26 million during a thirty year period, which significantly exceeds the costs of additional purification equipment. The policy implication is that the city should consider ways to make possible public access to a valuable recreation area within the immediate neighbourhood of the city centre.

4. CONCLUSIONS

This thesis explores three different aspects of the **costs and benefits of environmental policies**. The first aspect relates to the allocation of **environmental expenditure in the public budget**. The second aspect is about the economic efficiency of using **feed-in tariffs (FIT)** as a means to reduce the environmental impact of the energy sector. The third aspect relates to the **valuation of non-market goods** via stated preference methods. Table 6 below summarizes the information from the cost-benefit analyses of the three aspects of this thesis.

Table 6 Summary of cost-benefit analyses

Aspect	Area of environmental policy	Comment
1	Investments in biodiversity protection	Passes C-B criterion
1	Investment in sewerage service and waste water treatment	Passes C-B criterion only if improvement of water bodies is added
1	Drinking water investment	Does not pass C-B criterion
1	Waste management investment	Does not pass C-B criterion
2	Feed-in tariffs	Does not pass C-B criterion
3	Zoo animal wellbeing	National level benefits from improved animal wellbeing are not accounted for
3	Drinking water provision in Tallinn	Additional cost of drinking water provision neglected due to foregone recreation benefits

The cost-benefit analyses covered by the first aspect showed that only biodiversity protection passed the cost-benefit criterion of the suggested investments funded by EU Cohesion Funds. Adding of quality improvements of water bodies to the benefits of investments in sewerage services and waste water treatment made these investments socially beneficial. However, investments into drinking water infrastructure and waste management did not pass the cost-benefit criterion. It is possible to add that both of these investments would provide local level environmental improvements. This in turn implies that from the perspective of fiscal federalism, these investments are, for efficiency purposes, not expected to be covered by EU-funding. The evaluation of the second aspect showed that the Estonian feed-in tariffs are costly and non-efficient. Additionally, they lead to adverse impacts on income distribution. The case studies of the third aspect implied that current decision making does not account for all benefits and might, therefore, lead to under-provision of public goods. The policy implication of the case studies is that application of cost-benefit analysis has potential to increase the precision of current policies and thereby improve efficiency of environmental policy making.

The enquiry into the **allocation of environmental expenditure in the public budget** showed that Estonia's, Latvia's and Lithuania's EU-fund allocation plans to the environment did not consider sustainability as priority. Neither did the investments apply to the cost-benefit rule. Rather than efficiently reducing externalities and providing public goods, the implication seems to be that EU funds have been put into use for improving living standards in new member countries by raising water quality and waste management standards. Had cost-benefit analysis been applied when preparing the investment plans for the period 2007–2013, there had been potential to enhance the social benefits of the investment plans.

Based on public expenditures for the time period 1995–2008 and preliminary budget data concerning 2009, it was found that during the time period 2007–2009, local government spending on environmental protection fell, while central government environmental protection expenditure increased in Estonia. This observation suggests that, at least in the case of Estonia, the timing of the EU budget with significant inflow of environmental spending during the crisis years has had potential counter-cyclical effects. Although unexpected, Estonia might have received unintended stabilisation support via the environmental funds of the EU. The observations about the environmental fund allocation from the EU budget thus suggest that instead of being primarily allocative, which is what could be expected of environmental expenditure, the funding has instead been redistributive and possibly had stabilisation function.

The second aspect of the theme of the thesis concerns the **use of feed-in tariffs** (FIT) as a support scheme to promote production of renewable energy. The assessment found that the Estonian FIT system has effectively supported establishment of CHP capacity, the administrative costs have been low and the avoided external costs have exceeded the cost of the support. However, the conclusion about external costs changes when assuming that the EU-ETS becomes operational. This is because the climate impacts of FITs become zero due to the quantity constraint of the EU-ETS. In addition, the assessment showed that the current system is costly and non-efficient: the costs of the Estonian FITs have increased at a rapid rate and these have been paid collectively by consumers while beneficiaries are overcompensated. One major concern when comparing Estonian FITs with those of Spain and Germany, is the significantly more rapid increase in consumer costs in Estonia.

In Estonia the FITs were introduced in order to achieve quantity goals. The goal for 2020 is to achieve 20 per cent renewable electricity of gross production. Using a pricing measure to achieve a quantity goal has its drawbacks. One is that it is not possible ex ante to predict the impact on quantity. Since it is not easy to select a monetary support matching the goal, the regulation by FIT needs continuous revision. In addition to the country level concerns, there is an urgent need to revise the renewable energy support schemes in the EU and to make them compatible with the EU-ETS.

The third aspect of the theme of the thesis is **valuation of non-market goods.** The cross-country comparison of attitudes toward environmental protection found that Estonia places itself in the middle position with respect to making financial sacrifices for environmental protection, but in the lower end concerning cuts in the standard of living. The basic cross-country statistical analysis showed that the higher the level of income is in a country, the larger is the share of positive attitudes toward environmental protection. This is the case when respondents consider paying higher prices and accepting cuts in the living standard. From the general perspective, Estonia's shift in positions between these two questions is somewhat unexpected. The explanation might be that a not so generous social security system reduces the willingness to accept cuts in living standard to protect the environment. When respondents are asked about their willingness to pay much higher taxes for environmental protection there is no correlation to the country level of income. Instead a country's tax burden provides some explanation to the attitude towards paying much higher taxes for the sake of the environment.

The survey of human valuation of zoo animal wellbeing showed that people express both use and non-use value for improvements in the living conditions of the animals at Tallinn Zoological Gardens. The finding that individuals experience non-use value for zoo animal wellbeing is probably the most important finding of the third aspect. Still however, the total benefit to the adult population of Tallinn is not sufficient to cover the cost of investments foreseen by the zoo development plan. There are two possible policy implications. One is that the zoo is too large for Tallinn and should be reduced in size. This policy implication rests on the argument that the benefits from the scale of the development plan are too small to cover the costs. The second policy implication is based on the theory of fiscal federalism and suggests that responsibility is centralised. This is because the existing funding system disregards the fact that there are beneficiaries of the improvements outside the territory of the City of Tallinn, while the tax payers who cover the public funding of the zoo are Tallinn residents.

The analysis of the welfare loss of the lake closure to assure drinking water quality showed that the loss to Tallinn inhabitants is about Euro 1.8 million annually. According to the city's practices there is no trade-off between drinking water provision and recreation use. This is in contrast to water protection policies elsewhere. A change in views would make possible public access to a valuable recreation area within the immediate neighbourhood of the city centre.

The general conclusion of the case studies reported in the research papers is that those areas covered by environmental policies often are too costly in relation to their benefits, while neglected areas deserve greater attention. The cost-benefit criterion and ecological indicators suggest too much is invested in waste management and drinking water infrastructure. Similarly, resources devoted to the production of renewable electricity are put in inefficient use by producing electricity at a high cost and in a larger quantity than foreseen by Estonia's goal. The loss of recreational values from lake closure implies that the annual costs of drinking water provision are higher than measured by the annual costs of water treatment and distribution. Further, the case studies suggest that there is under-provision of biodiversity enhancement and high quality living conditions of zoo animals and that current water policies neglect significant recreation benefits of city dwellers.

Appendix A. Tobit regression

Papers 5 and 6 apply a two-step regression approach. Binary logit regression is used for finding the determinants of a positive rather than a zero WTP amount. In the second step, the papers apply Ordinary Least Square (OLS) estimation in order to determine the significance of socio-economic factors to the amount people are willing to pay. As discussed in Section 4.3 the share of zero responses is high in the Ülemiste survey. In statistical terms observations are censored at zero implying that the variable is not observed over its entire range. When a large fraction of observations are censored at the maximum or the minimum, OLS produces biased results. Since this bias remains in large samples, bias might be a serious issue in the results of paper 6. Therefore, based on the same data set, new regressions were run in order to determine the possible range of the bias. Out of 1,241 questionnaires that contained a WTP amount, 24 observations which lacked socio-economic variables were dropped. Dummies were formulated for gender; D_G (male=1, female=0), secondary education, D_S and university education D_U. Age and Income remain coded as in paper 6. The Tobit model assumes that WTP_i^* is the latent and WTP_i is the observable variable. The observable variable equals the latent variable for positive values; i.e. when $WTP_i > 0$ then $WTP_i = WTP_i^*$. Observed zeros correspond to zero and negative values of the latent variable; i.e. $WTP_i = 0$ then $WTP_i^* \leq 0$.

The new regression equation is:

$$WTP_i = \beta_0 + \beta_1 D_G + \beta_2 D_S + \beta_3 D_U + \beta_4 AGE + \beta_5 INCOME + v_i + u_i \quad (4)$$

Table 7. Determinants of the WTP amount according to Tobit regression in comparison to OLS. Based on 1,217 observations, of which 644 observations are censored to the left.

	Tobit			OLS		
	Coeff. β_i	S.E	t-ratio	Coeff. β_i	S.E	t-ratio
Constant	-7.89327	5.98063	-1.32	4.42599	3.16722	1.40
Gender	1.96531	2.12922	0.92	2.00361	1.14462	1.75
Secondary education	0.13395	5.67348	0.02	0.22143	2.94199	0.08
University education	2.05167	5.80986	0.35	0.21502	3.02683	0.07
Age	-2.98688***	0.58273	-5.13	-1.00453***	0.30706	-3.27
Income	1.89555***	0.59097	3.21	1.08410***	0.31844	3.40

*** significant on 99 per cent level.

All coefficients except the constant have the same sign in the Tobit and in the OLS regression. Both estimation methods show that age and income are significant on the 99 per cent level. Comparison of the coefficients shows that gender and secondary education are of the same order of magnitude. The coefficients of university education, age and income are larger in the Tobit regression than according to the OLS. These results suggest that the bias of the OLS only influences the size of coefficients, not significances, implying that the results presented in paper 6 are reasonable.

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Appendix 1. “The Impact of EU Cohesion Policy on Environmental Sector Sustainability in the Baltic States”

The impact of EU Cohesion policy on environmental sector sustainability in the Baltic states

Sirje Pädam, Üllas Ehrlich, Koidu Tenno[1]

Abstract

This article analyses investment from European Union cohesion policy funds into the Estonian, Latvian, and Lithuanian environmental sectors during the budget period 2007-2013. Total investment from these funds in Estonia, Latvia, and Lithuania during that period will be about 14.7 billion euros, of which about 18 percent covers the environmental sector.

The purpose is to analyse whether allocation of expenditure to the environment is sustainable. In their analysis the authors apply sustainability criteria based on the cost-benefit rule and the Environmental Performance Index (EPI). The main finding is that the Baltic States allocate least environmental funds to those fields found to be most relevant to sustainability.

Keywords: environmental investment, EU funding, sustainability
JEL classification: H59; Q20; Q28; Q58

1. Introduction

Vincent and his co-authors (2002) note that despite strong reasons for analysing public expenditure and the environment, only limited literature is available within this field. So far, most analyses concerning public expenditure on the environment have been undertaken by the World Bank and the OECD. This paper aims to fill the gap and offers a novel perspective into the study of allocation of public expenditure to the environment by comparing EU cohesion policy fund allocation to the environment in three countries of similar size and corresponding economic prerequisites.

The analysis concerns the structure of EU cohesion policy funding for the environment in Estonia, Latvia, and Lithuania during the period 2007-2013. Since all countries eligible for funding are subject to the same regulations, it is expected that funding choices will be similar. However, country specific time schedules for fulfilling EU directives agreed on during membership negotiations can be a source of differences. The overall purpose of the analysis is to assess whether budgetary allocation to the environment according to funding plans supports sustainability of the environmental sector. Funding plans, the outcome of negotiations

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between each beneficiary country government and the EU Commission, are documented in National Strategic Reference Frameworks 2007-2013 and Operational Programmes.[2] The Operational Programmes for Estonia, Latvia, and Lithuania represent the primary source of data of this paper.

We begin by describing the theoretical framework for defining an efficient and environmentally sustainable resource allocation. Based on the theoretical framework, we then present an outline for step-wise assessment of sustainability of budgetary allocation to the environmental sector. In the following section we present the outcome of cohesion policy fund allocation in the three Baltic States. After that we carry out step-wise assessment and, based on the results, we classify spending priorities according to their relevance to sustainability. The conclusions are presented in the last section.

2. Theoretical framework

Environmental regulations and public expenditure directed to the environment are generally justified by efficiency reasons. This is because unregulated markets pay too little attention to environmental protection, i.e. environmental quality. Supply of environmental goods may be insufficient since they are public goods, while oversupply of activities that give rise to negative externalities can also occur. The role of government expense on the environment is thus to redirect tax income to provision of public goods and to tax activities that give rise to negative externalities. To some extent, environmental protection can be self financing if taxes and charges paid by polluters are directed to rehabilitation and pollution control.

In this paper we deal with supra-national funding where Member State payments are reallocated among EU countries. For this reason, the concept of fiscal federalism can be applied to allocation of public expenditure. Fiscal federalism addresses the problem of vertical allocation of economic responsibilities by level of government. Efficient allocation assigns the responsibility to the territorial authority where beneficiaries correspond to that of taxpayers (see Pitlik, 2007). If the benefits of public goods spill over to a neighbouring territory or country, this gives reason to centralize responsibility. Fiscal federalism would thus predict that EU funding to the environmental sector is devoted to environmental issues with cross border characteristics. In addition, efficiency reasons would motivate higher levels of funding when neighbouring countries benefit from improvements. Pitlik (2007) finds that almost half the financial resources of the EU budget are allocated to spending categories in which EU responsibilities are questionable from the viewpoint of fiscal federalism. Since one of the intentions of cohesion funding is to reduce disparities among Member States, regions, and individuals[3], it is likely that the concept of fiscal federalism is not applicable for our purpose.

Our main focus of environmental spending involves sustainability. Analogous to sustainable development, sustainability represents resource use that meets human needs while preserving the environment so that needs can be met for both present and future generations. The literature suggests a close relationship between efficiency and sustainability (see e.g.

[2] See list of references.
[3] See COM 2008/301 (2008).

Pädam, 2003). Efficiency implies resource allocation that considers peoples' preferences and accounts for resource constraints. By allowing for reallocation of resources in case human needs are not met and by adopting a dynamic perspective, efficiency will overlap with sustainability. According to the interpretation by Stavins *et al.* (2003), sustainability can be understood as dynamic efficiency along a feasible consumption path. Sustainability entails non-wastefulness, implying that the choice of a consumption path is such that the economy is on the Pareto frontier. Following Stavins *et al.*'s application of a Ramsey type of presentation, welfare, *W,* of such a path can be evaluated over time as:

$$W(t) \equiv \int_t^\infty U(C(\tau))e^{-r(\tau-t)}d\tau \qquad (1)$$

where *U* denotes the utility function which depends on consumption, *C,* including both direct consumption and enjoyment of non-market goods and services. Time is denoted by τ and t (τ, $t \geq 0$) and the time horizon is taken to be infinite. The utility discount rate is denoted by *r*. Since *C* contains two types of goods, the argument of the utility function can be rewritten as:

$$C = f(x(\tau), z(\tau)) \qquad (2)$$

where x(τ) denotes market goods and z(τ) denotes non-market goods, including environmental goods and services. In order to be sustainable, current decision making must consider the perspective of inter-temporal public goods and inter-temporal externalities. Securing future supply of environmental goods and services implies production of inter-temporal public goods, which need to be provided so as to include the preferences of future generations. Stavins *et al.* formulate a condition of intergenerational equity requiring non-decreasing welfare:

$$\frac{dW(t)}{dt} \geq 0 \qquad (3)$$

The requirement that the stream of welfare does not decline over time implies that future generations will not be worse off. Although constant consumption at no more than subsistence level could in principle meet the definition of sustainability, Stavins *et al.* (2003) argue that this definition would not be accepted as meeting reasonable social goals. For evaluating sustainability they propose a decision rule similar to the Kaldor-Hicks criterion, i.e. that those who are made better off by a policy in theory can fully compensate those who are made worse off. A policy that fails the Kaldor-Hicks test cannot pass the stricter Pareto test. In a dynamic context, intergenerational transfers could be applied to achieve non-declining welfare. This is the justification for their proposal to use dynamic efficiency as a criterion to find policies that are potentially sustainable.

Although intuitively appealing, the approach of Stavins *et al.* (2003) disregards two central issues: one is the implicit assumption they make about **natural capital** and the other is the **preferences of future generations**.

The implicit assumption that they make about **natural capital** is that natural environments and ecosystems can be represented by equations that are convex sets and that are at least twice differentiable. However, this need not be the case. The reason is that regeneration paths of natural environments and ecosystems tend to exhibit nonlinear dose-response relations, implying that marginal changes in anthropogenic pressure may result in irreversible effects (see Dasgupta and Mäler 2004). Therefore no guarantee exists that equation (3) is non-negative or that the inequality can be defined in a meaningful way. In an analysis of policy reforms in imperfect economies, Arrow *et al.* (2004) show that social welfare might or might not be sustained between two periods. Reasons why an imperfect economy is incapable of sustaining welfare over time include e.g. scarcity of resources and limited substitution possibilities among capital assets. At the same time, Arrow *et al.* (2004) show that the general cost-benefit rule holds for guiding sustainable investment decisions in an imperfect economy. But, in order to certify that the cost-benefit rule produces correct estimates, it will become necessary to derive proper accounting prices, which can to a large degree differ from market prices (*ibid*). In the absence of proper accounting prices, the need arises to find other ways to consider scarcity and the need for preservation of key natural resources.

Finding information about proper accounting prices is not only hindered by lack of know ledge about non-linear dose response relations of natural environments. Another difficulty in determining sustainable development over a long period or even more so over an infinite time span is lack of information about the **preferences of future generations**. Current decisions that affect sustainable development would need to take into account estimates of willingness to pay by unborn persons in the distant future. Taking a closer look at decision making, we can see that people do not tend to give up decision making in those cases where their decisions tend to have an impact on future generations. In several cases people even include the welfare of their children or grandchildren in their decisions. Monchareva and Gudas 2009 report that a large portion of respondents declare that improving the water quality in the Nevezis river basin is important "for children and for future generations' wealth". This implies that current generations have the capacity to represent future generations. Assuming that the preferences of current generations contain the requests of future generations on the natural environment implies that willingness to pay estimates based on generations now alive can be approximated as representative of the preferences of future generations.

However, we cannot expect to find the whole answer from willingness to pay estimates based on generations now alive. The failure of humans to put an accurate value on critical natural assets is due to the inherent complexity of the natural environment. Taking into account that human preferences cannot correctly sense when ecosystems are at risk implies a need to use knowledge of ecological science in order to identify critical environmental assets.

3. Combined approach

Since it may prove impossible to collect proper accounting prices by estimating willingness to pay (WTP) for natural environments and ecosystems, i.e. the accounting prices of $z(\tau)$ in equation (2), from generations now alive, the implication is that a need exists for a combined approach to assess the sustainability of environmental spending. In our analysis we will consider the cost-benefit rule in the first step for assessing sustainability and in the second step we will use ecological knowledge in order to certify that investments will be undertaken in critical fields of $z(\tau)$. For the purpose of the second step we use the Environmental Performance Index (EPI), (see Esty *et al.* 2008). This index is based on empirical data about the environment in 149 countries and allows cross country comparisons. The index has been developed by first identifying specific environmental targets and then measuring the distance between the target and current national achievement (*ibid*). Although the authors identify several data gaps, EPI is a comprehensive measure based on ecological knowledge.

In terms of our purposes, EPI is no substitute for WTP estimates. Instead we need EPI in order to complement the information of the cost-benefit rule. Since EPI is available for a broader range of environmental issues than WTP estimates, we will use EPI as an indicator for suggesting additional policy implications when WTP estimates are missing. However, EPI cannot assess the range of required investment and cannot measure whether a certain level of investment passes the cost-benefit rule.

4. Budget allocation to the environmental sector

Cohesion policy funding included by the Convergence Objective during the programming period 2007-2013 amounts to about 346 billion euros. Among the Baltic States, funding per country is between 3.4 and 6.8 billion euros. Estonia obtains less than Latvia, and Lithuania receives more than the two other Baltic states. The ranking of the contribution to the environmental sector shows similar positions between countries. Lithuania devotes most, then Latvia, while Estonia assigns least funds to the environment, see Table 1.

Table 1. Allocation of cohesion policy funding to the environment, in total and per country 2007-2013, euros current prices**

Priority theme	Estonia	Latvia	Lithuania	Community Wide*
Euro, million				
Environment	781.3	792.7	1,053.4	46,735.9
Total	3,403.5	4,530.4	6,775.5	346,150.8
Euro, per capita				
Environment	582	347	311	270
Total	2,535	1,986	2,002	1,997
Percent				
Environment	23.0	17.5	15.5	13.5
Total	100.0	100.0	100.0	100.0

Sources: Operational Programmes, COM 2008/301(2008) annex 1 and Eurostat (2008). Population data for January 2007: Estonia 1,342,409, Latvia 2,281,305, and Lithuania 3,384,879.

*Community wide covers Member States and regions falling under the convergence objective covering 35 percent of the Union's population.

** All amounts expressed in current prices. To accommodate inflationary expectations during 2007–2013, EU countries agreed to adjust financial framework ceilings (expressed in 2004 prices) by using a yearly 2 percent price deflator between 2004 and 2013.

The primary reason why funding differs between countries is due to country size. Dividing funding by population puts these figures into another perspective. The per capita allocation of cohesion policy funding to the environment is highest in Estonia and lowest in Lithuania. In comparison to the community wide allocation of cohesion funding that falls under the convergence objective, all three Baltic states devote more to the environment than is directed by cohesion funding on average.

The definition of community funding devoted to the environmental sector includes 12 out of a total of 86 priority themes. The chosen priority themes include all but one theme of the category "Environmental protection and risk prevention" and two priority themes of "Tourism": see EU (2006) for a complete list of priority themes. Our definition of the environmental sector is closely connected to fields commonly included in environmental protection expenditure of the general government budget. The fields used in general government expenditure include waste management, waste water management, pollution abatement, protection of biodiversity and landscape, and R&D in environment protection.

Expenditure to reduce contribution to climate change is not explicitly included in our definition other than forming part of pollution abatement. One reason is the choice to follow the fields in general government expenditure. Another reason for not including climate change is that the Baltic states have different starting points depending on major differences in energy supply between countries. Leaving out investment in energy efficiency, renewable energy, and environmentally friendly transportation thus allows for a more equivalent base when making cross country comparisons between the Baltic states. In addition, a comparison of impacts of EU cohesion funding on climate change has been made elsewhere (see CEE Bankwatch Network 2007). Table 2 shows allocation of funding by priority theme of the three Baltic states and a comparison with community wide allocation for Member States and regions falling under the Convergence Objective.

Table 2. Cohesion policy funding for the environment, per priority theme 2007-2013, euros per capita current prices** and percent

Priority theme	**Estonia**		**Latvia**		**Lithuania**		**Community wide***	
Management of household and industrial waste	52	8.9%	57	16.4%	82	26.5%	36	13.4%
Management and distribution of water (drinking water)	152	26.1%	123	35.5%	61	13.0%	47	17.3%
Water treatment (waste water)	152	26.1%	123	35.5%	41	19.6%	81	29.9%
Air quality	10	1.7%	0	0.0%	51	16.3%	6	2.2%
Integrated prevention and pollution control	0	0.0%	0	0.0%	0	0.0%	4	1.6%
Mitigation and adaptation to climate change	0	0.0%	0	0.0%	0	0.0%	2	0.7%
Rehabilitation of industrial sites and contaminated land	103	17.7%	21	6.2%	4	1.4%	20	7.4%
Promotion of biodiversity and nature protection (including Natura 2000)	16	2.7%	11	3.2%	26	8.3%	16	5.8%
Risk prevention (including drafting and implementing plans and measures to prevent and manage natural and technological risks)	29	5.0%	11	3.2%	0	0.0%	34	12.6%
Other measures to preserve the environment and prevent risks	50	8.6%	0		23	7.5%	10	3.6%
Promotion of natural assets	9	1.6%	0	0.0%	23	7.5%	7	2.5%
Protection and development of natural heritage	9	1.6%	0	0.0%	0	0.0%	8	3.0%
Environmental sector, total	582	100.0%	347	100.0%	311	100.0%	270	100.0%

Source: Authors' calculations based on Operational Programmes, COM 2008/301(2008) annex 1 and Eurostat (2008).
*Community wide covers Member States and regions falling under the convergence objective covering 35 percent of the Union's population.
** All amounts expressed in current prices. To accommodate inflationary expectations during 2007–2013, EU countries agreed to adjust financial framework ceilings (expressed in 2004 prices) by using a yearly 2 percent price deflator between 2004 and 2013.

Not all priority themes have been covered by the Baltic states. It is interesting to note that no Baltic state will invest in the priority themes of "Integrated Prevention and Pollution Control" or "Mitigation and Adaptation to Climate Change". Moreover, community wide investment in these two priority themes is low. Estonia is the only Baltic state to allocate funds to "Protection and Development of Natural Heritage". The other priority themes are covered by at least two Baltic states. In total, Estonia's funding covers 10 priority themes, Latvia's 6, and Lithuania's 8.

All Baltic states prioritise drinking water distribution and waste water treatment. These two priority themes are top priorities in Estonia and Latvia. Lithuania puts top priority on waste management, while drinking water and waste water treatment come at numbers two and four, respectively. Air quality is the third priority for Lithuania, while Latvia's third is waste management. Estonia's third priority is rehabilitation of contaminated land. Ranking of priority themes by expenditure is relatively similar in Estonia and Latvia for common fields, while Lithuania shows another ranking of priorities in that it includes a relatively large share of promotion of biodiversity and natural assets. Community wide priorities rank waste water treatment as top priority, followed by drinking water supply, and waste management.

Notwithstanding comparable economic prerequisites and similar country size, funding plans for Estonia, Latvia, and Lithuania reveal larger differences than were expected. One reason for greater focus on drinking water in Estonia and Latvia may be that that Estonia and Latvia were granted transitional periods for fulfilling the directive on drinking water quality, while Lithuania was expected to fulfil the requirements on accession. In addition, Estonia's funding plans cover a larger number of priority themes than Latvia's and Lithuania's and shows larger per capita spending on the environment. These differences may be due to the fact that Estonia's production of electricity gives rise to significant pollution and that environmental protection was a major issue during the struggle to regain independence. Latvia has chosen fewer priority themes than its Baltic neighbours, but will spend more per capita than Lithuania. In Lithuania, biodiversity and natural assets receive a larger share of funding than in the other Baltic States.

5. Assessment of sustainability

The observations above raise questions about whether more investment into the environmental sector is better from the viewpoint of sustainability and how the various priority themes add to sustainable development. Based on our initial discussion, sustainability can be assessed by the cost-benefit rule. However, since human preferences cannot correctly sense when ecosystems are at risk, WTP estimates might not produce proper accounting prices. Therefore, we will need to assess sustainability in two steps.

5.1. Cost-benefit rule

Applying efficiency motivations to spending priorities, market failure can motivate all priority themes that were included in the environmental sector. Several priority themes deal with alleviating negative externalities including waste management, waste water treatment, and pollution control. Other priority themes can be motivated by reasons of provision of public goods, including air quality, rehabilitation of contaminated land, and promotion of natural assets. Drinking water infrastructure is not a public good, but its provision can be classified as market failure since the supply of drinking water infrastructure is characterised by increasing returns to scale.

Existence of market failure is not sufficient to conclude that a certain priority theme needs funding for reasons of efficiency. In addition, the cost-benefit rule requires that total benefits exceed total costs, i.e. that willingness to pay (WTP) for the services or goods in question covers costs. Unfortunately, we know very little about whether willingness to pay covers the costs associated with the priority themes. However, some evidence exists for four priority themes.

5.1.1. Waste management and sewerage services

Bluffstone and De Shazo (2003) report estimates of willingness to pay for two priority themes in Lithuania. They estimated the cost of implementing EU directives on waste management and urban waste water treatment and conducted contingent valuation studies among Lithuanian households in Ukmerge municipality 40 kilometres north of Vilnius. The population is approximately 34,000 and the average monthly household income is close to the national median (see Bluffstone and De Shazo, 2003).

In interviews with households, the benefits of improved landfill construction and closure of old landfills were described in terms of avoiding pollution to surface and ground water and that after closure old landfills would be sealed and replanted to avoid future contamination. Respondents who indicated they had no access to the sewerage network were surveyed for their WTP to be connected to the municipal sewerage system. The benefits of municipal sewerage services were described in terms of there being no need to service their private septic system or pit toilet and no smell once connected to the municipal system. The authors found that at least 50 percent of respondents would be willing to pay 0.62 euros (2.73 litas) more per person and year for landfill upgrade and that half of respondents were willing to pay an additional 0.51 euros (2.24 litas) per person and per year for sewerage services. The average household size in Ukmerge is 2.67, thus producing household WTP of 1.7 and 1.4 euros respectively.

Assuming that the households studied are representative of Lithuania, the authors estimated that national WTP covers between 80-90 percent of the costs of improving waste management practices, but that WTP for sewerage services covers only 10 percent of the costs (*ibid*). This implies that neither of the directives produces benefits large enough to cover costs. However, one limitation of the benefit estimation of sewerage services is that benefits from improved environmental conditions are missing. These include, for example, benefits that arise from improved water quality in local surface water bodies, enhanced fisheries, and improved recreation opportunities. Since 1.4 euros per household covers only 10 percent of costs, the improved environmental conditions of water bodies resulting from the urban waste water treatment directive must cover at least the remaining 90 percent of the 14 euros (i.e. 12.6), in order to pass the cost-benefit rule.

5.1.2. Water quality

In a recent article, Monarcheva and Gudas (2009) review three contingent valuation studies that have presented monetary WTP estimates for improving the water status in the river basins of the Nevezis (Lithuania), the Ludza (Latvia), and the Valgejogi (Estonia). The Lithuanian and Latvian studies measured the WTP for improving water quality from poor to good, while the purpose of the Estonian study was to estimate the value of restoration of salmon and other rare fish species in the Valgejogi River. These studies differ in certain respects. Firstly, the Lithuanian and Latvian studies focus on water quality and the Estonian on restoration of fish populations. Secondly, the authors mention a significant socio-economic difference between the Latvian study and the other two, as the Latvian study area has low population density and low income levels. Since WTP estimates are generally strongly correlated to income, it is reasonable to expect that the Latvian WTP is lower than the Lithuanian. The results, expressed in annual WTP in euros per household, are reported in the table below. The values in brackets represent estimates when zero bidders are included.

Table 3. Willingness to pay (WTP) for improving water quality of river basins in the Baltic states, euro per year

Environmental good	WTP per year per household, euros
Restoration of salmon and other rare fish species in Valgejogi river (Estonia)	22.8 (22.8)
Improving the water quality of Lake Ludzes and the upper part of the river Ludza river basin (Latvia)	13.7 (6.2)
Water quality improvement of the Nevezis River basin (Lithuania)	20.5 (13.3)

Source: Monarcheva and Gudas (2009)

In order to use the WTP for water quality estimates we would like to know whether implementation of EU directives on urban waste water management will result in improvements that have been valued by the Estonian, Latvian, and Lithuanian studies. The Estonian estimate concerning restoration of fish stocks seems less suitable for our purpose. The Latvian and Lithuanian studies seem to be more in line with expected impacts from improved sewage treatment. Including zero bidders, the Latvian and Lithuanian estimates produce a span of WTP for water quality improvements ranging from 6.2 to 13.3 euros annually per household. Assuming that the Latvian and Lithuanian WTP estimates approximately relate to the water quality benefits of the EU directive, this suggests that benefits might not be sufficient to cover the remaining 90 percent of the costs of about 12.6 euros.

5.1.3. Drinking water

Experience from Poland implies that public willingness to pay for municipal services is higher for drinking water than for waste water services: see Stanek (2002). This seems logical based on the fact that people pay relatively more for safe drinking water, such as bottled water. However, a high WTP for drinking water does not seem to be the case for the Ukmerge municipality. One explanation may be that the WTP for safe drinking water only concerns a limited quantity of the water consumption of an average household. In the background documentation of the Ukmerge study, DEPA and DANCEE (2001) report the WTP for water supply. According to DEPA and DANCEE (2001), the quality of tap water distributed to households in Ukmerge municipality is checked regularly, but due to insufficient water pipe maintenance, households from time to time receive tap water with an orange/red colour or an odour. Respondents in Ukmerge were asked to value the benefit of upgrading the water supply pipes to ensure that the water supply system would be safe and so that no colour or odour would be present. The WTP was estimated to be 1.44 litas per person, per year, which corresponds to approximately 74,880 litas per year for the whole municipality. The estimated annual cost of upgrading the water supply pipe in Ukmerge municipality was estimated at approximately 10 million litas, suggesting that benefits cover less than 1 percent of investment.

5.1.4. Promotion of biodiversity

Ehrlich *et al.* (2008) compared the costs and benefits of biodiversity enhancement by expanding the area covered by semi-natural plant communities in Estonia. Semi-natural plant communities, such as meadows, were developed by scythe, axe, fire, and grazing. These landscapes can persist only with support from human activity, such as mowing, grazing, and brush cutting. In 2007, semi-natural plant communities covered approximately 10,000 hectares in Estonia and the area is declining. Since these semi-natural plant communities are a prerequisite for richness in biodiversity and for migrating birds, the decline of traditional farming activities has put biodiversity under threat.

Ehrlich *et al.* (2008) estimated that annual WTP was 265 euros per hectare of semi-natural plant communities. This amount was derived from the annual WTP estimate of 11.8 euros per person of the working age population. Based on an inventory covering all 31 protected areas in Estonia, the costs were collected for extending preservation of semi-natural plant communities to all Natura 2000 areas. This inventory was a base for Estonia's funding plan for the priority theme promotion of biodiversity and nature protection. The present value of costs for extending the preservation areas to 19,334 hectares was estimated at 56.3 million euros. The cost estimate includes both running costs and investment costs during a 30 year period using a discount rate of 5 percent. The present value of benefits was found to be 89.0 million euros. The results indicated that willingness to pay for biodiversity enhancement exceeded costs by 58 percent.

5.1.5. Benefit transfer

The cost-benefit rule can only be applied to four priority themes, and this scattered evidence gives point estimates for Lithuania in three out of four cases and in one case for Estonia. Benefit transfer from one country to another is a relatively common practice in literature and for policy purposes. However, differences in socio-economic characteristics and in the physical characteristics of study sites influence WTP. Generally, income is the most important variable affecting WTP estimates.

The authors of the Lithuanian studies (see DEPA and DANCEE, 2001 and Bluffsone and De Shazo, 2003) assume that their results can be transferred to Lithuania as a whole. This is based on the fact that the socio-economic characteristics of Ukmerge are assessed as representative of the whole country. The Estonian study is based on a representative sample of the working age population (see Ehrlich *et al.* 2008). There are some differences in GDP per capita between the Baltic States, but from an EU perspective the income levels are similar. In addition, EU directives have imposed comparable requirements on the Baltic States.

Assuming that it is possible to transfer the above results between the Baltic states implies that too many funds will be devoted to improving drinking water supply. Waste management funding probably also receives more funds than desirable. Connection to the municipal sewerage system and upgraded waste water treatment can only be motivated if the benefits of improved water quality are added to the benefit estimate. The scattered evidence further suggests that promotion of biodiversity receives too little funding. The results thus imply that from an efficiency point of view the funding plans will oblige the Baltic states to invest more than is socially desirable in drinking water and waste management. The implication is thus that support to drinking water and waste management should be reduced, while financing of biodiversity should be increased.

5.1.6. Cross border benefits

Prior to arriving at conclusions concerning the first step of the assessment, it will be important to assess potential cross border benefits. We expect that more expenditure is allocated to priority themes that give rise to cross border benefits than can be motivated by national benefits.

The priority themes of the environmental sector that can motivate costs exceeding national benefits are those that have significant cross border impacts. Potential priority themes include pollution control in those cases when air and water pollutants spread on a regional scale. Reduction of environmental risk could have cross border benefits if environmental damage spreads across national borders. Promotion of biodiversity, natural assets, and natural heritage might also have cross border benefits if citizens in other countries express use value or non-use value for preservation.

Probably the most important cross border benefits are those that concern the water quality of the Baltic Sea. In the mid 1990s an extensive inter-disciplinary study on the state of the Baltic Sea was carried out by Turner *et al.* (1999). The authors simulated a 50 percent nitrogen and phosphorus reduction scenario. According to Turner *et al.* this corresponds approximately to nutrient levels of the Baltic Sea in the 1960s before its drastic deterioration. Cost effective policies for reducing nitrogen levels were found to include increased waste water treatment capacity at sewage treatment plants, reduction of use of nitrogen fertilisers, and construction of wetlands. Sewage treatment was proposed as a relatively low cost reduction option for reduction of phosphorous. On the other hand, benefits of waste management and of improvement of drinking water are geographically limited and high funding levels cannot be motivated by cross border benefits.

The costs of nutrient reductions were compared to the benefits. Two WTP surveys were carried out: one in Poland and the other in Sweden, asking the adult population in each country for their willingness to pay for a 20 year action plan to reduce eutrophication in the Baltic Sea. The action plan would be financed by introduction of an extra environmental tax. The willingness to pay estimates were transferred to the other countries around the Baltic Sea by adjusting WTP estimates to the levels of GDP per capita. The Polish values were transferred to the formerly planned economies and the Swedish values were transferred to the other countries: see Turner *et al.* (1999).

In order to achieve a better fit to current circumstances, WTP estimates for improving the status of the Baltic Sea have been updated and new estimates have been derived by using meta-regression analysis based on a large number of willingness to pay studies for improved water quality: see Huhtala *et al.* (2009). The authors found an average WTP of 60 euros per person and per year. This was then converted to country specific estimates by using country specific data on GDP per capita. The results are shown in Table 4. The population figures represent an estimate of the adult population in the Baltic Sea drainage basin of each country.

Table 4. Distribution of benefits between Baltic Sea countries based on meta-regression results, benefits in euros 2007

Country	Average annual WTP per person	Population (in millions)	Benefits per year (million euros)	Percentage of total benefits
Estonia	45.2	1.05	47	1.8%
Latvia	38.8	1.78	69	2.7%
Lithuania	40	2.42	97	3.8%
Denmark	71	3.58	254	9.9%
Finland	68	3.86	262	10.2%
Germany	66.2	2.45	162	6.3%
Poland	36.6	25.85	946	36.9%
Russia	33.5	7.00	235	9.2%
Sweden	72.6	6.78	492	19.2%
Total		54.77	2 564	100.0%

Source: Huhtala *et al.* (2009)

Clearly, the WTP estimates for sea water quality in the Baltic States are higher than the WTP estimates for water quality in river basin areas reported in Table 3. This is in line with the findings of Huhtala *et al.* (2009) who report that the type of water body is influential in determining willingness to pay values. If the affected water body is a sea area, willingness to pay is on average 31–42 euros higher than for other water bodies. Although improved waste water treatment represents only one of the measures for achieving better Baltic Sea water quality, the WTP estimates in the table suggest that the benefits are substantial for all countries that border the Baltic Sea.

5.2. Environmental Performance Index

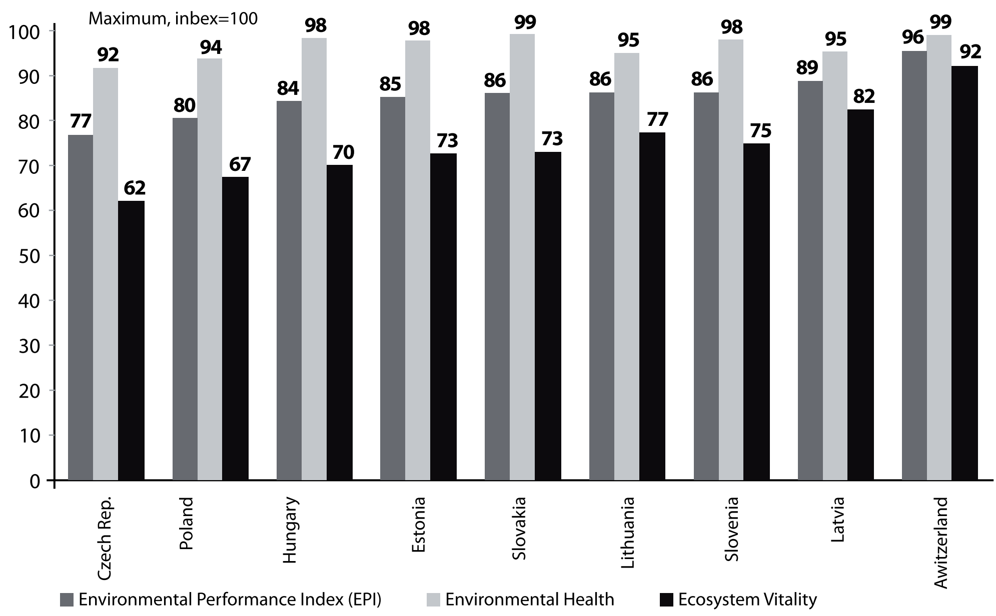
The efficiency criterion in terms of the cost-benefit rule carries information about the desirability of investment in a sustainability perspective. However, the problem of finding proper accounting prices necessitates collection of inputs from other sources about the state of the environment. For this reason and since available evidence of the cost-benefit rule is rather narrow, we have chosen an empirical source that allows cross country comparison about environmental status.

The Environmental Performance Index (EPI) gives input by assessing current national achievement towards environmental targets (see Esty, *et al.* 2008). The two overarching environmental objectives of EPI include: reducing environmental stress to human health (i.e. environmental health) and promoting ecosystem vitality and sound natural resource management. These objectives and the overall ranking of the eight Eastern European countries that joined the EU at the same time as Estonia, Latvia, and Lithuania plus Switzerland is shown in the figure below. The reason for including Switzerland is that this was the country with the highest EPI in 2008.

Latvia, with an index of 89, scores the highest value of the index among the East European countries that joined the EU together with the Baltic states. Lithuania and Estonia have 86 and 85 as index values, placing them top after Latvia and Slovenia. Based on Figure 1, it is also possible to conclude that the problems of environmental performance in Eastern Europe involve ecosystem vitality and sound natural resource management rather than environmental stress to human health (i.e. environmental health). This might be taken as an additional indication of over-investment in such priority themes as drinking water supply.

The ecosystem vitality index is further decomposed into four indicators. Figure 2 takes a closer look at this index of the Baltic States. Three of the indicators show the status of threats to ecosystems, such as water and air pollution and climate change, and the fourth indicator

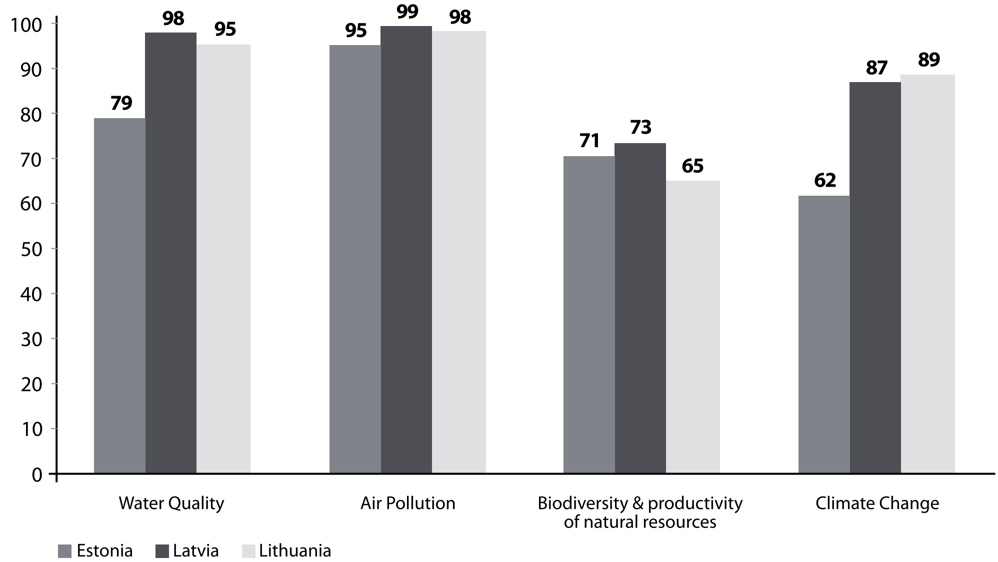
Figure 1. Environmental performance index of countries that became EU members in 2004 plus Switzerland, EPI 2008



Source: Esty *et al.* (2008)

measures the state of ecosystems including aspects such as species protection, forest, agricultural, and fishery re-productivity. The Baltic states score ranks lowest in biodiversity and productivity of natural resources. Estonia has a low position in climate change depending on large-scale use of oil shale in its energy sector.

Figure 2. Ecosystem vitality index decomposed into four fields (maximum index=100), Baltic States 2008.



Source: Esty *et al.* (2008)

The policy implications of EPI are that the Baltic states should pay more attention to biodiversity (e.g. conservation of habitats) and productivity of natural resources (e.g. fishery and cropland intensity). Both efficiency motivations and EPI thus suggest that more funds should be allocated for biodiversity and less for drinking water provision. In addition, EPI proposes that more attention is paid to enhancement of natural resource productivity, i.e. fisheries and cropland. In terms of priority themes, rehabilitation of industrial lands, promotion of biodiversity, and promotion of natural assets are put forward by EPI. Although water quality does not stand out as being at risk, productivity of fisheries needs attention, thus indicating the importance of upgrading waste water treatment.

6. Relevance to sustainability

The analysis in the previous section suggests some implications for sustainability of cohesion fund allocation to the environmental sector in the Baltic states in 2007-2013. However, it was not possible to include all priority themes in the analysis because of gaps in knowledge about the benefits and costs of several priority themes. Waste water treatment and promotion of biodiversity passed the cost-benefit rule. Management of household and industrial waste did not pass the criterion, although benefits were not far from covering costs. Drinking water supply was rejected.

EPI was used as a complementary input, and supported the scattered evidence of the cost-benefit rule. In addition, EPI suggested that rehabilitation of industrial lands, promotion of biodiversity, and promotion of natural assets would be important from an ecological point of view. In order to arrive at an overall assessment we will take the analysis one step further by classifying all priority themes under study into four fields (see matrix in Table 5).

Table 5. Classification of priority themes 2007-2013

Biodiversity and resource productivity	**Pollution control**
Rehabilitation of contaminated land Biodiversity and nature protection Promotion of natural assets Protection of natural heritage	Waste management Waste water treatment Air quality Integrated prevention and pollution control
Preventive measures	**Incidental environmental expenditure**
Mitigation and adaptation to climate change Risk prevention (plans and measures to prevent and manage natural and technological risks) Other measures to preserve the environment and reduce risks	Management and distribution of drinking water

Biodiversity enhancement in Estonia was supported by the cost-benefit rule. In addition, EPI further highlighted the need to promote biodiversity and resource productivity in the Baltic States. Four priority themes are classified as enhancing biodiversity and resource productivity and these will be classified as highly relevant for sustainability of the environmental sector.

The theoretical framework emphasized the long run perspective and proposed that sustainability concerns future generations into an infinite future. This long term perspective has so far not been highlighted by the analysis. Since preventive measures allocate funding to future environmental problems, this category will be classified as highly relevant to sustainability. Three priority themes are included among preventive measures: mitigation and adaptation to climate change, risk prevention, and other measures to reduce risks.

Four priority themes aim at reducing pollution. These include waste management, waste water treatment, air quality, and integrated pollution control. According to EPI, the status of water and air pollution is at satisfactory levels in Latvia and Lithuania. At the same time, WTP estimates for improving the water quality of the Baltic Sea show significant benefits. The level of funding of waste management did not pass the cost-benefit rule. These considerations imply that investment in pollution control can be considered as having medium to high relevance for sustainability.

The remaining expenditure is classified as incidental environmental expenditure. This classification follows Vincent *et al.* (2002) who classify incidental environmental expenditure as expenditure undertaken for non-environmental reasons. Drinking water infrastructure falls under this category. Neither the cost-benefit rule nor EPI suggests that drinking water is important from the perspective of sustainability. Incidental environmental expenditure is thus classified as having low relevance from a sustainability perspective. Table 6 below shows funding support from EU cohesion funds according to the four fields defined above.

The table shows that the Baltic States have allocated 10-24 percent of cohesion funds to biodiversity and resource productivity. Preventive measures receive 3-14 percent of funds. This implies that the two fields found to have high relevance for sustainability have been allocated less than half of the cohesion policy funding directed to the environmental sector. Latvia devotes least funds for investment in fields that will add most to sustainability (12.6 percent). Estonia allocates 37.1 percent and Lithuania 24.6 percent of environmental cohesion funding to these two fields.

Funds for reducing pollution were found to have medium to high relevance for sustainability. Pollution control receives more than half of the funds devoted to the environmental sector in Latvia and Lithuania and a little more than one third in Estonia. In per capita terms, funding is on a similar level in the three countries and will receive about 200 euros per capita in each country during the period 2007-2013. Incidental environmental expenditure, the field classified as having least relevance to sustainability, will receive between one quarter and one third of funding to the environmental sector.

It is evident that priority themes classified as having highest relevance to the sustainability perspective receive less funding than priority themes found to be of low and medium/high relevance to sustainability. Estonia, with its larger per capita contribution to the environment, also shows higher investment both in absolute terms and in percentages to fields highly relevant to sustainability. Lithuania, which ranks lowest according to its per capita funding, shows a better position than Latvia concerning allocation to fields highly relevant to sustainability.

Table 6. Cohesion funding for the environmental sector classified by relevance to sustainability, euros million current prices, euros per capita and percentages 2007-2013

Euro, million	**Estonia**	**Latvia**	**Lithuania**	**Total**
Biodiversity and resource productivity	184.2	75.0	180.7	439.9
Preventive measures	105.5	25.2	78.6	209.2
Pollution control	287.8	411.0	656.6	1,355.4
Incidental environmental expenditure	203.9	281.5	137.4	622.8
Total	781.3	792.7	1,053.4	2627.4
Euro, per capita				
Biodiversity and resource productivity	137	33	53	63
Preventive measures	79	11	23	30
Pollution control	214	180	194	193
Incidental environmental expenditure	152	123	41	89
Total	582	347	311	375
Percent				
Biodiversity and resource productivity	23.6%	9.5%	17.2%	16.7%
Preventive measures	13.5%	3.2%	7.5%	8.0%
Pollution control	36.8%	51.8%	62.3%	51.6%
Incidental environmental expenditure	26.1%	35.5%	13.0%	23.7%
Total	100.0%	100.0%	100.0%	100.0%

7. Conclusion

The purpose of this paper is to evaluate sustainability of investment plans of EU cohesion policy funds for the environment in Estonia, Latvia, and Lithuania during the budget period 2007-2013. Theoretical literature shows that the efficiency criterion, i.e. the cost-benefit rule, is applicable to identify sustainable investment. But since natural environments are complex, proper accounting prices may be hard to find when relying on human preferences. With these difficulties in mind, we applied a step-wise assessment to identify sustainability. Economic efficiency was considered by using the cost-benefit rule. In the second step we used the Environmental Performance Index (EPI) as a complementary indicator and to identify whether critical fields of investment in the environmental sector had been left out.

Use of the cost-benefit rule requires information on benefits and costs of planned investment. This information was only available for four out of twelve priority themes. Assuming that benefit transfer is possible between the Baltic states, available evidence suggests that too much funding is devoted to investment in drinking water infrastructure. Neither did management of household and industrial waste pass the cost-benefit rule, though benefits were not far from covering costs. Investment in sewerage services and waste water treatment were not possible to motivate unless benefits from environmental impacts on water bodies were included. Another implication is that investment in biodiversity protection could be extended since benefits significantly exceed costs.

The complementary input of EPI supported the scattered evidence of the cost-benefit rule. EPI showed that the Baltic States have no serious concerns related to environmental stress to human health, which might be taken as an additional indication that allocation of cohesion funds represents over-investment in drinking water infrastructure. The implication of the environmental performance index is that more attention should be paid to biodiversity (e.g. conservation of habitats) and productivity of natural resources (e.g. fishery and cropland intensity). Although water quality did not stand out as being at risk, productivity of fisheries was suggested by EPI to be at a low level, thus indicating the importance of upgrading waste water treatment.

Both steps of our analysis had similar implications, but neither was detailed enough to enable an assessment of all priority themes. In order to obtain an evaluation of all priority themes, we classified them into four fields. These fields were categorized according to their relevance to sustainability. The main finding is that the Baltic States allocate least investment to those fields of the environmental sector found to be most relevant to sustainability, i.e. preventive measures, and biodiversity and resource productivity.

Investment in drinking water was assessed as too large from the sustainability perspective. Having as an objective to reduce disparities between Member States, distributional considerations may have guided funding plans. It is not clear, though, how extensive investment in drinking water infrastructure promotes this purpose.

Another finding is that the three Baltic states, having large similarities concerning recent history, level of economic development, and natural environment, show significant differences concerning their priorities. Estonia has the highest per capita contribution to the environmental sector and also larger investment in fields with the highest relevance to sustainability. Lithuania ranks lowest according to its per capita funding, but shows a better position than Latvia concerning highly relevant fields. It is possible that Estonia's significant environmental problems stemming from oil shale based energy production have made the country more inclined than its Baltic neighbours to invest in the environmental sector and also more ready to direct investment into fields with high relevance to sustainability.

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Appendix 2. “Public Environmental Expenditure in Times of Crises in Estonia”

PUBLIC ENVIRONMENTAL EXPENDITURES IN TIME OF CRISIS IN ESTONIA

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Abstract

The purpose of this paper is to study the impact from the recent financial crisis on public environmental expenditure in Estonia. The data show different tendencies depending on the level of government. While the recent financial crisis has affected Local Government spending on environmental protection negatively, Central Government environmental protection expenditure increased by more than 30 percent between 2007 and 2008. Preliminary data indicate that this tendency continued in 2009. When comparing expenditures on environmental protection during times of crisis it is possible to detect differences between the developments in 1998-1999 and those in 2007-2008. Public expenditures on environmental protection were much more sensitive to declining GDP during the previous financial crisis than during the recent crisis. In the 2000s two important changes have affected environmental funding in Estonia. Accession to the EU in 2004 has made EU funding available for environmental protection. The ecological tax reform introduced in 2005 has increased the revenues of environmental charges earmarked for environmental purposes.

Keywords: public environmental expenditure, financial crisis

JEL Classification: H59, Q5, Q28, Q58, H72

1. Introduction

The recent financial crisis has resulted in major cuts of public expenditure in Estonia. In order to bring the state budget into balance, the Estonian Government reduced state budget expenditures by 3.4 percent in 2008 (Ministry of Finance 2008). These cuts affected the Ministry of Environment by 7.5 percent. In 2009, falling tax revenues called for further adjustments.

Public expenditure management is an important aspect of a country's environmental policy. Major budget cuts in times of crisis might jeopardize attainment of critical long run objectives. Observations during the Asian financial crisis suggest that public environmental expenditure is more sensitive to cuts in public expenditure during periods of crisis than other public expenditure (Vincent 2002). Vincent and his co-authors find that environmental expenditures in Indonesia declined much more than budget cuts on average. A comparison to other Asian countries showed that environmental expenditures declined much more in Indonesia than they did in Malaysia, Thailand and Korea during the same time period. However, the article does not discuss potential reasons for these differences nor does it make any comparisons of environmental policies.

The study by Vincent and his co-authors represent one of several World Bank reviews about public environmental expenditure. The Public Environmental Expenditure Reviews or PEERs have had a wide variety of purposes including measuring the impacts of a financial crisis, preparing a ministry for budget cuts, tracking funds, and determining future resource requirements (Swanson, Lundethors 2003). However, a low level of public environmental spending is not in itself an argument for more expenditure. Studies on transition economies in Central Europe, as well as observations of environmental finance in Turkey, conclude that it is not primarily the lack of financing that limits environmental recovery; it is rather weak institutional capacity and unclear priorities that hamper environmental spending (Prekzko, Zylicz 1998; Sezer 2003).

Estonia has prioritized budget balance since independence and has succeeded much better than most other CEEE countries in keeping government budgets under control and reaching fiscal sustainability (Aristovnik, Bercic 2007). This position makes Estonian environmental expenditure in times of crises an interesting case study. The purpose of this paper is to follow the development of public environmental expenditure in Estonia and study the impact of financial crisis. Another purpose is to link developments to funding principles of the Estonian environmental policy.

We begin by describing the framework of funding of environmental policy in Estonia and after that we review data sources of environmental expenditure. Then a presentation of developments of public environmental expenditure during the time period 1995-2009 follows. Special attention is devoted to two periods of crisis 1998-1999 and 2008-2009. After that we discuss the results and present conclusions based on the observed developments.

2. Framework of Environmental Taxes and Charges

Estonia has used economic instruments for environmental protection since early 1990s. The principal legislation that regulates environment taxes and charges in Estonia include the "Alcohol, Tobacco, Fuel and Electricity Excise Duty Act" (RT I 2007, 45, 319), the "Packaging Excise Duty Act" (RT I 1997, 5/6, 31) and the "Environmental Charges Act" (RT I 1999, 24, 361). A specific feature of the environmental taxes is that these accrue to the state budget for financing the general needs of the state. The packaging excise duty is an exception though, since 50 percent of revenues must be used for environmental protection and the remaining 50 percent is available for general needs. The proceeds from environmental charges are earmarked for environmental protection. Other sources of financing for environmental protection include the European Union funds, guided by the Estonian National Development Plan for the Implementation of the European Union Structural Funds − Single Programming Document 2004-2006, and the Operational Programme for the Development of the Living Environment for the years 2007-2013.

The environmental taxes in use are excise duties on fuel and packaging, and heavy goods vehicle tax. In 2008, an excise duty was imposed on electricity. Unlike many other countries, there is no separate vehicle tax on passenger cars. In 2008, environmental taxes contributed 7 percent of the state tax revenue (approximately 5 billion EEK) (Keskkonnaülevaade 2009).

The most important source for the accomplishment of environmental policy objectives and implementation of the "polluter pays" principle are environmental charges. The purpose of the environmental charges is to prevent or reduce the possible damage related to the use of natural resources, emission of pollutants into the environment and waste disposal. Environmental charges are paid into the state budget where they are allocated for maintaining the state of environment, restoration of natural resources and remedying of environmental damage. A part of the environmental charges are paid into the local government budgets where they are used according to local needs (not necessarily for environmental purposes).The environmental charges paid into the state budget contributed approximately 1.5 percent of total tax revenue in 2008 (Keskkonnaülevaade 2009). The pollution charge was the most important revenue source, contributing about 1.3 percent in 2008. In the years prior to the ecological tax reform pollution charges contributed about 1 percent of total tax revenue.

There are two different types of environmental charges: the natural resource charge and the pollution charge. The pollution charge is levied on emissions of pollutants into the ambient air, water bodies, groundwater or soil, and on waste disposal. The natural resource charge in turn is divided into: the forest stand cutting charge, mineral resources extraction charge, water abstraction charge, fishing charge and hunting charge.

Since 1994, over 6 billion EEK have been paid for pollution, extraction of mineral resources and water abstraction charges (Keskkonnaülevaade 2009). About 76 percent (ca 4.6 billion) have been paid into the state budget and the rest into local government budgets. Environmental charge rates were initially set very low, considering the ability to pay of the population and for promotion of economic development.

With the economic advancement it has become possible to pay more attention to environmental protection. Already in 1996, the annual pollution charge rates were raised by 20 percent and the annual natural resource charge rate by 5-10 percent. In 2005, the Government decided to introduce an ecological tax reform. The key principle of an ecological tax reform concept is to increase the use of environmental taxes and reduce the burden on employment related taxes (income or social taxes). One of the aims of Estonian ecological tax reform is also that the overall tax burden (ratio to GDP) would not increase. As a first step personal income tax was lowered from 26 to 24 percent in 2005. All main environmental charges were raised substantially in 2006. By following the logic of the ecological tax reform the increase in charges was induced by the need to make economic instruments more effective and give producers and the general public a clear signal that Estonia wants to use its natural resources and environment in a sustainable way. The level of charges continued to increase in 2007 through 2009.

Major payers of environmental charges are enterprises with substantial environmental effects – oil-shale industry companies, chemical and paper industry, water supply and waster disposal enterprises, enterprises extracting and processing mineral resources. In 2007, ten major natural resource users paid 80 percent of the charges (Keskkonnaülevaade 2009).

2.1. Financing Environmental Measures

Environmental charges have been an important source for financing the renovation of sewage disposal plants, investments into pollution abatement equipment and environmentally adapted waste disposal sites. Funds paid into the state budget for using natural resources are used according to the Environmental Charges Act through the Environmental Investment Centre (EIC) to promote environment protection. EIC's environmental programme is the main national measure for financing environment protection. The fields supported by the EIC programme include water management, waste management, nature conservation, forestry, fishery and environmental awareness.[1]

In total, 3.4 billion EEK were paid out under the environmental programme during 2000-2008. As the European Union has established strict fixed-term requirements for the quality of drinking water, purification equipment and sewage systems, most of the proceeds from environmental charges have been used for bringing the water supply into conformity with the requirements. Significant contributions have been made also into fulfilling the requirements established for waste treatment and disposal. Approximately 2 billion EEK in total were given through the environmental programme for the development of water supply and waste disposal infrastructure in 2000-2008 (Keskkonnaülevaade 2009). This amount was increased by the recipient's own contribution.

An important source of finance of environmental investment in addition to the environmental programme is foreign aid. In 2005-2008, Estonia received approximately 2 billion EKK worth of foreign aid for the development of environment protection infrastructure and environment protection activities (Keskkonnaülevaade 2009). The aid was received mainly from the EU Cohesion Fund, and three thirds (or 1.5 billion) were used in water supply for various investments for the improvement of the quality of drinking water and organization of sewage disposal and purification.

3. Data on Public Environmental Expenditure

Statistics Estonia produces data on general government revenues and expenditures. The data set is available for the time period 1995-2008 (www.stat.ee) and is classified according to the United Nations Classification of the Functions of Government (COFOG)[2]. One of these government functions is environmental

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[1] http://www.kik.ee/?op=body&id=105
[2] http://unstats.un.org/unsd/cr/registry/regcst.asp?Cl=4

protection and covers activities that reduce negative externalities. The definition of environmental protection set by OECD and Eurostat includes "activities aimed directly at the prevention, reduction and elimination of pollution or any other degradation of the environment resulting from the production processes or from the use of goods and services expenditure on waste management, waste water treatment, pollution control, protection of biodiversity and landscapes, and other environmental protection activities" (Swanson, Lundethors 2003). Environmental protection is broken down into six sub-categories:

- Waste management
- Waste water management
- Pollution abatement
- Protection of biodiversity and landscape
- Research and Development (R&D)
- Other environmental protection expenditures

These data make it possible to follow the Central Government and Local Government expenditure on environmental protection and distribution by domain during 14 years.

Specification of investments into and current expenditure on environmental protection can be followed in another time series. These data are available for the time period 2001-2008 and for Local Governments only. Statistics Estonia collects information in a survey following SERIEE classification, which is similar to the COFOG system, but provides codes in greater detail. In addition, Local Governments are asked to allocate activities covering more than one code by specifying percentages. The COFOG system allocation is based on the majority principle implying that investments covering two fields will be categorized according to the major field of expenditures (Salu 2009). Even on aggregate level there might be discrepancies between these two sources when they cover more than one group of government functions. This is the case of waste water management included in government function of environmental protection and water supply, which is classified as the government function of housing and community amenities.

Since the purpose is to study the development of public environmental expenditure during the financial crisis, we are interested in covering latest developments. However, data for 2009 are not yet available. In order to assess most recent developments, preliminary budget data for 2009 have been collected from the Ministry of Finance. Another difficulty is that Estonia receives foreign aid for environmental protection purposes, which makes it is difficult to detect "pure" public sector expenditure on the environment. In order to give an approximate estimate, we present assessments for certain years in our time series. The time series for environmental protection expenditures are presented at constant prices using the GDP deflator.

4. Budget Cuts in Time of Crisis

Central Government budget expenditure for environmental protection was 1,450 million EEK at current prices in 2007. During the first year of crisis, in 2008, expenditure increased to 2,083 million EEK at current prices. In terms of Central Government budget expenditure, environmental protection was 2.3 and 2.8 percent respectively. Figure 1 shows the development of state budget expenditure, GDP and state budget expenditure on environmental protection during the time period 1995-2009.

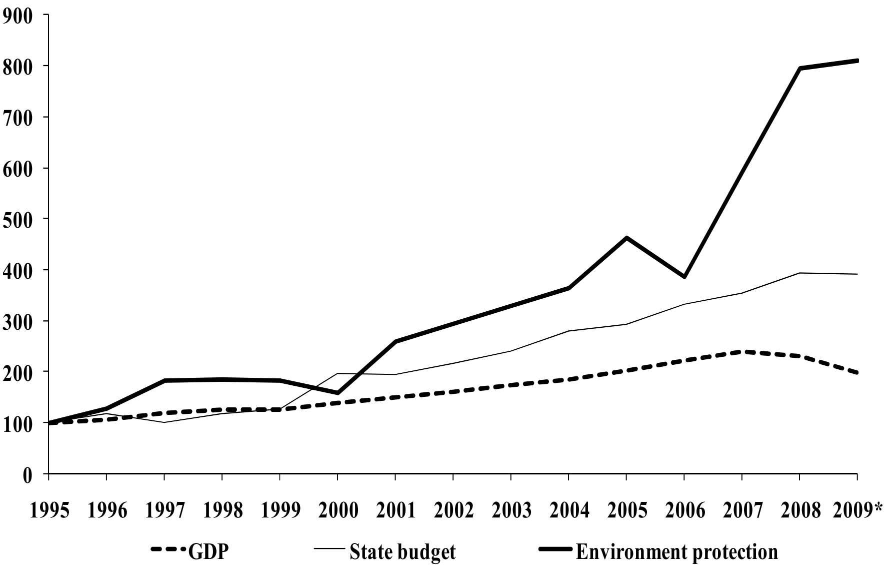

Figure 1. Gross domestic product, central government expenditure and central government expenditure on environmental protection, volume (constant prices), 1995 Index=100. (Authors' calculations, Statistics Estonia*, Bank of Estonia and Ministry of Finance)

The figure shows that expenditures on the environment have grown significantly during the past few years. There have been reductions in spending on environmental protection, but these cut-downs occurred in 2000 and 2006. Between 2008 and 2009 there was a small positive increase in expenditure on environmental protection, while total state budget expenditures remained on the same level as a year before. Data thus suggest that Estonian state environmental expenditures have not suffered from budget cuts during the recent financial crisis.

Local Government expenditure on environmental protection was 907 million EEK in 2007. This means that Local Government expenditures were about 40 percent smaller than Central Government expenditures on environmental protection. As is the case of the Central Government, Local Government expenditures increased in 2008 and totaled 945 million EEK. However, at constant prices, Local Governments decreased spending on environmental protection during the first year of crisis. The share of environmental protection expenditures was 3.8 percent in 2007 and 3.4 percent of Local Government expenditures in 2008. Figure 2 shows the development of Local Government expenditure in total and Local Government expenditure on environmental protection at constant prices during the time period 1995-2009.

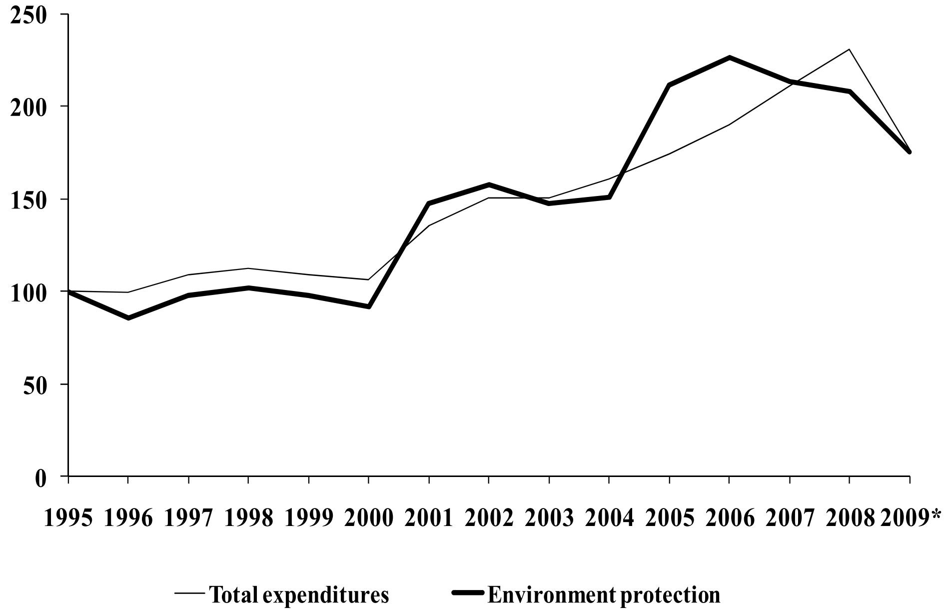

Figure 2. Local government expenditure in total and expenditure on environmental protection, volume (constant prices), 1995 Index=100. (Authors' calculations, Statistics Estonia and Ministry of Finance*)

The figure shows that expenditures on environmental protection have grown significantly during the time period 1995-2009. Since 2000, developments have been cyclic, with one peak in 2002 and another peak in 2006. The peak in 2006 coincides with a reduction in Central Government expenditure and with the introduction of the ecological tax reform. The growth in environmental protection expenditure is well correlated with total budget expenditure and in contrast to the central budget, local budget expenditures on environmental protection have declined over the past few years. This decline continued in 2009. Environmental protection expenditure decreased at a similar pace as the Local Government expenditure between 2008 and 2009. The significant correlation between Local Government budget expenditure and expenditures on environmental protection is probably related to Local Government responsibility for waste management and waste water treatment. To some degree these activities are financed by tariffs and when incomes decrease so do expenditures.

The recent financial crisis is the most severe, but not the only economic crisis that has hit Estonia since independence. The available time series covers the economic crises of 1998, which resulted in negative growth records in 1999. Table 1 shows annual change in GDP, annual percentage change of expenditure on environmental protection during the time period 1996-2009.

The year-to-year changes in expenditures on environmental protection have fluctuated significantly during the time period under study. When comparing expenditures on environmental protection during times of crisis it is possible to detect differences between the developments in 1998-1999 and those in 2007-2008. The Central Government expenditures on environmental protection were more sensitive to declining GDP during the previous economic crisis than during the recent financial crisis. In addition, Central Government expenditure on environmental protection continued to contract in 2000 when the economy had recovered. It is difficult to predict the timing of recovery from the current crisis, but preliminary data on 2009 and the state budget of 2010 suggest further expansion of expenditure to the Ministry of Environment. According to the Ministry of Finance, growing expenditures are based on increases in EU funding (Ministry of Finance 2010). The direction of the development of Local Government expenditures on environmental protection has, on the other hand, been sensitive to budget cuts during the two crises.

Table 1. Annual percentage change of GDP, annual percentage change of expenditure on environmental protection at Central and Local Government in constant prices.

	GDP	Expenditure on environmental protection	
		Central Government	Local Government
1996	5.7%	27.9%	-14.3%
1997	11.7%	42.6%	13.9%
1998	6.7%	1.5%	4.5%
1999	-0.3%	-1.6%	-4.4%
2000	10.0%	-13.3%	-6.0%
2001	7.5%	64.4%	60.7%
2002	7.9%	13.1%	7.0%
2003	7.6%	11.9%	-6.6%
2004	7.2%	10.9%	2.1%
2005	9.4%	26.7%	40.3%
2006	10.0%	-16.4%	7.1%
2007	7.2%	53.1%	-5.8%
2008	-3.6%	34.7%	-2.4%
2009*	-14.2%	1.8%	-15.9%

Source: Authors' calculations, Statistics Estonia, Bank of Estonia and Ministry of Finance. Preliminary data*.

Looking at a possible link between GDP growth and change in environmental protection expenditures suggests that impacts differ between Central and Local Governments. In six years out of fourteen, the direction of the year-to-year changes in expenditure on environmental protection differs for the two levels of government. The Central Government expenditures on environmental protection have grown more than GDP during ten years, while the same is true for Local Government expenditure only during four years. However, those years that GDP has grown at least 9 percent there has been a two digit growth in environmental expenditure in both levels of government during three single years: 1997, 2001 and 2005. On the other hand, a high level of economic growth does not seem necessary for growing expenditure on environmental protection (see Central Government expenditure on environmental protection in 2006).

The different trends in expenditure on environmental protection between the central and local levels since 1995 can also be detected by looking at the environmental protection shares of budget expenditure (see Figure 3). While environmental protection expenditures have grown significantly as a share of Central Government expenditures from about 1 percent in 1995 to 2.8 percent in 2008, Local Government budget expenditures on environmental protection have been on a constant level of about 4 percent during the whole time period 1995-2008.

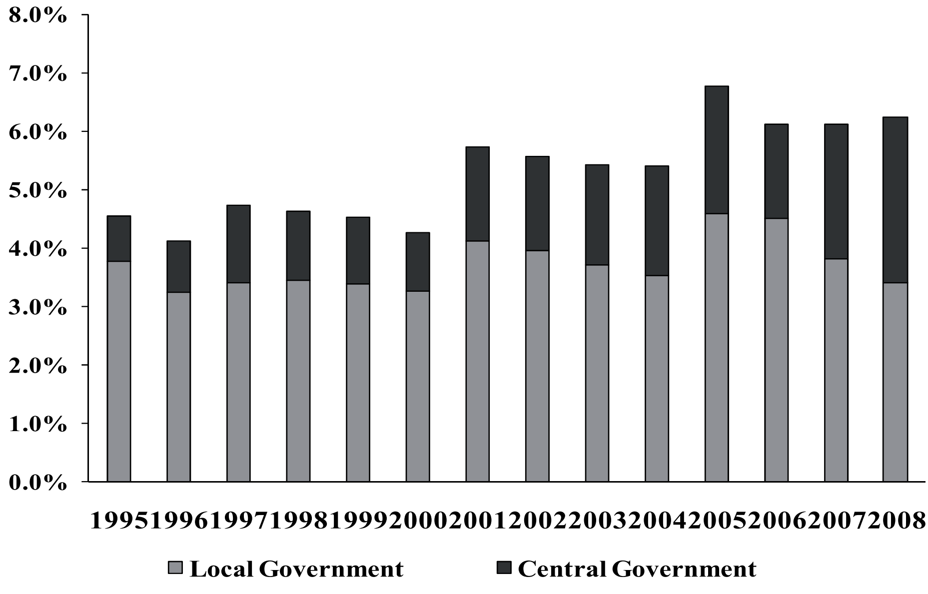

Figure 3. Environmental expenditure as percentage of total expenditures of local and central budget expenditures, 1995-2008. (Authors' calculations and Statistics Estonia)

4.1. Environmental protection expenditure by domain

Government expenditures on environmental protection can be followed up by domain. While Local Governments made small adjustments, including cuts in waste water management, and pollution abatement expenditures in 2008, the Central Government increased its expenditures on waste water management and on protection of biodiversity and landscape. During the previous economic crisis, Central Government expenditure declined on waste management, waste water management and pollution abatement. Local Governments reduced all environmental protection spending except expenditures on waste management between 1998 and 1999.

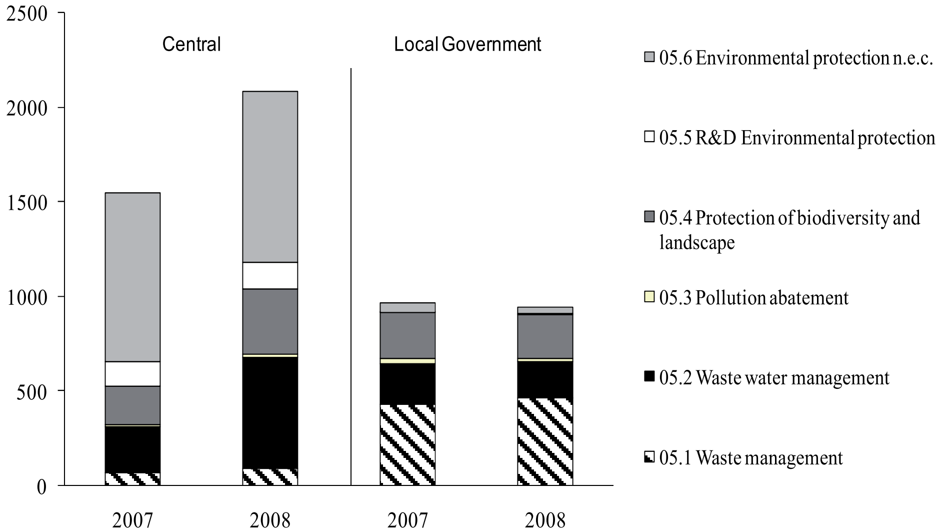

Figure 4. Environmental expenditure by function, local and central budget expenditures, 2007-2008. Constant prices (price level 2008), million EEK. (Authors' calculations and Statistics Estonia)

4.2. Investments

The available data on investments cover only Local Governments and show that spending has grown over time, but do not reveal any specific trend in terms of investments or current expenditures. During the time period 2001-2008, between 40 and 60 percent of Local Government expenditures on environmental protection concerns investments. From the beginning of 2000s until the end of the decade the focus has shifted from a dominance of waste water investments to an increasing share of waste management investments.

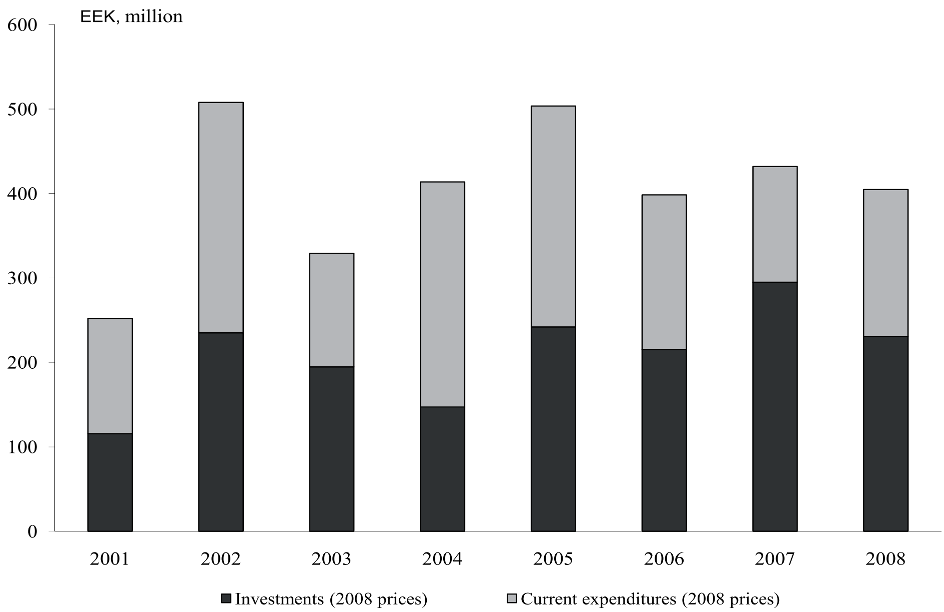

Figure 5. Local Government budget expenditures on environmental protection SEIREE method (prices 2008), million EEK. (Authors' calculations and Statistics Estonia)

4.3. Foreign Aid

There is no comprehensive data set covering foreign aid payments to environmental protection expenditure in Estonia. Generally foreign aid only includes investments. According to gross estimates, foreign aid has made up the lion part of government investments into waste water treatment and to waste management. The share of foreign aid in the State Investment Programme on environmental investments varied between 50 and 60 percent during the time period 2002-2004 (Statistikaamet different years). In 2005-200, foreign aid made up 40-50 percent of environmental investments in the state sector (Keskkonnaülevaade 2009). According to this source, foreign aid increased from about 600 million EEK in 2007 to about 700 million EEK in 2008. At constant prices, this corresponds to an increase of about 35 percent. Expenditures for co-financing environmental investments more than doubled − from 143 to 395 million EEK between 2007 and 2008. Since this source includes investments into the water supply system it does not exactly correspond to the earlier data set. The indication though is that foreign aid probably was an important driving force of the observed increase in environmental protection expenditure between 2007 and 2008.

In an overview about the use of environmental charges in Local Governments, Salu collected information about EU funding for environmental protection purposes (Salu 2009). The results indicate that between 4 and 16 percent of environmental protection expenditures of Local Governments were financed by various EU funds during the time period 2001-2007. However, Salu's data did not cover LIFE and

INTERREG programmes. Data for recent years and developments during the financial crisis period have not been possible to access.

5. Conclusions

Data on public environmental expenditure show that the recent financial crisis has decreased Local Government spending on environmental protection, while this is not the case of the Central Government. Between 2007 and 2008, Central Government expenditure increased by more than 30 percent while Local Governments cut down their expenditure by 2.4 percent. Preliminary data indicate that this tendency has continued in 2009. When comparing expenditures on environmental protection during times of crisis it is possible to detect differences between the developments in 1998-1999 and those in 2007-2008. Public expenditures on environmental protection during the previous financial crisis were much more sensitive to declining GDP than during the recent crisis.

Another finding is that environmental spending of Local Governments is closely correlated to total budget expenditure. The expenditures on environmental protection have been on a constant level of the total budget of about 4 percent during the time period 1995-2008. The level environmental spending of the Central Government is not equally sensitive to total budget expenditures and their share of total Central Government expenditures grew from about 1 percent in 1995 to 2.8 percent in 2008.

In the 2000s two important changes have affected environmental funding in Estonia. Accession to the EU in 2004 has made EU funding available for environmental protection. In addition, the ecological tax reform has increased the revenues of environmental charges earmarked for environmental purposes.

The environmental policy aims of Estonia as a small member state of the European Union are closely interlinked with the respective ambitions of the EU, having been fixed in EU directives and other regulations. The European Union has decisively committed to ensuring environment protective development.

The ecological tax reform that shifts tax burden from negative taxes for welfare (e.g. employment related taxes) to positive taxes for welfare (e.g. taxes on activities that damage the environment, such as exploitation of natural resources or pollution) is necessary to contribute to solving environment related problems. At the same time, a long-term change in taxation presumes relatively stable income from the environment related tax base.

Estonia has in general fulfilled the environment related tax base stability condition due to the framework of environmental taxes and charges that are periodically adjusted. Environmental taxes and charges, according to law earmarked for financing environmental expenditure (a certain share of pollution and resources taxes goes to the Estonian Environmental Investment Fund), have allowed a relative independence of environmental spending from macroeconomic conjuncture. For example, the environmental tax rates were raised 20 percent in 2009 on request of the Green Party, despite the economic recession.

Both central government and local sector expenditure on environmental protection in Estonia have regularly increased over the period discussed in the paper and have stayed relatively stable and independent from the fluctuations in the gross domestic product. Particularly remarkable has been the increase in environmental expenditure since 2007, which can be explained by opening of the EU Cohesion Fund resources for the budget period 2007-2013. Remarkable finance of environmental activities (above all sewage and waste disposal) from EU structural funds also explains the growth of government sector environmental expenditure in the period when the gross domestic product declined.

To sum up, the growth and stability of environmental expenditure in Estonia are based on a carefully though out and regularly adjusted system of environmental taxes and charges, and the allocation of the tax proceeds for environmental expenditure is provided by law. Local and central government sector expenditures on the environment are increased by a significant amount of foreign aid from the EU structural funds, which are used mainly for water supply and waste disposal related environmental investments, as well as for nature protection expenditure. As a co-effect of various measures, Estonia has managed to preserve stability of environmental expenditure.

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Appendix 3. “Subsidising Renewable Electricity in Estonia”

Courtesy of WIT Press from the book Energy and Sustainability III edited by Villacampa Esteve, Y., Mammoli, A.A. and Brebbia, C.A., 2011, page 197

Subsidising renewable electricity in Estonia

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Abstract

The purpose of this paper is to assess the impact of Estonia's feed-in tariffs (FIT) on combined heat and power (CHP) plants. The assessment follows previous practice and provides a novel approach by including a case study based on company data. The results of our assessment show that the Estonian FIT system has effectively supported the establishment of CHP capacity and that the administrative costs have been low. In contrast to experiences in other countries we find that the avoided external costs exceed the per MWh cost of FIT. Another feature is that the consumer costs of the FIT scheme have grown more rapidly than elsewhere. Although avoided external costs cover FIT, resources are not used cost-effectively. The case study of two CHP plants suggests that resources are used for supporting production that would have been profitable without FIT.
Keywords: renewable electricity, feed-in tariffs, CHP, energy policy, Estonia.

1 Introduction

Feed-in tariffs (FIT) is the most widely used support scheme for renewable electricity: implemented in 20 EU countries and 30 countries worldwide in 2009 [1]. Denmark and Germany were the first countries to introduce FIT in the mid-1980s and 1991, respectively [2]. Success stories about countries that have exceeded initial goals for renewable electricity seem to be forceful arguments for additional implementation. Further backing from economists supporting the use of price rather than quantity based regulation could be another reason for the popularity of FIT.

According to the national electricity development plan 2005–2015 [3] the goal is to increase the share of renewable electricity to 5.1% of gross consumption in Estonia by 2010. In the succeeding development plan, which stretches until 2018, the goal has been set to extend the share of electricity from renewable resources to 15% by 2015 [3, 4]. For Estonia, these goals imply

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significant changes. In 2007, the share of renewable fuels in electricity production was 1.75% of gross production while the main supply originated from oil shale electricity, which made up 93.6% [4]. Based on capacity under construction, it is estimated that Estonia outperforms the goal in 2010and reaches 9.7% renewable electricity [5].

Estonia's goal to 2020 is to increase electricity produced from renewables in combined heat and power plants (CHP) to 20% of gross production [4]. Following introduction of FIT in 2007, there has been a substantial increase in energy produced from renewable fuels in CHP plants. In 2009 Tallinn and Tartu CHP started operation and the share of renewable electricity is further increasing. Pärnu CHP is under construction and several small CHPs are being planned in different parts of Estonia. Recently, also oil shale electricity producers have begun to use biomass as an input. It seems thus that Estonia shares the experiences of other countries that report a rapid increase of renewable electricity following introduction of FIT [2, 6].

Besides the positive effects, the change seems to have come at a high cost. The costs of FITs have increased from 6 million to almost 55 million Euros between 2007 and 2011 [5]. This cost is collectively paid by consumers by an addition to the price of electricity. In 2010, this addition makes up about 10 percent of the consumer price and the Estonian Competition Authority, who regulates the price of electricity, has questioned the size of the subsidy [7].

The purpose of this paper is to assess Estonia's FIT scheme on CHP plants. Assessments have been carried out by several other authors, see [6] for references. The goal of this paper is to assess whether the current tariff level paid to CHP plants is motivated from an efficiency perspective, and its implications on consumer costs. Another aim is to find the benefits in terms of avoided external costs. The authors are not aware of previous assessments concerning CHP plants, suggesting that this paper may represent the first assessment of FIT on CHP plants. In addition, the case study of this paper applies a novel approach by using company level data.

The next section provides a literary overview about FIT assessments. In section 3, we give details about the Estonian FIT. Section 4 presents calculations that assess the company level impact of FIT on two CHP plants and compares the outcome to marginal cost and cost price. In section 5 we calculate the external costs of electricity produced from oil shale and compare this with electricity produced by biomass and peat in CHP plants. Section 6 summarizes the assessment and the last section concludes the paper.

2 Literary review

A feed-in-tariff (FIT) denotes a guaranteed price to producers of electricity generated from renewable sources, combined with a purchase obligation by grid companies [6]. There principally are two different ways to cover the costs of the policy measure, either by consumers via the electricity bill or via the public budget. An important reason to subsidise renewable electricity is that production costs typically are higher than that of non-renewable electricity [6]. In this sense

FITs represent a second-best policy by giving a subsidy to a preferred choice rather than correcting for external costs of electricity from non-renewable sources. Not only the choice of which market to regulate, but also the FIT levels have been questioned. In an overview of support schemes in 2005, it was shown that German support levels typically were twice the level of those of the Nordic countries, mainly using quantity based regulation combined with green certificates [8]. The same study indicated that the costs of FITs on the margin cannot be motivated by the social benefits from renewable electricity [8, 9]. At the same time, there seems to be efficiency arguments to use FIT for wind power [1]. Most probably these efficiency reasons denote dynamic efficiency in order to provide technology change and support market take-off [6].

Based on German and Danish experiences, Sijm [2] has assessed the sustainability of feed-in tariffs. The German FITs were until 2000 based on a percentage of earlier consumer prices of electricity and varied by the source of energy. After implementation prices rose significantly and due to a rapid expansion of wind power, the system led to competitive distortions between grid companies in different parts of the country. When the German market for electricity was liberated, the system needed urgent revision. The new FITs are based on the production costs of various renewable energy resources with digressive payments during 20 years [2, 10]. Denmark revised its FIT in 2000 for reasons of a high burden on the state budget [2]. In his assessment of FITs, Sijm [2] concludes that FITs are effective in promoting electricity generation from renewable sources, but costly, inefficient and distortive.

Spain is another country that has been successful in renewable energy promotion. In their assessment del Rio and Gual [6] find that the Spanish system has been effective in its support of wind energy, but not equally successful concerning other energy sources. They conclude that although consumer costs were relatively low, increasing from 0.14 to 0.26 eurocents /kWh between 1999 and 2003, the costs are relatively high compared to the externalities avoided.

3 Feed-in tariffs in Estonia

According to the Estonian Electricity Market Act production of electricity from wind, small hydropower and biomass receive the same level of FIT [11, 12]. The FIT for CHP plants differs according to fuel. Generating electricity in efficient cogeneration regime by biomass (wood chips), the producer is paid support at the rate of 54 €/MWh for selling electricity to the network. While operating in efficient cogeneration regime and using waste or peat as a fuel, the producer is paid support at the rate of 32 €/MWh. If wood chips, peat, waste or other fuels are combined, the support granted for selling electricity to the network is calculated in proportion to the fuel used. The FIT schemes apply within twelve years as of the commencement of electricity generation.

After introduction of FIT on May 1st in 2007, the expenses for financing FIT are funded by network charges paid by consumers. In 2010 the renewable energy charge is 0.8 € cents/kWh. An additional line setting out the renewable energy charge was added to the electricity bills of end users enabling customers to see how much they pay for financing feed-in tariffs.

The Estonian electricity market is divided into two – an open market and a closed market. 35% of the market was opened on 1 April 2010. Starting from 2013, the market is going to be fully liberated. While selling electricity in the closed market, approval must be obtained under the law [11] according to the weighted average price limit of electricity. In its approval, the Estonian Competition Authority takes into account operating expenses and returns on invested capital. In order to determine the price, the authority considers the undertaking's annual average residual value of fixed assets and adds 5% as profit margin. The justified rate of return is the undertaking's weighted average cost of capital (WACC).

4 The impact of FIT on CHP plants

The case study takes as its starting point, two 25 MW_{el} CHP plants that began operations in 2009. The evaluation of the investment decision and profitability of the CHP plants are based on annual reports [13, 14]. In order to assess profitability without FIT, we apply the rules of the Estonian Competition Authority and we calculate the per MWh revenue without FIT. The results are then compared to marginal cost and the cost price of electricity (the cost price is the price that exactly balances production costs, not adding profit).

4.1 Ratio analysis

The annual reports consist of the balance sheet, income statement and notes on the accounts. The methodological approach used in the evaluation of the financial reporting is based on ratio analysis, carried out as comparison with accounting benchmarks. Ratio analysis is the main instrument in financial analysis that enables to elicit relations between financial indicators and compare different undertakings with one another.

The investments in the plants were of the same order of magnitude, i.e. approximately 77 M€ respectively. Although no trend analysis can be made on the basis of the publicly available financial results for 2009 of the CHP plants, the data still allow evaluating, in general lines, plant profitability in 2009. Results of the evaluation are displayed in Table 1.

Table 1: Ratio analysis of two CHP plants, 2009.

Ratio	Bench-mark	CHP 1		CHP 2	
		Ratio	Evaluation	Ratio	Evaluation
Net profit margin	5.0%	37.6%	High	10.3%	High
Operating profit margin	17.0%	48.1%	High	23.9%	High
Rate of return on equity capital	15.0%	100.0%	High	34.5%	High
Rate of return on assets	9.0%	11.8%	Normal	2.4%	Weak
Debt coefficient	40.0%	88.2%	High (risk)	93.0%	High (risk)

The table shows that the power plants' rate of return on equity capital is high indicating efficient management in using the capital invested by shareholders. Profit margin that characterises profit on every euro of turnover is also high. The debt coefficient, pointing at how big a proportion of total funds are financed from borrowed funds, is extremely high in both plants. The profitability of assets shows the rate of return on the funds invested in the company irrespective of their source. Profitability is weak in CHP 2 being approximately 5 times lower that of CHP 1.

It can be concluded from the above that due to the implementation of FIT, the new power plants have managed to start profitable economic activity. Despite a large debt burden and strong dependence on borrowed capital, the rate of return on equity capital and the net profit margin hint at management efficiency and ability to gain initial results in activity.

However, case study data covers only one year. Additional sources of uncertainty include the development of prices of renewable fuels and the impact of market liberalisation. Notwithstanding these uncertainties, there are reasons to believe that the plants will continue operations successfully. It is possible to argue that these plants are well prepared to meet changes in input prices. In case of a rapid price increase, there is flexibility to shift fuels. Both plants are licenced to use wood chip and peat as fuel. Boiler technology allows additional fuels and the plants have fuel producing companies as subsidiaries. While market liberalisation will take place on electricity sales, the profitability of heat production can be predicted to be stable due to the continuation of a closed heat market. Since electricity prices in the Estonian market currently are below Nordic spot market prices [15], market liberalisation is expected to lead to price increases.

In theoretical terms, each power plant could generate a maximum of 25 MW * 7200 h=180 GWh of electricity per year. The generated volume of electricity depends on the number of operational hours. A smaller number of stop pages and standstill periods imply more operational hours and more generated electricity.

Pursuant to the actual annual report of 2009, CHP 1 generated circa 128 GWh and CHP 2 generated circa 110 GWh of power. Electricity generation in the plants were in the range of 68%-80% of the theoretical maximum. In CHP 1 the size of support comprised 54 €/MWh * 128 * 10^3 MWh ≈ 6.9 M€. Since CHP 2 used peat, the support size was32 €/MWh * 110 * 10^3 MWh ≈ 3.5 M€. Regarding different plants, FIT revenue accounts for approximately 50–60% of the operating profit, and excluding FIT comprise approximately 40–50%. Dependence of operating profit on the size of FIT can be expressed by eqn. (1).

$$\pi_{el} = \pi_{el}^{excl\,FIT} + (Q_{el} \times FIT_i) \tag{1}$$

where π_{el} denotes operating profit on electricity sales, $\pi_{el}^{excl\,FIT}$ operating profit on electricity sales excluding FIT, Q_{el} generated electricity and FIT_i feed-in tariff for *i*=1,2 (1=wood chip and 2=peat). According to the annual report, operating profit on the electricity sales of CHP 1 amounted to circa 12 M€; excluding FIT, operating profit would be 5.1 M€. The respective sums for CHP 2 are circa 7.5 M€ and 4 M€. These results suggest that the operating profits of both plants would have been positive also without FIT.

4.2 WACC

Assuming that the plants had operated without FIT and that their electricity prices were set by the Estonian Competition Authority, we apply the method of the regulator [16] according to eqn. (2), which shows the Weighted Average Cost of Capital (WACC).

$$WACC = k_e \times \frac{OK}{(VK+OK)} + k_d \times \frac{VK}{(VK+OK)} \tag{2}$$

where:
k_e– is cost of equity capital (%);
k_d– is cost of borrowed capital or external liabilities (%);
OK – is proportion of equity capital determined by the regulator (%);
VK – is proportion of borrowed capital determined by the regulator (%).

Taking into account the value of the debt coefficient for the financial year 2009 of the power plants CHP 1 and CHP 2 and applying eqn. (2), we find that:

$$WACC_{CHP1} = (6.31 \times 88 + 9.61 \times 12)/100 = 6.74\% \tag{3}$$

$$WACC_{CHP2} = (6.31 \times 93 + 9.61 \times 7)/100 = 6.54\% \tag{4}$$

Assuming that all economic indicators, except investments, are evenly distributed over a 25-year period (according to accounting principle), and taking into consideration the expenditure and revenue (9.7M€ and 24.9M€, respectively) as well as investments of CHP 1, we find that the internal rate of return (IRR) of the plant is 19% on invested funds. Setting IRR equal to WACC, we find that, revenues corresponding to 16.3 M€ would be sufficient to receive WACC from the investment of the undertaking.

Considering the fact that revenue from the sale of heat is a fixed value 12.9 M€ (the amount of generated heat corresponds to the need/weather conditions, and the limit price for heat is confirmed by the Estonian Competition Authority), we gain the needed income from the sales of electricity for achieving the WACC rate that comprises 16.3–12.9=3.4 M€. As the volume of electricity sold in 2009 was 128 GWh, the regulated price per MWh of electricity would equal 3.4 M€/128 GWh=27 €/MWh. By applying the same method as above for CHP 2, we find a price of52 €/MWh. These prices can be compared to the regulated price of oil shale electricity which was 29 €/MWh in 2009 [17]. In principle, this level is the guaranteed or lowest electricity selling price for all plants. Thus, even without supports, provided that electricity is sold at 29 €/MWh, CHP 1 would earn more than necessary for achieving WACC, while CHP 2 would earn less.

There could be several reasons why we receive significantly different results for the two plants. One could be that the plants use different fuels. However, it cannot be excluded that the method of regulation gives incentives to plants to adjust their financial accounts. According to the ratio analysis the rate of return on assets and the debt coefficient are surprisingly weak in CHP 2.

4.3 Price comparison

Since the results of the WACC calculations are somewhat inconsistent, we derive the price excluding FIT from observed sales data. Assuming that the price of electricity was equal to the regulated price implies that the per MWh revenue was 83 € for CHP 1 and 61 €CHP 2, respectively. Using these revenues, we find that electricity sales were 145 GWh and 123 GWh. Since reported sales were smaller, it can be concluded that CHP 1 and CHP 2 earned higher revenues than in the closed market setting. This can be regarded as a result of beneficial contracts entered into with balance providers (Nord Pool Spot's operations). Calculations show that, the average revenues were 40 €/MWh of CHP 1and 36 €/MWh of CHP 2. Based on our analysis, including the above calculations and the previous section suggest that CHP 1 would have operated successfully even without FIT. The evidence of CHP 2 is inconclusive though.

In order to take the analysis one step further we compare the prices to general information about production costs. From a theoretical point of view, we ideally would like to compare prices to marginal costs [18]. Since marginal costs are not available, we approximate marginal costs by average variable costs. In a forthcoming article by Latõšov et al. [19], the authors present cost data of different sized CHP plants in an Estonian context. Using data for the 25 MW_{el} plant, it is possible to calculate the variable cost. Depending on the method of allocating costs between electricity and heat, we arrive at an interval of 4.7–6.7 €/MWh. This is in the same order of magnitude as the average variable cost in the Nordic market, which is 8-9 €/MWh, according to estimates based on [20]. The result shows that both plants receive prices substantially above marginal costs. Comparing revenues to the cost price will provide another benchmark to our case study observations.

The above case study concerns relatively large CHP plants and since unit costs depend on the size of the plant [21] it might not be possible to generalise our results to all plant sizes. In [19], the authors estimate the cost price of electricity of different sized CHP plants. They use data collected in Estonia and the Nordic countries and make calculations of plants with capacity of 1, 10 and 25 MWh_{el} respectively. Assuming a fixed heat price, they derive the per MWh_{el} cost price. Using these observations for fitting a curve, it is possible to approximate the cost prices of a wide range of different plant sizes.

Figure 1 below, indicates that the cost prices of CHP plants with capacity less than 10 MWh_{el} have significantly higher cost prices than larger plants and that there is a rapid increase in cost prices when plant sizes become smaller. Subtracting the FIT from the cost price (see lower curve in Figure 1) shows an even more interesting picture: the FIT covers the cost price of electricity production from a CHP plant with capacity of 25 MW_{el} and when FIT is excluded its cost price is similar to a plant of 4 MW_{el} that receives FIT. These findings confirm the results of the case study and indicate that large plants are overcompensated by the current FIT, while small plants might not receive sufficient support.

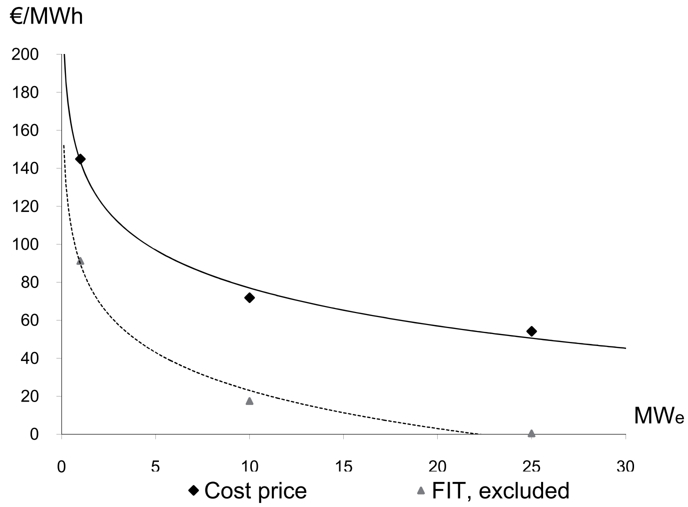

Figure 1: Cost price of CHP plants, euro per $MWh_{el.}$

5 Avoided external costs

A gradual shift from oil shale electricity to renewable sources will have a positive impact on the environment. In order to assess the benefits of FIT in terms of avoided costs, the external costs of air emissions of electricity production from oil shale, wood chip and peat have been calculated. The emission factors are shown in Table 2.

Table 2: Emission factors in g/ $MWh_{el.}$

	Oil shale	Wood chip	Peat
Carbon dioxide, CO_2 (kg)	1156	306	386
Sulphur dioxide, SO_2	7147	400	1676
Nitrogen oxides, NO_X	1075	353	2236
Particulate matter, PM_{10}	494	75	280

Sources: [17, 22–24].

The emission factors of oil shale are based on emission measurements at the Eesti power plant in Narva [22], where about 20% of electricity is generated in fluidized bed combustion and about 80% in pulverised combustion. The external costs were collected from ExternEestimates [25]. Although, Estonia is not represented in ExternE, we follow the application in [26] and base the external costs on Czech brown coal. This transfer of external costs could result in an upward bias, since the estimates also include health effects of pollutants. The risk of bias is due to the fact that population density is higher in the Czech Republic than in Estonia, and the values in use might therefore exaggerate health costs. In the Czech values, health costs make up about 40% of the external cost of brown coal combustion.

Table 3: External costs €/MWh_{el}.

	Oil shale	Wood chip	Peat
Carbon dioxide, CO_2	22.0	5.8	7.3
Sulphur dioxide, SO_2	40.6	2.3	9.5
Nitrogen oxides, NO_X	3.3	1.1	6.8
Total suspended particulates, TSP	3.3	0.6	2.1
Sum	69.2	9.7	25.8

The external costs show relatively large differences. Every MWh of oil shale electricity that can be substituted by electricity produced from wood chip in CHP plants reduces external costs by almost €60 and if replaced by peat, the avoided cost would be about €43.Comparing these values to the Estonian FIT of €54/ MWh and €32/MWh respectively, show that the estimated environmental benefit are higher than the FITs. However, since power plants pay environmental charges, internalisation already takes place. The pollution charges are relatively low though: only about €2 per MWh of oil shale electricity is currently being internalised [17]. Assuming that the influence of a possible upward bias is at an equally low level, the cost of the Estonian FITs are supported by arguments of avoided external costs. An important additional requirement is that the renewable electricity replaces oil shale electricity. So far this replacement has not taken place, but in 2016 when more stringent EU regulation will come into force, pulverized combustion must be equipped with flue gas purification otherwise these boilers have to be shut down [4].

6 Overall assessment

In our evaluation of the Estonian FIT for CHP plants we follow the assessment criteria used previously in literature [2, 6]. One problem though is that the period of assessment is relatively short, stretching from mid-2007 until 2010. Based on evidence so far, Estonia will outperform the target set for 2010, suggesting that the FIT has been effective [5]. The case study showed that large CHP plants have received substantial investment security during the 12 year support period and that the increase of renewable electricity since 2007 has mainly concerned electricity generated by CHP plants. Nevertheless, significant wind power capacity is under construction. According to forecasts, wind energy FIT will double in 2011 compared to 2010 [5].

Since electricity from renewable energy sources receive the same FIT, the Estonian FITs can be judged as technology neutral. However, there are other reasons to question the Estonian FITs from an efficiency perspective. Although the cost price is not covered by the market price of electricity, the case study suggests that 25 MW_{el} CHP plants would have been profitable also without FIT. In addition, market prices significantly exceed the marginal costs of producing electricity from biomass in a 25 MW_{el} CHP plant. On the other hand, pricing at marginal cost would not cover costs since production of electricity in a large CHP plant is characterised by increasing returns to scale.

Construction of small CHP plants has not been encouraged to the same extent by Estonia's FITs. One reason is that small plants have significantly higher generation cost per unit. It is interesting to note that German FITs, which are based on production costs, are differentiated by plant size and do not cover CHP plants fired by biomass that exceed 20 MW_{el} [10].

Another argument for paying a higher FIT than the cost-effective level relates to dynamic efficiency. One motivation is to support a technology to reach market take-off more rapidly than otherwise. Another is general innovation support. However, generation of electricity from biomass in a CHP plant is a mature technology. Therefore, FIT is questionable also from the perspective of dynamic efficiency. From an efficiency point of view, only arguments of avoided external costs can support the current level of FIT. In contrast to experiences in other countries, we find that the avoided external costs exceed the per MWh costs of FIT. The main reason is the high external cost of oil shale electricity.

Between 2007 and 2010, the per kilowatt hour consumer cost has increased from 0.1 to 0.8 eurocents /kWh. In comparison to the Spanish experiences almost a decade earlier, the starting point is equal, but the speed of increase is significantly more rapid in Estonia. The beneficiaries of Estonian FITs have increased their revenues from 6 to almost 54 million Euros during the same time period [7].

The Estonian FIT has low administrative demands as the same FIT has been applied to different energy sources. Setting prices on the closed market according to WACC is rather demanding, though. Our analyses indicate that the current practice might produce distortive incentives and to increase the share of borrowed capital.

7 Conclusion

The purpose of this paper was to assess the impact of the Estonian feed-in tariffs on renewable electricity generation. We have found that the Estonian FIT system has effectively supported establishment of CHP capacity, the administrative costs have been low and the avoided external costs have exceeded the cost of the support. However, the costs of the Estonian FITs have increased at a rapid rate and these costs have been paid collectively by consumers while beneficiaries include large CHP plants.

Besides distributional concerns, there are other reasons to revise the current FIT scheme. The case study of two CHP plants and the comparison of our findings to average cost and cost prices have shown that the current FIT scheme is not efficient. The targets set for 2010 will be exceeded and from an efficiency perspective, this cannot be assessed cost-effective. In addition, the results indicated that resources are used for supporting production that is profitable also without FIT. Even though the current FITs are administratively attractive, the large differences in unit costs depending on plant size, suggest that there is a need to differentiate the FITs to plant size.

The major drawback of pricing measures, such as subsidies and taxes, is that there is uncertainty about the range of impact. In Estonia, as in most other EU countries, FITs are used to reach quantity targets. It is not an easy task beforehand, to choose the level of an FIT that matches the target. Therefore, regulation by FIT requires revisions. Inevitably revisions pose challenges to the investment climate. Therefore regulation by FIT involves a trade-off between the challenges of revisions and the continuation of costly support schemes. Our findings and the forthcoming market liberation, suggest that it is important for Estonia to reform its FIT scheme.

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Appendix 4. "Paying for Environmental Protection in Estonia in International Comparison"

Paying for environmental protection in Estonia in international comparison[1]

Sirje Pädam and Üllas Ehrlich[2]

Abstract

For the first time it has been possible to compare Estonian responses cross-nationally to the questions about willingness to pay for environmental protection included in the International Social Survey Program (ISSP). In international comparisons, Estonia is found in the middle position when willingness to pay is measured in terms of financial support for environmental protection. But Estonia places itself in the group of countries that are least prone to accept cuts in living standard for the sake of environmental protection.

According to the hypothesis of the Environmental Kuznets Curve (EKC) cross-country differences in the willingness to pay for the environment can be explained by different levels of economic development. The analysis shows support for the EKC concerning the willingness to pay higher prices and the acceptance to cuts in the living standards, but there is no correlation between GDP per capita and the willingness to pay higher taxes for environmental protection. A part of the explanation for the missing correlation seems to be that people in high tax countries generally are less willing to accept further tax increases.

1. Introduction

In a cross-country comparison about the use of European Union cohesion funding to the environment it was found that Estonia will devote significantly more funds on a per capita basis to the environmental sector than Latvia (see Pädam et. al 2010). Is this observation backed by stronger public support to pay for environmental protection in Estonia? Is the driving force more severe environmental problems in Estonia than in Latvia, or is there a more general explanation that the demand for a cleaner environment tends to increase with income?

Numerous studies indicate that individual wealth is positively correlated with the willingness to pay for specific environmental goods (see e.g. Flores and Carson

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1997). However, these studies do not relate to environmental protection in general, neither is it possible to use the willingness to pay estimates for specific environmental goods in order to make meaningful cross-country comparisons.

The purpose of this paper is to compare public willingness to pay for environmental protection cross-nationally and to study the main determinants of cross-country differences. Support to environmental protection is measured as the willingness of individuals to make financial sacrifices or accept cuts to one's standard of living to protect the environment. For the first time it is possible to compare Estonian responses to those of the countries that are included in the International Social Survey Program (ISSP).

We begin by a theoretical background to willingness to pay for environmental protection. In the background section we also refer to studies that have tested the theoretical implications. In section 3, we present the data. Section 4 describes the results of the Estonian survey in a cross-country perspective and in section 5 we present the results of the preliminary analyses of the cross-country comparison. The last section concludes the paper.

2. Theoretical Background

The literature presents several alternative explanations to why there are cross-country differences in attitudes towards the environment. Research in environmental sociology proposes that environmental concern in society grows with prosperity. Inglehart (1977, 1995) suggests that there is a change in values from materialist to post-materialist values which takes place when people have met their basic materialist needs for food, shelter and safety. Values are shaped in young ages and they remain relative stable, which imply that growing up under economic affluence leads to prioritization of post-materialist quality of life issues over materialist issues. From this follows the hypothesis that inhabitants in wealthy societies express more concern for the environment. Another implication is that support for environmental protection is higher in countries whose grown-up population has not experienced war or other hardships. In an analysis of ISSP data Bean (1998) found that the willingness to accept lower living standards to protect the environment is higher in Western countries than in former planned economies, quoted by Ivanova and Tranter (2008).

Probably the most well known attempt to explain cross-country differences in environmental protection activities is the Environmental Kuznets Curve (EKC). The EKC suggests that in early stages of economic growth environmental degradation and pollution increase, but beyond some level of income per capita, the trend reverses. The relationship between environmental quality and income can be described by an inverted U-shaped function. When the World Bank made the EKC popular they hypothesized that the mechanisms behind the EKC is that demand for improvements in environmental quality increases with income and so do resources available for investment (IBRD 1992). This proposes that demand for environmental quality increases with income.

Contingent value (CV) surveys, which measure the willingness to pay of individuals for environmental goods, generally find a positive relationship between the environmental good in question and income (Flores and Carson 1997; Hökby and Söderqvist 2003). In a handbook from the Swedish EPA, positive correlation between the environmental good and individual income is specified as one criterion for assessing the quality of an environmental willingness to pay study (Naturvårdsverket 2005). Also the practice of benefit transfer, i.e. transferring willingness to pay estimates, from one country to another typically adjust for GDP per capita (see for instance Turner et al. 1999 and Huhtala et al. 2009). The values and preferences of individuals thus provide important explanations to variations in how people choose to allocate money between improvements in environmental quality versus other goods, and general practices assume that these differences are detectable on the national level.

On the other hand, growing demand for environmental quality could also stem from scarcity due to the environmental degradation that occurred during earlier stages of economic growth, implying that when the quality of the environment deteriorates the marginal willingness to pay for environmental protection increases.

In principle both sociological and economic research provide similar implications to the differences we may expect to find cross-nationally when comparing the willingness to pay question of attitude surveys. Ivanova and Tranter (2008) have tested the post-materialistic hypothesis using willingness to pay questions of the ISSP in a cross-country perspective. They found that individuals expressing post-materialist values are more willing to pay for environmental protection than materialists. In their study, the main drivers for expressing positive willingness to pay for the environment are found to be tertiary education, value orientation and the concern about environmental risk. Their study did not assess how cross-country differences in the level of income affect the willingness to pay for the environment, neither did they test the influence of environmental quality on the willingness to pay.

3. Data

The International Social Science Program (ISSP) has carried out a series of international surveys. The latest survey concerns 25 countries providing about 29,500 observations. Unfortunately, Estonia has not been covered. In a recent survey a selection of ISSP survey questions were for he first time collected for Estonia. This makes it possible to compare the willingness to pay attitudes of Estonia cross-nationally. The survey was conducted during December 2009 to February 2010. About 1,200 respondents were contacted by interviewers in Tallinn and in rural areas of Estonia. Almost 850 filled in questionnaires were returned, giving a response rate of about 70 percent.

In this paper we examine three questions covered by the Estonian survey and by the ISSP. The questions are “How willing would **you** be to pay **much higher prices** in order to protect the environment?”, “How willing would **you** be to pay **much higher taxes** in order to protect the environment?” and “How willing would **you** be to **accept cuts** in your **standard of living** in order to protect the environment?” (Bold emphasis appears in original questions). The responses to these questions were given on a six point scale: “very willing”, “fairly willing”, “neither nor”, “fairly unwilling”, “very willing” and “don’t know”.

By using an attitude survey rather than quantitative responses from contingent valuation (CV) studies provides less precision. The contingent valuation questions are precise about the environmental good in question, e.g. protection of a wetland or a species. In addition, CV surveys typically ask for quantitative amounts in terms of how much the respondent is willing to pay for a given improvement. Another widely used way to elicit the willingness to pay is the referendum type of question: “Do you accept to pay X Euro (as a voluntary contribution/tax increase) for the environmental good Y?” The respondents reply “yes” or “no” to this question. However, in the attitude surveys protection of the environment is very general and the method of provision remains uncertain. Rather than finding a monetary amount, the Estonian Environmental Attitude Survey and the ISSP offer qualitative response categories.

Irrespective of these shortcomings, we can capture attitudes towards the willingness to pay. This information is valuable since sacrifices are hypothesised in terms of payment and lower standard of living. By this the responses differ from general attitude questions. Furthermore, by adding the responses of “very” and “fairly” the responses will be similar to the referendum type of CV study. Another advantage is that the attitude survey allows cross-country comparisons.

For the purpose of cross-country comparisons we use GDP per capita data based on purchasing-power-parity (PPP) from the international monetary fund, (IMF 2009). In the analysis we use GDP per capita of the survey year. Since survey year data is not yet available for Estonia we have adapted the IMF forecast of GDP per capita in 2010. The 2010 tax burden variable of the index of economic freedom has been used for analysing the influence from taxes on the responses. (Heritage Foundation, 2010). In order to capture cross-country differences in environmental quality we adopt the Environmental Performance Index (EPI), (see Emerson et al. 2010). This index is based on empirical data about the environment in 163 countries. The index has been developed by first identifying specific environmental targets and then measuring the distance between the target and current national achievement (ibid). Since common methodology is used, the EPI serves well for our purposes.

4. Willingness to Pay in Cross-country Comparison

Figures 1, 2 and 3 present the results of the cross-country comparison in a generalized way according to the referendum type of replies. The answers “very” and “fairly” have been added up and are shown in columns “Willing” and “Unwilling” in the figures. The answers that did not express any clear preference in the 6-point scale are omitted from the analysis.

Responses to the question “How willing would you be to pay much higher prices in order to protect the environment?” across countries are presented in Figure 1. The countries are listed by order of the percentage of “willing” responses in ascending order. The smallest (21.5%) is the percentage of “willing” responses in Latvia, somewhat surprisingly followed by the old EU member states Portugal and Finland (22.2 and 23.1 percent, respectively). It is difficult to explain such a low percentage of positive responses particularly in Finland, which is a country with high standard of living and high environment quality. The position of Finland among the countries is the more surprising since production and consumption of organic and nature-friendly products is very wide-spread, suggesting that in real life Finns are often willing to pay higher prices compared to so-called conventional products. “Willing” answers were fewer than 30 percent also in Bulgaria and Russia, which is quite in line with the income level of these countries. The biggest percentage of “unwilling” answers was in Bulgaria (57.6%) and Latvia (56.9%). Finland’s position in the rear is also justified by its large percentage of “unwilling” responses (49.8%).

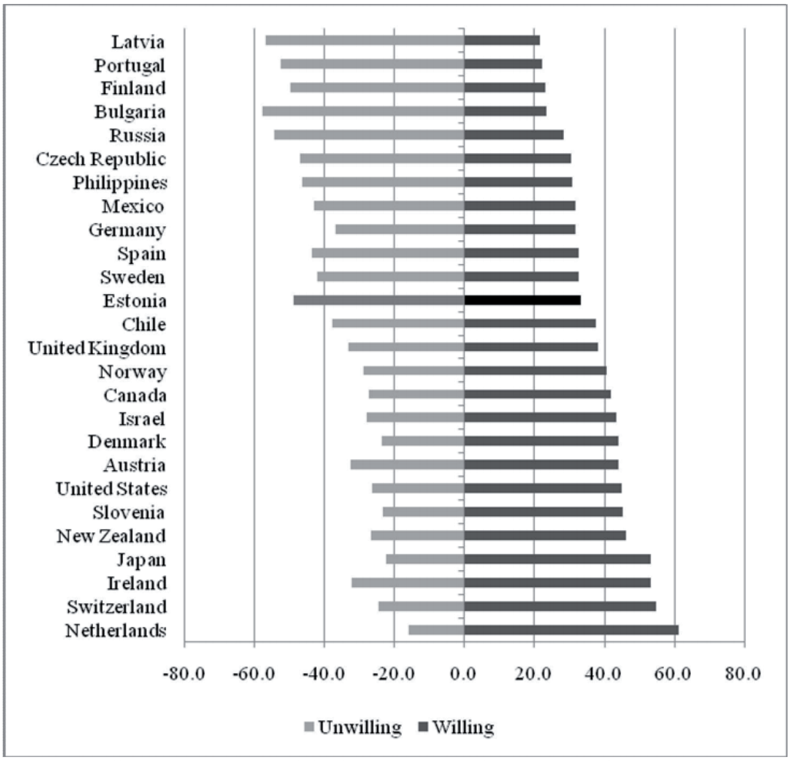

Fig. 1: Willingness to pay much higher prices in order to protect the environment

"Willing" responses between 30 and 40 percent are found in the Czech Republic (30.4%), Philippines (30.9%), Mexico (31.6%), Germany (31.7%), Spain (32.6%), Sweden (32.7%), Estonia (33.4%), Chile (37.5%) and the UK (38.0%). This interval represents countries with very different income levels and varying quality of the environment. While the Czech Republic, Philippines, Mexico, Estonia and Chile were expected to be among 30−40 percent "willing" responses, then it is rather surprising to find countries with high environmental awareness in this group such as Germany and Sweden. Percentages of "unwilling" responses in this interval are largely variable: lower than 40 percent in the UK (33.1%), Germany (36.7%) and Chile (37.9%) and over 45 percent in the Philippines (46.3%) and Estonia (48.8%). Hence, a conclusion is that although people are not so willing to pay much higher prices in Germany and the UK in order to protect the environment, they are definitely not very much against it.

"Willing" responses are found in the interval between 40 and 50 percent in Norway (40.7%), Canada (41.7%), Israel (43.8%), Denmark (43.9%), Austria (44.1%), the US (44.9%), Slovenia (45.1%) and New Zealand (46.1%). That countries with high standard of living are in this interval is according to expectations. The high position of Slovenia is surprising. Slovenia has by a vast majority the highest percentage of "willing" responses among the former planned economies included in the comparison. The percentage of "unwilling" responses among countries in the interval of 40−50 percent is relatively even: between 32.6 percent (Austria) and 23.4 percent (Slovenia).

The biggest percentage of "willing" responses is in Japan (53.1%), Ireland (53.3%), Switzerland (54.6%) and the Netherlands (61.6%). The latter is leading by a vast majority the willingness to pay higher prices among the countries included. By far the smallest share of unwilling respondents is found in the Netherlands (16%). There are no surprises in the leading group of countries; all the countries here have both high standard of living and high environmental awareness.

Generalized answers to the question "How willing would you be to pay much higher taxes in order to protect the environment?" across countries are presented in Figure 2, according to the principles of the previous question. As the question is about direct large tax payments, it may be assumed that the topic is more sensitive than paying higher prices. The average share of "willing" responses is relatively smaller here, compared to the previous question, and that of "unwilling" responses higher than to the previous question about higher prices. This question has both the smallest percentage of "willing" (12%) and the highest percentage of "unwilling" (67.3%) responses in Finland. "Willing" responses below 20 percent are also represented by Bulgaria (16.5%), Portugal (17.1%), Latvia (17.3), Czech Republic (17.3%), Germany (18.2%), Austria (19.2%) and Sweden (19.5%). It is interesting to note that the countries with the smallest percentage of "willing" responses include also so-called welfare countries such as Germany, Austria, Finland and Sweden. A possible explanation for the low willingness to pay in these countries might be that taxes in general are high and that people are well aware of revenues from taxes being used for the environment and that they therefore do not regard tax increases necessary. All these countries are also on top of the ranking list on the basis of percentage of "unwilling" responses.

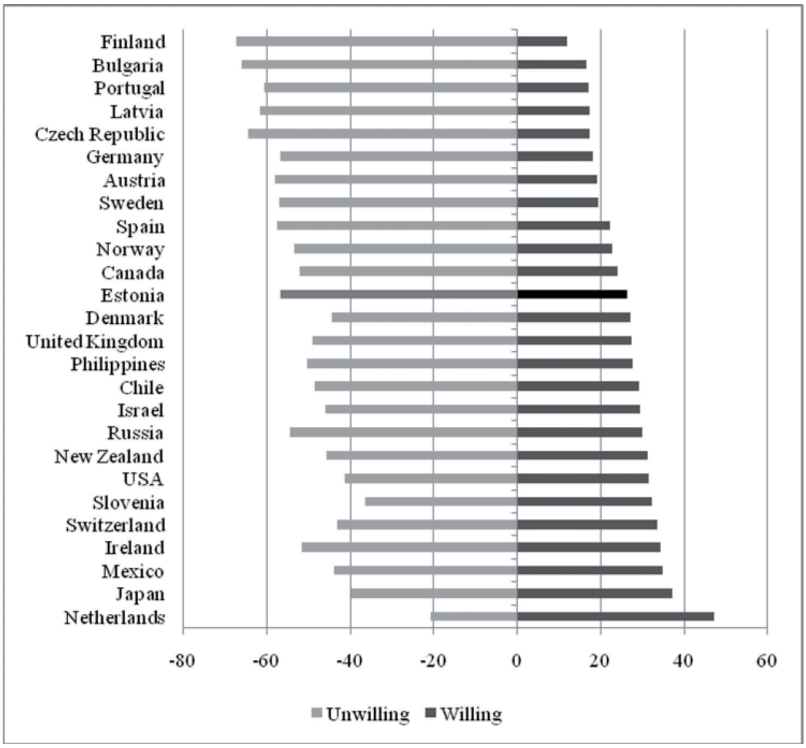

Fig. 2: Willingness to pay much higher taxes in order to protect the environment

In the next willingness to pay interval where the percentage of "willing" responses is between 20 and 30, we find Spain (22.2%), Norway (22.8%), Canada (24%), Estonia (26.4%), Denmark (27.1%), the UK (27.3%), Philippines (27.5%), Chile (29.2%), Israel (29.4%) and Russia (29.9%). While other countries in this group were lower than average also in the ranking list on the basis of the previous question, then Denmark was rather among the countries which were above average concerning the share of willing to pay higher prices. In order to explain the low willingness to pay higher taxes of many high income countries such as Norway, Canada, Denmark and the UK we could make a similar hypothesis as for Germany, Austria, Finland and Sweden in the previous interval. All countries in this group also show up by high percentage of "unwilling" responses (over 50%).

The countries where the percentage of "willing" responses to pay higher taxes is over 30 include New Zealand (31.1%), the US (31.6%), Slovenia (32.3%), Switzerland (33.5%), Ireland (34.3%), Mexico (34.7%) and Japan (37.2%). Like in the previous question, the Netherlands is on top of this ranking list by a vast majority with 47.2 percent. The Netherlands also has the smallest percentage of "unwilling" responses. The group of countries with the highest percentage of "willing" responses resembles the countries with the highest percentage of "willing" responses to the previous question about higher prices. A big exception here is Mexico, which is the third by the percentage of "willing" responses to the tax question, but in the ranking list of the price question was among the countries with a low percentage of "willing" responses. The authors have no reasonable explanation for that. In general, standing out among the countries with a high percentage of "willing" responses are Russia (54.3%) and Ireland (51.5%) with their remarkably high percentage of "unwilling" responses. In addition to the Netherlands, the smallest percentage of "unwilling" responses is in Slovenia (36.5%), which places this country, analogously with the ranking list formed on the basis of the previous question, in an exceptional position among former planned economies.

Generalized answers to the question "How willing would you be to accept cuts in your standard of living in order to protect the environment?" across countries are presented in Figure 3, according to the principles of the previous questions.

The difference in the percentages of "willing" responses to this question is the largest, ranging from 5.7 percent in Latvia to 57.2 percent in Switzerland. In addition to Latvia, the countries with the smallest percentage of "willing" responses include Bulgaria (12.1%), Portugal (16.8%) and Estonia (18.5%). The highest percentage of "unwilling" responses is in Latvia (78.1%). Surprising is also that Estonia ("unwilling" responses 58.9%) belongs to the group of countries which are the most clinging to the standard of living, although in the ranking lists on the basis of previous questions it was in the middle. In addition to the above-mentioned countries, the percentage of "willing" responses to the standard-of-living question was below 30 also in the Czech Republic (21.0%), the UK (23.0%), Chile (27.6%) and the US (29%). Differences in this group are proportional to the percentage of "unwilling" responses. In the UK it is 54.8 percent in the USA only 43.9 percent.

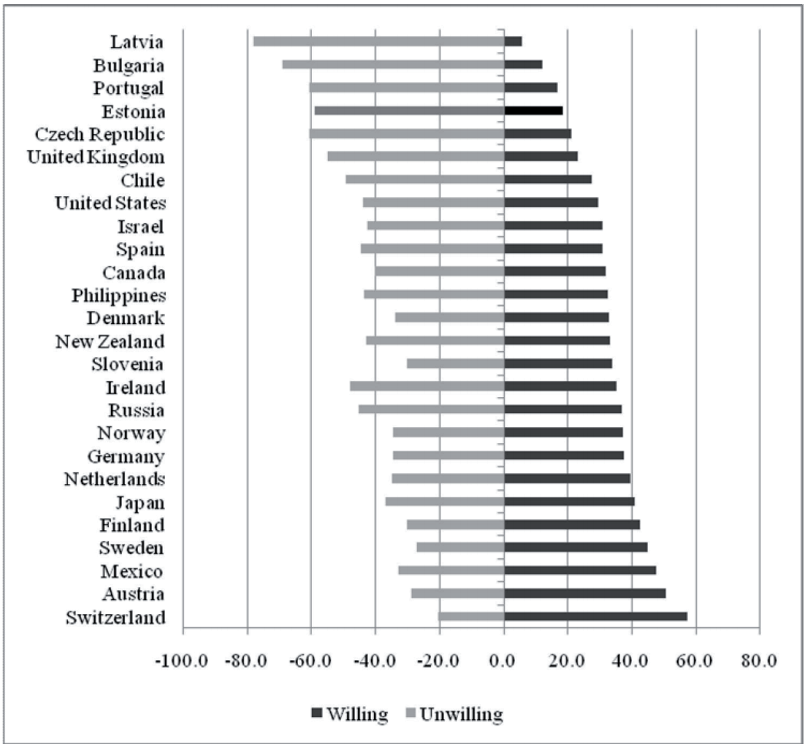

Fig. 3: Willingness to accept cuts in standard of living in order to protect the environment

The group of countries with "willing" responses between 30 and 40 percent is the most numerous. Following countries belong to this group: Israel (30.8%), Spain (30.9%), Canada (31.8%), Philippines (32.5%), Denmark (32.8%), New Zealand (33.0%), Slovenia (33.9%), Ireland (35.5%), Russia (36.8%), Norway (37.3%), Germany (37.4%) and the Netherlands (39.6%). In the ranking list on the basis of the standard-of-living question, the Netherlands has fallen into the "middle" group, which in the ranking list on the basis of the previous questions was the dominant leader, as well as Slovenia, which belonged to the leading group by percentage of "willing" responses. The percentage of "unwilling" responses in this group of countries varies between 48.0 (Ireland) and 45.3 percent (Russia).

The countries where the percentage of "willing" responses to the question is higher than 40, include Japan (40.8%), Finland (42.4%), Sweden (44.8%) and

Mexico (47.7%). The highest percentage of "willing" responses is in Austria and Switzerland with 50.5 and 57.2, respectively. While Switzerland is among the countries with the highest percentage of "willing" responses in the ranking lists on the basis of all three questions, then Austria's percentage of "willing" responses to the willingness-to-pay question was below average. This is an interesting dilemma – they are not willing to pay higher taxes in order to protect the environment, but can accept cuts in the standard of living for the same purpose. The same applies to Sweden, which on the basis of two other questions is among the countries with lower than average percentage of "willing" responses. It is surprising that Mexico has a position among the countries with the highest percentage of "willing" responses. Unlike Austria, Mexico can be found among the countries with the highest percentage of "willing" responses also on the basis of the tax question, whereas Mexico has lower than average willingness-to-pay higher prices in order to protect the environment. Finland surprises with the willingness to accept cuts in the living standard in order to protect the environment, but on the basis of two previous questions belongs to the group of countries with a small percentage of "willing" responses.

To sum up all three questions it may be noted that Latvia, Portugal, Bulgaria and the Czech Republic are placed among the countries with lower percentage of "willing" responses to all three questions. In the leading group on the basis of the percentage of "willing" responses to all questions we find Switzerland, the Netherlands and Japan. The reasons that determine the percentages of "willing" and "unwilling" responses in countries require further investigation.

5. Analysis

According to the EKC, cross-country differences in the willingness to pay for environmental protection can be explained by different levels of economic development. In order to test for this, correlations were calculated between the share of "willingness" responses and the GDP per capita, adjusted for PPP. The results show that a yes to the willingness to pay much higher prices and a yes to accept cuts in the living standards are correlated to GDP per capita, while a yes to the willingness to pay much higher taxes is not. There is some correlation between a country's tax burden and the willingness of its population to pay higher taxes for environmental protection, but this is less distinguishable than the correlations between GDP and the two other replies. However, contrary to our expectations, there is no correlation between the unwillingness to pay higher taxes and the tax burden of a country. Since environmental quality could influence the environmental attitudes, correlations between the share of yes responses and the environmental performance index (EPI) were calculated. Table 1 shows the results.

	GDP per capita	Tax burden	EPI 2010
“Willing” to pay much higher prices in order to protect the environment	0.361	0.013	0.009
“Willing” to pay much higher taxes in order to protect the environment	0.026	-0.138	-0.058
“Unwilling” to pay much higher taxes in order to protect the environment	-0.087	0.038	0.020
“Willing” to accept cuts in living standards in order to protect the environment	0.203	0.000	0.214

Tab. 1: *Correlations between survey questions and GDP per capita, tax burden and EPI 2010*
Sources: ISSP, Estonian Valuation study, IMF (2009), International heritage fund (2010) and Emerson et al. (2010)

The EPI is correlated to a yes to acceptance of cuts in living standard, but not to the other questions. It is interesting to note that the EPI is correlated to the “willingness” to accept cuts in one’s living standard. However, at closer inspection this correlation shows that people in countries with high quality natural environments are more willing to sacrifice living standard in order to protect the environment, than people who live in countries with lower quality environment. The reason for this correlation could be that richer countries have both higher environmental quality and higher living standard and that it is the higher living standard which explains why people in rich countries are more prone to accept cuts in their living standard. Since this result only seems to confirm the correlation between GDP per capita and living standard, the analysis does not give any support to the possible scarcity arguments for stronger demand for environmental protection in countries with environmental degradation.

6. Conclusions

In this paper we have for the first time compared public willingness to pay for environmental protection in Estonia cross-nationally. Three questions covered by a recent Estonian survey and by the ISSP were compared and analysed. The cross-country comparison puts Estonia in the middle position when willingness to pay is measured in terms of paying higher prices and higher taxes for environmental protection. However, in terms of acceptance to cuts in living standard for the sake of environmental protection, Estonia places itself in the least willing group of countries. It is also interesting to note that former planned economies included in the sample differ somewhat in their positions. Slovenia places itself in the top and in the middle, while most former planned economies included in the comparison are found in the lower end of the ranking list. Latvia, Bulgaria and the Czech Republic are placed among the countries with lower percentage of “willing” responses to all three questions. In the leading group on the basis of the percentage of "willing" responses to all questions we find Switzerland, the Netherlands and Japan.

The initial question set out in this paper seems thus to have a positive answer: the higher per capita EU cohesion fund allocation to the environment in Estonia than in Latvia is backed by a stronger public support to pay for environmental protection in Estonia. In our paper we also find support for the hypothesis that demand for the environment tends to increase with income.

The analysis shows support for the EKC concerning the willingness to pay higher prices and the acceptance to cuts in the living standard. At the same time the results indicate that the willingness to pay much higher taxes neither depends on income, nor on differences in environmental performance. As pointed in the cross-country comparison, the responses to the tax question do not follow the expectations about the positions of different countries, since there seem to be a low willingness to pay higher taxes in some of the rich countries. There is some correlation between a high tax burden and the reluctance in willingness to pay higher taxes. The results thus indicate that there might be other motivations than environmental protection influencing the responses to the tax question.

The analysis has covered cross-country differences. Further study will be needed in order to make a distinction between the influence of country and individual characteristics respectively. For this purpose a regression analysis that covers both individual and country specific characteristics need to be applied.

The results also indicate that there is a need for further studies about the anomalies in responses to the question about willingness to pay higher taxes. Since CV surveys in many cases use tax increases as the payment mechanism, the implications of our observations deserve more in-depth analysis about the validity of using hypothesized tax increases in willingness to pay questions of CV surveys.

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Appendix 5. “The Willingness to Pay for Improving Animal Wellbeing at Tallinn Zoological Gardens”

The Willingness to Pay for Improving Animal Wellbeing at Tallinn Zoological Gardens

*Sirje Pädam and Üllas Ehrlich**

Abstract

Tallinn Zoological Gardens lacks necessary funding for its development, and it remains uncertain whether the zoo will achieve its goals concerning animal wellbeing, i.e. provision of adequate accommodation to all zoo inhabitants. In order to find the willingness to pay for improving animal wellbeing, we conducted a contingent valuation survey (CVM) among the adult population of Tallinn. The sample consisted of zoo visitors and non-visitors. The reason for making a distinction between visitors and non-visitors is to find potential non-use value. The analysis shows that there is significant non-use value from improvements of zoo animal wellbeing. It was also found that the average WTP for animal wellbeing increases with the frequency of visits. However, the aggregated WTP of the Tallinn adult population does not exceed the costs of planned investments. Since Tallinn Zoo attracts visitors from all of Estonia, the increase in human welfare from improving animal wellbeing is experienced by a larger number of people than the population of Tallinn. By expanding the WTP to the adult population of Estonia we find that planned investments are motivated from a cost benefit perspective.

Introduction

Tallinn Zoological Gardens is the only zoo in Estonia that is a member of international zoo associations. As a member of World Association of Zoos and Aquariums (WAZA) and its European counterpart European Association of Zoos and Aquaria (EAZA), Tallinn Zoological Gardens has devoted itself to participate in a number of activities that enhance animal wellbeing and to participate in reproductive biology, conservation, research and to carry out biological education. For several years, however, the city of Tallinn has been unable to provide necessary funding for the development of the Tallinn Zoological Gardens and during the economic crisis, the situation has worsened. Due to the cuts in city funding, the

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modernisation of the polar bear enclosure, new heating system and improved enclosures for bearded vultures have been postponed.

The development strategy of the zoological garden foresees investments of about 40 million Euros during the five year period 2008–2012 in order to provide modern facilities (Tallinn, 2007). The document points out that on the territory of the zoo there still are "temporary buildings and structures adapted from former military warehouses, which do not meet the needs of the animals nor the European Community Directives" (ibid. p 3). The referred Directive is the Council Directive 1999/22/EC, which defines requirements regarding animal welfare, e.g. that zoos provide adequate accommodation for zoo animals in order to satisfy biological and species-specific needs. A non-satisfactory state of animal welfare at Tallinn Zoological Garden has been reported by the British Born Free Foundation that in its assessment points out that 27 per cent of enclosures failed to meet the requirements specified by the Estonian minimum standards (Born Free Foundation, 2011).[1] However, with the current level of funding, planned investments may be delayed by a decade or two and it remains uncertain whether the zoo can achieve the goals of animal wellbeing.

Zoological gardens produce a variety of services. An obvious service is the event of a visit. Animal wellbeing is only one aspect of the visit. Since zoo visitors experience a range of services during their visit to the zoo, only direct questions about animal wellbeing can help to extract the human valuation of animal wellbeing from other values of a zoo. Earlier research has shown that zoo visitors find animal health and natural habitat-like enclosures important quality aspects of a zoo visit (Tomas et al. 2003). It has also been shown that zoo visitors support promotion of animal welfare and would be willing to pay for it (Zhao and Wu, 2011). However, whether non-visitors yield benefits from improved zoo animal wellbeing, i.e. non-use value, has not previously been a topic of research.

The main focus of this article is to identify whether the planned, but postponed investments in adequate accommodation are matched by public benefits. Another issue of interest is whether there is an element of non-use value in human valuation of zoo animal wellbeing. For these reasons a contingent valuation survey was conducted in autumn 2010. The survey covered a sample of individuals who do and who do not visit Tallinn Zoological Gardens.

In the next section we present the finances of Tallinn Zoological Gardens and discuss the principles of zoo funding. After that, in Section 2, we provide an overview of the potential benefits that arise from improvements in animal wellbeing and discuss possible ways to assess them. In section 3, we report the details of the survey and provide descriptive statistics. Section 4 presents the statistical analysis and the estimate of the consumer surplus. The final section concludes the paper.

[1] This claim has been disputed by the Estonian Veterinary and Food Board responsible for inspections at Tallinn Zoo, (Tallinn Zoo fulfils minimum requirements 2011)

1. Zoo Funding

Since zoos produce public goods, there are efficiency reasons to finance zoos via the public budget. However, the ownership of zoos varies from case to case and so does public sector funding. In most countries zoos receive partial support from public funds even though the zoo is privately owned. The zoos of Cologne and Frankfurt are supported by the municipality but are run by zoological societies, while the Paris zoo is one of the many institutions directly supported by the French Ministry of Education (Zoo 2006). Tallinn Zoo is owned and funded by the municipality. Additional income mainly constitute of admission fees. Riga Zoo has a similar ownership and history as Tallinn Zoo, but has been recognized as a national zoo and receives additional funding from the Latvian ministry of the environment.

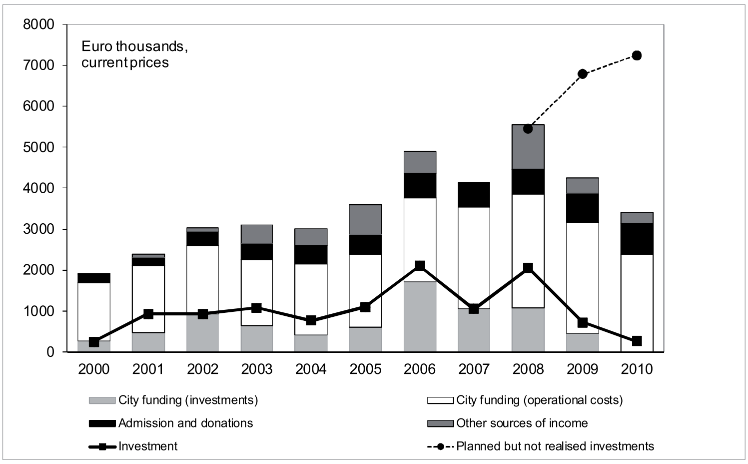

Fig. 1: Investments and sources of revenues at Tallinn Zoo, Euro thousands
Source: Authors' calculations based on Tallinn (2007) and Annual reports 2000−2010

The Figure shows that city funding of investments has fallen substantially. In 2010, the contribution of the city to investments was zero. The year before, the city's investment contribution was about 0.5 million Euros, which represented a cut by half compared to 2008. According to annual reports, the investment share of the Tallinn Zoological Gardens has on average been about 30 per cent of turnover during the past decade (Annual reports 2000–2010). The lowest levels of investments were recorded in 2010 and 2000, with 8 and 13 per cent of turnover respectively. Corresponding shares of 2008 and 2009 were 37 and 17 per cent respectively. If investments had matched the pace foreseen by the development strategy, their share of turnover would have reached an even large share than historically: the planned investments of an additional 13 million Euros in 2010 would have reached a level of more than 300 per cent of total costs.

It thus seems as Tallinn Zoological Gardens with one major funder has been vulnerable to the impact of the economic crisis. In times of crises it might be reasonable to cut costs, but the postponed investments have a price in terms of a lower level of animal wellbeing.

2. Benefits of Animal Wellbeing

In order to guide zoos about animal wellbeing, legislation has provided definitions about minimum levels of animal care and zoo design. Estonian legislation prescribes that animal keepers must allow appropriate amounts of food and drinking water in relation to animal species and age, and that zoos shall be designed to ensure animal health and welfare and to prevent escapes (RT I 2001, 3, 4, § 3 and § 28). Details about zoo construction and enclosure design are regulated by zoo planning requirements (RT I 2004, 57, 408). However, using the definitions of the legislation does not tell us the value of improving the wellbeing of captive animals or whether the value is sufficiently large to cover the costs of investments.

2.1. Animal Evaluation of Animal Wellbeing

Ideally the benefits of improved animal wellbeing should be evaluated by the animals themselves. Jordan (2005) suggests that lack of animal welfare can be assessed by observing animal behaviour, i.e. deprived welfare is detected if an animal exhibits similar behaviour to that of animals whose physiological responses have been confirmed as indicating poor welfare. There are, however several limitations to this approach. One obvious problem is that the response in animal behaviour cannot be compared to the size of the planned investments. In addition, there are further limitations. Goulart et al. (2009) found that most research about animal welfare is based on studies focussed on only a few species of mammals (e.g. chimpanzees). Further they identified that there are significant gaps in our knowledge concerning the welfare of fish, amphibian, reptile and invertebrate, as well as the link between health, physiology and zoo animal welfare and how to convert theoretical knowledge into practical solutions for zoo animal welfare (ibid).

2.2. Human Evaluation of Animal Wellbeing

Therefore, out of practical reasons, the benefits arising from improvements of animal wellbeing at zoos will be defined as the value humans place on improvements of animal wellbeing at zoo. The literary review of Tomas and her co-authors suggests that the benefits people primarily seek when paying a visit to the zoo are related to social interaction during a zoo visit (Tomas et al. 2003). In their own survey among visitors at Forth Worth Zoo they find the following ranking of six benefits domains: 1) Family Togetherness, 2) Wildlife Enjoyment, 3) Wildlife Appreciation and Learning, 4) Companionship, 5) Escape and 6) Introspection/Meeting New People. From these results the authors conclude that people generally do not come for escape, introspection or to make new acquaintances. People visit the zoo for reasons of social interaction and an enjoyable experience interacting with wildlife. Although the questionnaire did not directly relate to animal wellbeing, the authors found that the service quality attributes concerning the health of the animals and viewing them in natural-like habitats were ranked as the two most important.

Since visitors to zoos find animal health and natural habitat-like enclosures important, the findings reported by Tomas et al. (2003) suggest that animal wellbeing includes use value. Assuming that there also is non-use value related to the enhancement of the wellbeing of zoo animals makes it probable that the willingness to pay for animal wellbeing is higher among zoo visitors, than among non-visitors.

An in-depth enquiry about the economic and social contribution of the zoological industry in Australia identifies the following five economic values of a zoo: Production Value, Consumer Value, and the values of contribution to Conservation, Education and Bio Security (Aegis Consulting and Applied Economics 2009). The authors of the Australian enquiry do not discuss animal wellbeing, but they point out that visitor expenditure and contribution from zoo friends is an indicator of the minimum benefit of consumer valuation (p. 20). They note that contingent valuation study or travel cost methods need to be applied in order to find the real user value. In this respect the Australian study does not recognise any other non-use values of the zoo industry than possibly that of zoo contributions to conservation. Owing to the fact that animal wellbeing is merely one aspect of the benefits experienced by zoo visitors, data about visitor expenditures and contributions from zoo friends as a minimum indicator is not applicable for the purpose of this research. Only direct questions about how individuals value animal wellbeing can help to extract the human valuation of animal wellbeing from other zoo benefits.

A recent survey that was carried out at six Chinese zoos showed that although the term animal welfare was unfamiliar to most Chinese zoo visitors, a vast majority of the interviewees were in favour of implementing legislation to protect zoo animals and almost 90 per cent reported they would be willing to pay for animal welfare (Zhao and Wu 2011). Since no questions about monetary contributions were included in the Chinese survey, information about the value that zoo visitors put on animal wellbeing is missing.

By finding the willingness to pay for the planned improvements at Tallinn Zoological Gardens, it will be possible to compare the costs to the monetary value humans put on the enhancement of animal wellbeing. If animal wellbeing at zoos is a pure use value, it would be sufficient to ask zoo visitors. The reason for making a distinction between visitors and non-visitors is that the benefits of animal wellbeing might not only be related to people's visits to the zoo. The concept of non-use value stems from Krutilla (1967) who suggested that people derive utility from natural assets just because of the existence of natural assets. Hence, the hypothesis is that the total economic value of a zoo includes non-use value of animal wellbeing.

3. Tallinn Zoo CVM Survey

As a part of their undergraduate studies in Economics and Business Administration at Tallinn University of Technology, students taking environmental economics were asked to distribute ten questionnaires each to a sample of different age groups representative of the adult population of Tallinn. Because participants received course credits, response rates were high: 86 per cent. In total, 1,029 questionnaires were returned out of the 1,200 that were originally distributed. Since 25 questionnaires lacked a willingness to pay (WTP) statement, 1,004 replies remained for further analysis. Due to an overrepresentation of the two youngest age groups and underrepresentation of age groups above 60 years, the sample was weighted according to the age structure of the adult population of Tallinn.

The WTP question was open ended, including a reminder that the respondent should consider his or her budgetary means when replying. The WTP question was stated in terms of the annual willingness to pay during a five year period for additional investments to enhance animal wellbeing according to the improvements of the development strategy. The average annual WTP equals 19.8 € and the median 6.4 €. Excluding questionnaires that were not completely filled in, omitting two extremes with WTP exceeding 600 Euros leaves 990 responses for further analysis. Weighing the remaining sample in accordance to the age structure of Tallinn population results in an average WTP of 17.3 € and a median of 5.6 €.

A potential problem with WTP questions arises if respondents are of the opinion that funding is the responsibility of somebody else, and for this reason do not accurately report their willingness to pay. Typically, this kind of attitude could result in so called protest votes, i.e. respondents who incorrectly report that their WTP is equal to zero. In order to detect potential protest voting, the questionnaire included a question about the respondent's opinion about zoo financing from alternative sources. In total 85 respondents, i.e. 8.5 per cent reported that they did not support the idea of using alternative funds. However, only 37 of them reported zero WTP.

Table 1 below shows the number of responses and the average WTP by the frequency of visits to Tallinn Zoo.

	How often do you visit Tallinn Zoological Gardens?				
	Never	Seldom	Once a year	Once a month	All responses
WTP>0, responses	89	397	302	17	805
WTP=0, responses	55	88	39	3	185
Number of responses	144	485	341	20	990
potential protest voters	7	17	10	3	37
Average WTP, €	7.7	15.5	23.0	32.9	17.3

Tab. 1: *Willingness to pay with respect to frequency of visits to Tallinn Zoo*
Source: Authors' calculation of responses to Tallinn Zoo CVM

The most common reply to zoo visit frequency is "seldom", followed by "once a year", "never" and "once a month". Those who reported that they never visit Tallinn Zoo, but still reported a positive WTP can be interpreted as expressing non-use value for animal wellbeing. The rest of the sample express both use and non-use value. The non-use value of 7.7 € makes up almost 45 per cent of the sum of the use and non-use value. It is also interesting to note that the average WTP increases with the frequency of visit. This could imply that animal wellbeing is rated as more important by frequent visitors because of the quality aspect. Another interesting finding is that the propensity of being a potential protest voter increases with frequency of visit.

The next table shows willingness to pay according to socio-metric variables.

		Average WTP, €	Difference from total average, %
Gender	Male	15.4	89.0
	Female	18.8	108.4
Education	Primary	13.6	78.5
	Secondary	12.1	70.2
	Secondary technical	17.9	103.3
	Higher	21.4	123.7
Age	18-23	8.9	51.6
	24-29	17.0	98.2
	30-39	20.2	116.8
	40-49	18.8	108.6
	50-59	21.4	124.0
	60-69	23.8	137.7
	> 70	13.7	79.3
Average monthly income (net), €	<128	7.4	43.0
	128-255	8.6	49.9
	256-383	11.7	67.5
	384-511	14.0	81.0
	512-703	20.1	116.4
	704-958	22.7	131.4
	959-1278	22.2	128.1
	>1278	45.3	261.8
Total average		17.3	100.0

Tab. 2: *Willingness to pay (WTP) with respect to socio-metric variables*
Source: Authors' calculation of responses to Tallinn Zoo CVM

4. Statistical Analysis

The statistical analysis is carried out in two steps. In the first step we use a binary logit regression to allow us to assess the influence of socio-metric variables to the decision to pay or not to pay. In the second step, an OLS regression is applied to the sub-sample that has a positive WTP in order to determine the relationship between the stated amount and the socio-metric variables. Finally the positive WTP replies are used as an input for finding the aggregated consumer surplus.

The statistical analysis is based on 990 fully completed questionnaires. The main reason for excluding some of the questionnaires was the absence of a WTP amount. Two questionnaires with extremely high WTP amounts (more than 600 € per year) are also dropped out, the other amounts are between 0 and 400 € and have been checked to be believable in comparison to the stated average monthly income as well as the attitude of the respondents.

4.1. Determinants of a Positive WTP

The binary logit regression allow us to assess the influence of socio-metric variables on the decision to pay (WTP>0) or not to pay (WTP=0). In the logit regression we consider the decision to pay (DTP) as a dichotomous choice problem (DTP=0 if WTP=0 and DTP=1 if WTP>0) and assuming a logistic distribution, the probability that individual *i* is willing to pay some positive amount (P_i) is expressed as:

$$P_i = \Pr(DTP_i = 1 \mid X_i) = \frac{\exp(a + \sum_{k=1} b_k x_{ik})}{1 + \exp(a + \sum_{k=1} b_k x_{ik})} = \frac{1}{1 + \exp(-a - \sum_{k=1}^{K} b_k x_{ik})} \qquad (1)$$

where $X_i = (x_{i1} \ldots x_{ik})$ is the vector of socio-metric variables of individual *i* (gender, age, education, income) and *a*, b_k the k=1,...,K parameters. The logit model is given as a linear relationship between natural logarithms of the odds ratios and the explanatory variables. The explanatory socio-metric variables used in the logit analysis are gender, age, education and income. Gender is a dummy variable (male=1, female=0), other variables are ordered categorical variables. The logit model is estimated by the maximum likelihood method. The initial model that identifies the factors that influence the probability of a positive WTP decision take into account all socio-metric variables:

$$\ln\left(\frac{P_i}{1 - P_i}\right) = a + b_1 GENDER + b_2 AGE + b_3 EDUC + b_4 INC + u_i \qquad (2)$$

The estimation results, see Table 3, suggest that only gender affects WTP significantly at the appropriate level of significance ($p<0.05$). According to the regression results, women are more likely to state a positive WTP than men. The level of education seems to be positively related to decision to pay but is significant on 10 per cent level only. The coefficients of age and income have the expected sign, but they are not statistically significant. Estimation of possible interactions between variables does not give significant results so the final regression includes only gender and the education level of the respondent.

	Coeff	S.E.	Z	p>z	Odds ratio
Constant	-0.014	0.392	-0.040	0.971	
Gender	**0.708**	**0.169**	**4.200**	**0.000**	**2.029**
Age	-0.061	0.044	-1.380	0.166	0.941
Income	0.030	0.049	0.600	0.550	1.030
Education	0.163	0.094	1.740	0.083	1.177

Tab. 3: *The influence of socio-metric variables on WTP >0, logit model*
Source: Statistical analyses of Tallinn Zoo CVM

Dropping the constant and two variables, makes the level of education significant at appropriate level ($p<0.05$). The probability of a positive decision to pay can be calculated on the basis of equation (3):

$$\ln\left(\frac{P_i}{1-P_i}\right) = 0.638\,GENDER + 0.165\,EDUC + u_i \tag{3}$$

The odds ratio, see Table 4, shows that the growth of education by one level increases the likelihood of pay by approximately 1.2 times, being a woman increases the likelihood by about 1.9 times.

	Coeff	S.E.	Z	p>z	Odds ratio
Gender	0.638	0.120	5.330	0.000	1.892
Education	0.165	0.061	2.710	0.007	1.180

Tab. 4: *The influence of socio-metric variables on WTP >0, logit model2*
Source: Statistical analyses of Tallinn Zoo CVM

4.2. Influence of Socio-Metric Variables on WTP Amount

In the second step, we examine the influence of the socio-metric variables to the amount of WTP. Analyses indicate that the appropriate form is a semi-logarithmic regression equation. The regression result suggests that all socio-metric variables, except education, have a significant impact on the amount of WTP. The WTP amount is positively related to average income as expected, and age is negatively related. Men are more likely to state a higher WTP than women once the payment decision has been done. Education, which was an important factor making the payment decision does not have significant influence to the WTP amount. However, further analysis shows that there is significant spread in the WTP amount among those with higher education. Those with higher education and low income, have a low WTP, while those with higher education and high incomes have a high WTP.

In the final model for estimating WTP amounts based on socio-metric features we drop education as it is non-significant, see equation (4).

$$\ln(WTP_i) = a + 0.303GENDER_i - 0.080AGE_i + 0.216INC_i \qquad (4)$$

Replacing gender, age and income with average indicators of Tallinn population gender 0.45 in [0,1] scale, age 4.4 in [1,7] and income 4.4 in [1,8] the WTP is approximately 6€ which is equivalent to the median, but smaller than average WTP of the sample data. Table 5 shows that the size of the amount that people are willing to pay is statistically dependent on gender, age and income. The goodness of fit (Adj. R^2) is low. However, a low R^2 is common in cross-sectional data.

	Coeff	S.E.	T	p>t	95% confidence interval	
Constant	1.031	0.219	4.700	0.000	0.600	1.461
Gender	0.303	0.097	3.130	0.002	0.113	0.493
Age	-0.080	0.026	-3.130	0.002	-0.131	-0.030
Income	0.216	0.025	8.500	0.000	0.166	0.266
Summary statistics	F(3,797)=26.12 Prob>F=0.000 AdjR2=0.12 Number of obs=801					

Tab. 5: *The influence of the socio-metric indicators to the WTP amount, OLS results*
Source: Statistical analyses of Tallinn Zoo CVM

4.3. Estimation of consumer surplus

The estimation of the aggregated Hicksian demand curve for animal wellbeing of Tallinn's adult population is based on the actual distribution of WTP amounts obtained from the survey. The results are generalized to the proportion of the population with positive WTP, which is 81 per cent i.e. about 265,964 persons 18 years of age or older in Tallinn as of January 1st, 2011. Based on the distribution of WTP, the exponential model is the most appropriate functional form, for presenting the demand curve, see equation (5)

$$WTP = ae^{-bX} \tag{5}$$

where WTP is the euro value of the willingness to pay; X is the number of people in thousands willing to pay this amount; and *a* and *b* the parameters under estimation. The results of the regression, using the least squares method are shown in Table 6. Both parameters are statistically significant ($p<0.01$) and the value of coefficient of determination ($R^2=0.98$) indicate a very high goodness of fit.

	Coeff.	S.E.	T	p>t	95% confidence interval	
a	90.383	3.139	28.79	0.000	83.940	96.825
b	0.0189	0.002	22.15	0.000	0.030	0.037

Tab. 6: *Parameter estimates of the demand curve, OLS*
Source: Statistical analyses of Tallinn Zoo CVM

Based on the estimation we can substitute *a* and *b* into equation (5) and obtain the WTP, i.e. the demand curve for animal wellbeing of Tallinn adult population:

$$WTP = 90.383e^{-0.0189X} \tag{6}$$

By integrating (6) we find the consumer surplus (CS) of the adult population:

$$CS = \int_{X_1}^{X_2} WTP = \int_{x_1}^{x_2} ae^{-bx} = -\frac{a}{b}\left(e^{-b_{X_2}} - e^{-bx_1}\right)dx \cong \frac{a}{b} \tag{7}$$

where $x_1=0$ and x_2 are the number of people with positive WTP i.e. about 266 thousand. Replacing the values of parameters *a* and *b*, the estimated aggregated WTP amount is calculated as:

$$CS \cong \frac{a}{b} = \frac{90.383}{0.0189} = 4782.17 \cong 4.78 \; million \; € \tag{9}$$

The resulting consumer surplus of the Tallinn population from improving animal wellbeing is thus about 4.8 million Euros annually and 24 million during a five year period, which is not sufficient to cover the investments of 40 million Euros that was foreseen during the time period 2008–2012. Figure 2, below depicts the consumer surplus of Tallinn adult population, shown as WTP replies and the fitted demand curve.

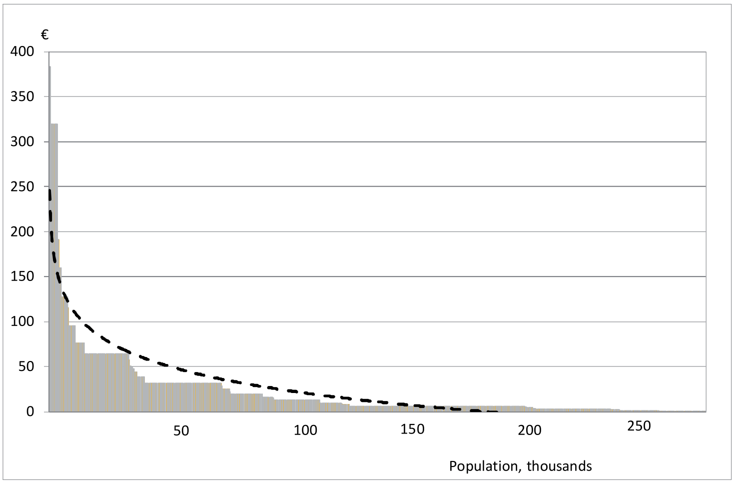

Fig. 2: The demand curve of Tallinn and Estonia for improving animal wellbeing at Tallinn Zoo
Source: Authors' calculations based on Tallinn Zoo CVM

However, as discussed initially, the Tallinn Zoological Gardens attracts visitors from all over Estonia, implying that the increase in human welfare from improving animal wellbeing is experienced by a significantly larger number of people than the adult population of Tallinn. By expanding the results to the adult population of Estonia, we have about 886.3 thousand people with a positive WTP. Fitting equation (6) to the whole adult population we find that the annual WTP will exceed the annual cost of investments.

$$CS \cong \frac{a}{b} = \frac{90.383}{0.056} = 1613.983 \cong 16.14 \textit{ million } € \tag{10}$$

Conclusions

The benefit of zoo animal wellbeing has been defined as the value humans place on the level of living conditions of zoo animals. By choosing this approach it is possible to assess whether the increase in human benefits is large enough to cover the costs of providing adequate accommodation to all zoo inhabitants by definition of the development strategy of Tallinn Zoological Gardens. If animal wellbeing at zoos is a pure use value, it would be sufficient to ask zoo visitors. Since there might be a broader interest in the wellbeing of zoo animals it is important to try to distinguish between the value of visitors and possible non-use value expressed by those who do not visit the zoo. The analysis shows that improvement of animal wellbeing at Tallinn Zoo has non-use value. The non-use value was found to be 7.7 € annually, while the average annual WTP is 17.3 €, which makes non-use value about 45 per cent of the sum of the use and non-use value. It was also found that average WTP increases with the frequency of visits. This could be a result of the fact that improved animal wellbeing enhances the quality of a zoo visit. Another interesting finding is that the propensity of being a potential protest voter increases with frequency of visits.

The statistical analysis shows that gender and education have a significant impact on the probability of stating a positive WTP rather than reporting zero WTP. In the statistical analysis that covers only positive WTP responses, all socio-metric variables, except education have a significant influence on the amount people are willing to pay. As can be expected from theory, income is positively correlated with the WTP amount. Once the payment decision has been made, men are more likely to state a larger WTP amount than women. The impact from age is negative, but at the same time the size of the coefficient is small.

By aggregating the WTP of the Tallinn population and estimating the consumer surplus we find that the costs of planned investments do not cover the resulting benefits. Since Tallinn Zoo attracts visitors from all over Estonia, the increase in human welfare from improving animal wellbeing is experienced by a considerably larger number of people than the Tallinn population. By expanding the WTP to the adult population of the whole of Estonia we find that planned investments are motivated from a cost benefit perspective.

During the economic crisis, funds from the owner of Tallinn Zoological Gardens, which is the city of Tallinn, dropped significantly. The cuts concern primarily investment funds. In 2008 the Tallinn city contributed by about one million Euro. In 2009 the city's investment contribution was approximately half of that sum and in 2010 the contribution was zero. It thus seems as the dependence on one major funder has made the zoo extremely vulnerable to the impact of the economic crisis. Not only vulnerability, but also the fact that Tallinn Zoo has a catchment area much larger than Tallinn city, calls for further study in order to find a sustainable solution for zoo financing.

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Appendix 6. “The Foregone Recreation Value of Lake Ülemiste”

THE FOREGONE RECREATION VALUE OF LAKE ÜLEMISTE

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Abstract

Since Soviet times Lake Ülemiste has been closed to public access. The current practice of Tallinn may entail unnecessary losses of benefits to the local population. The aim of this paper is to find the value of the foregone benefits. In order to find this value, a contingent valuation (CVM) survey was conducted involving a sample of the adult population of Tallinn.

According to the survey the average willingness to pay is 6.6 Euro and the recreational benefits foregone were estimated to 1.8 million Euros annually. In order to safeguard the quality of the drinking water, additional measures may be needed. Discounted over a 30 year period allows investments of a maximum of 26 million Euro. Applying the current investment plan of Gothenburg to Tallinn shows that the recreational value of opening the lake to the public is sufficiently large to cover Gothenburg's coal filter investments to be carried out in Tallinn.

Keywords: contingent valuation, recreation value, drinking water reservoir

JEL Classification: C25, Q25, Q26, Q51

1. Introduction

It has been found that provision of parks and recreational areas in urban neighbourhoods have significant positive health impacts on the urban population (see e.g. Foster *et al.*, 2005, Duncan and Mummery, 2004 and Suminski *et al.*, 2005). In order to ensure that cities recognise the positive values of green area provision, the EU has supported various initiatives, including the COST Action C11 and an internet site about green structure. However, not all recreational areas are open to the public. Located only 2 kilometres from the city centre, Lake Ülemiste, which is the largest lake in the Tallinn area has since the Soviet times been closed to the public (RT I 1994, 40, 655). Recently the restrictive zone that surrounds the lake was extended by a decision of the Environmental Board (Tallinna vesi, 2010). The main motivation is to protect Tallinn's drinking water reservoir.

In comparison to surface water reservoir protection policies in other countries, including neighbouring Finland and Sweden, the protective measures of Lake Ülemiste seem exaggerated. In Sweden, the City of Gothenburg, which is of a similar size as the City of Tallinn, is supplied by drinking water from Delsjöarna Lakes and the River Göta Älv (Göteborgsregionen, 2003). These water bodies are open to the public. While Delsjöarna Lakes are located in a forest area, the River Göta Älv serves as a fairway and there are polluting industries located in its vicinity. This does not imply that Sweden does not protect drinking water reservoirs, but

Swedish policies do not regard recreation use as a threat. Protective measures include restrictions in the use of pesticides, petroleum products, spread of manure, installation of sewage systems and waste dumping (NFS, 2003:6).

Judging from practices elsewhere, the restrictions that are imposed on Lake Ülemiste imply a loss in welfare to the population of Tallinn. The aim of this paper is to find the value of the foregone benefits and also to discuss the costs of possible additional water protection measures. However, this study does not investigate the range of additional costs that may stem from recreational use of Lake Ülemiste. Instead data on the costs of additional water protection measures are based on current plans of the City of Gothenburg (Göteborgs stad, 2010). The benefits foregone are measured according to what inhabitants themselves would be willing to pay to make the lake accessible for recreation. In order to find the value of the loss in benefits, a contingent valuation (CVM) survey was conducted in the autumn 2010. Previous research that has estimated recreational values of lakes have generally applied travel cost estimates (see e.g. Fleming and Cook, 2008, Okrazai, 2008). CVM studies on lakes have instead set out to estimate the willingness to pay for an improvement in the water quality of the lake (see e.g. Carson and Mitchell, 1993 Monarchova and Gudas (2009). Since Lake Ülemiste is closed, the travel cost method has not been available for finding the recreational value of this study.

In the next section we present a general overview about non-market values. After that, in Section 3, we report the details of the survey and provide descriptive statistics. In section 4 we carry out the statistical analysis and estimate the benefits foregone. Section 5 presents the investment programme of Gothenburg and uses cost-benefit analysis to assess whether the investments of Gothenburg can be motivated in Tallinn. In section 6 we conclude the study.

2. The value of a non-market resource

The value of a good or a service is determined either by markets or assessed by different methods developed for revealing individual preferences for non-marketed goods. Value, according to economic theory, relates to the utility individuals derive from goods and services. The choices individuals make reflect their preferences and concerns. When individuals make a choice, either in relation to what to buy or how to spend their time, they appraise the value they will receive from a particular choice. Many goods are not subject to market transactions and they can be enjoyed for free, e.g. bird watching and swimming in a lake. In his seminal paper, Krutilla (1967) went even further by suggesting that people receive utility from natural assets just because they exist. Thus, utility may originate from the pure knowledge of conservation of a certain wilderness area. Through human choices the value of these activities can be assessed. For an overview of non-market valuation see e.g. Smith, 1993 or Freeman, 2003.

The closure of Lake Ülemiste implies that the recreational value and possibly the aesthetic value of the lake currently are cut off from use. Allowing recreational use of Lake Ülemiste would imply an increase in the indirect use value. Table 1 classifies the types of economic values that can be attributed to the benefits of Lake Ülemiste.

Table 1. The economic values of Lake Ülemiste and their expressions

Economic value	**Category**	**Typical expressions of the value**
Non-use value Existence value	General ecological	Provision of conditions for life Conservation of species
Non-use value Intrinsic value	General ecological	Provision of water Preservation of pure water resources
Non-use value Intrinsic value	Biotic regulation	Conservation of species and genetic resources Provision of multiplicity of ecological systems
Non-use value Bequest value	Future value	Provision of biodiversity and pristine environment in the future
Use value Option value	Future value	Preservation to allow future drinking water supply, recreation, research, etc.
Indirect use value	Human use of ecosystem services	Regulation of water, prevention of erosion etc.
Indirect use value	Recreational (including health impacts)	Supply of recreational services (e.g. swimming, skating, boating, walking on the shoreline)
Indirect use value	Educational and scientific	Opportunities for educational and research work
Indirect use value	Cultural-historical	Lake mythology
Indirect use value	Aesthetic	Recognizing beauty of landscapes and natural objects
Direct use value	Agricultural	Fishing
Direct use value	Industrial	Production of drinking water

The contingent valuation method (CVM) is a survey method that seeks to elicit people’s preferences for changes in non-market good provision by finding the amount of money people are willing to pay in order to receive the change in question. The value attached to the object by the respondents in the form of the willingness to pay is contingent in relation to the constructed or simulated market (or market scenario) in the questionnaire (Portney, 1994). If there is no actual market for some goods, it has to be created hypothetically. The hypothetical scenario is then presented to people and they are asked how much money they would agree to give up if the change was undertaken, alternatively to avoid the change. Theoretically, the maximum amount of money an individual is willing to pay for a welfare increasing change is equivalent to the amount that he or she would give up while keeping his or her utility constant (Freeman, 2003).

The indirect approach or revealed preference (RP), estimates the value by studying human behaviour in complementary markets, i.e. money and time spent on travelling to a lake (travel cost method) or how the local environment affects housing prices in urban areas (hedonic model). Use values can be estimated by direct and indirect methods. However, since human behaviour is a prerequisite for the travel cost and hedonic approaches they cannot elicit non-use values. Non-use values can be estimated only by using direct methods (Freeman, 2003).

3. Ülemiste CVM Survey

As a part of their undergraduate studies in Economics and Business Administration at Tallinn University of Technology, students taking environmental economics were asked to distribute ten questionnaires each to a sample of different age groups representing the 18+ population of Tallinn. Because participants received course credits, response rates were high: 95 per cent. In total, 1,523 questionnaires were returned out of the 1,600 that were originally distributed. Since 282 questionnaires lacked a willingness to pay (WTP) statement, 1,241 replies remained for further analysis. Apart from overrepresentation of the two youngest age groups and underrepresentation of age groups above 60 years, the sample is representative to the Tallinn population, see Figure 1, below. Since the total deviation with respect to age groups is only about 4.7 per cent, weighting was not undertaken prior to analysis.

The questionnaire used an open ended WTP question including a reminder that the respondent should consider his or her budgetary means when replying. In order to reduce the complexity of stating the recreational value of a lake area the respondents have never visited, the WTP question was stated in terms of the annual willingness to pay for additional water protection measures that would certify maintenance of drinking water quality.

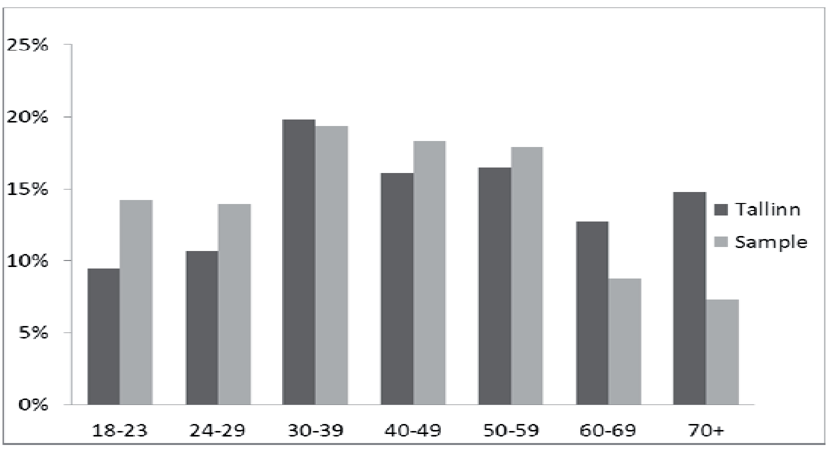

Figure 1. Age distribution of the sample and in Tallinn. Sources: statistics Estonia and Ülemiste CVM survey.

The survey included three attitude questions about the idea of opening Lake Ülemiste for the public. The questions were:

Q1: Do you think, that opening Lake Ülemiste for the public does not jeopardise the quality of the drinking water in Tallinn?
Q2: Do you support the idea that Lake Ülemiste should be opened for Tallinn inhabitants?
Q3: In case Lake Ülemiste was opened, would you use the opportunity to spend free time at the lake and its surroundings?

Table 2. Attitudes towards opening Lake Ülemiste

	Q1	Q2	Q3
Yes	28.5%	32.1%	34.6%
No	51.7%	56.8%	49.4%
Don’t know	19.6%	11.0%	16.0%
Total	1,520	1,522	1,523

There were relatively many “don’t know” replies to all three attitude questions, see Table 2. The first question whether recreational use would be a threat to water quality had the highest share of “don’t know” replies. It also seems as judging whether or not to visit Lake Ülemiste if made accessible (Q3), received a high share of “don’t know” replies.

About one in three supports the idea of recreational access to Lake Ülemiste while more than one half of the respondents oppose to the idea. The share of those opposed to opening the lake (“no” to Q2), is higher than the share of those who express concern for the drinking water (“no” to Q1). It is interesting to compare the result to a survey about attitudes to bathing in Delsjöarna Lakes. In Gothenburg, the attitudes are much more favourable towards recreational use, possibly because the lakes have never been restricted to public access. About 65 per cent of the respondents were of the opinion that bathing should be allowed everywhere and almost 90 per cent reported that they had visited Delsjöarna Lakes (Morrison and Bost, 2008).

In the Ülemiste CVM survey 27 per cent of those respondents that were opposed to the suggestion of opening the lake to the public, stated that they would not visit the lake if it was opened, but still reported a positive WTP. The interpretation of these responses, which make up about 2 per cent of the observations, could be that the concerns about a potential negative impact on drinking water quality from recreational use results in a willingness to pay for precautionary measures, if public access is allowed.

The average willingness to pay of the 1,241 respondents who gave a WTP reply is 6.6 Euro per year. Assuming that those who did not fill in a WTP response had stated a zero WTP gives an average WTP of 4.3 Euro, which is about 65 per cent of the mean of those who had filled in WTP statements. Table 3 shows the average WTP with respect to socio-metric variables. According the averages, men were prepared to pay more than women. It is not possible to differ between the WTP of those with secondary and higher education. However, those who have primary and secondary technical education generally gave lower WTP values than those with the two aforementioned education levels. While younger age groups have a higher average WTP than those in older age groups, those with higher incomes generally gave a higher average WTP than those belonging to lower income groups.

Table 3. Willingness to pay (WTP) with respect to socio-metric variables

		Average WTP, €	Difference from total average, %
Gender	Male	8,2	124,4
	Female	5,3	80,6
Education	Primary	4,0	61,2
	Secondary	7,3	110,4
	Secondary technical	5,2	78,6
	Higher	7,3	110,0
Age	18-23	8,7	131,9
	24-29	8,3	125,8
	30-39	7,5	114,0
	40-49	6,4	96,6
	50-59	4,4	67,4
	60-69	5,2	78,4
	> 70	4,1	61,5
Average monthly income (net), €	<128	3,6	55,0
	128-255	4,6	69,7
	256-383	4,2	64,0
	384-511	5,8	88,2
	512-703	7,0	106,2
	704-958	10,3	155,5
	959-1278	6,5	97,9
	>1278	12,3	185,7
Total average		6,6	100,0

4. Statistical analysis

The statistical analysis of data is carried out in two steps. In the first step we use a binary logit regression to allow us to assess the influence of socio-metric variables to the decision to pay or not to pay. In the second step, an OLS regression is applied to the sub-sample that has a positive WTP in order to determine the relationship between the stated amount and the socio-metric variables. Finally the positive WTP replies are used as an input for finding the demand curve and the consumer surplus.

4.1. Determination of a positive willingness to pay

Since survey data fits a standard logistic distribution, a logit-model is applied for describing the relationship between the binary dependent variable and the explanatory variables. The probability (P_i) that an individual states a positive WTP is expressed as:

$$P_i = \Pr(y_i = 1 \mid X_i) = \frac{1}{1+e^{-\beta' X_i}} \tag{1}$$

where y_i is the binary dependent variable: ($y_i = 1$, WTP>0, and $y_i = 0$, WTP=0), X_i is the vector of independent variables and β_i is the vector of parameters.

$$\ln\left(\frac{P_i}{1-P_i}\right) = \beta_0 + \beta_1 GENDER + \beta_2 AGE + \beta_3 INCOME + \beta_4 EDUCATION + u_i \tag{2}$$

where $\frac{P_i}{1-P_i}$ is the odds ratio, $\ln\left(\frac{P_i}{1-P_i}\right)$ is the log odds ratio and u_i is the error term, which is assumed to have a zero mean.

The interpretation of the resulting logit model parameters is not straightforward, as the estimated probability is not a linear function of the parameters. It is only possible to estimate the direction of the correlation, i.e. in case $\beta_i>0$ and that the value of X_i is increasing the probability increases, and vice versa. By using the odds ratio there will be a direct relationship between the change and its influence on the dependent variable. Table 4 shows the results of the regression.

Table 4. The influence of sociometric variables on WTP >0, logit model

	Coeff (β)	S.E.	Wald	Probability	Exp(β)
Constant	0.013	0.257	0.003	0.959	1.013
Gender	-0.078	0.124	0.399	0.527	0.925
Age	**-0.182**	**0.034**	**29.316**	**0.000**	**0.834**
Income	0.050	0.035	2.108	0.146	1.052
Education	0.105	0.072	2.143	0.143	1.111

According to the logit regression, age is the only statistically significant parameter and it is significant on the 1 per cent level. The negative β_i implies that the choice of stating a positive WTP depends negatively on age. The log odds ratio that is shown in the column Exp(β) shows that the increase in age by one age group reduces the probability of a positive WTP by 0.834 times. Individuals belonging to the oldest age group (70+) are $0.834^7=0.281$ times less likely to state a positive WTP than individuals belonging to the youngest group of 18-23 years.

The significant influence of age on the payment decision is potentially explained by the fact that young people have grown up in the free Estonia and are therefore more prone to take their rights for granted. Those who grew up during the Soviet time are more accustomed to restrictions and might therefore have higher acceptance for the closure of the lake.

4.2. Influence of socio-metric variables to the willingness to pay amount

In the second step, we examine the influence of the socio-metric variables to the amount of WTP. The subsample of positive WTP is used in the following OLS regression model:

$$\ln(WTP) = B_0 + B_1 GENDER + B_2 \ln(AGE) + B_3 \ln(INCOME) + B_4 \ln(EDUC) + u_i \quad (3)$$

where gender is a dummy variable (male=1, female=0) and all other variables are categorical variables.

Table 5 shows the regression result:

Table 5. The influence of socio-metric variables on the WTP amount, OLS model

	Coeff(B)	S.E.	t-ratio	Probability	95% conf. interval	
Constant	**3.915**	**0.206**	**19.006**	**0.000**	**3.511**	**4.320**
Gender	0.076	0.111	0.689	0.491	-0.142	0.294
Age	**-0.203**	**0.094**	**-2.156**	**0.031**	**-0.388**	**-0.018**
Income	**0.411**	**0.116**	**3.547**	**0.000**	**0.183**	**0.638**
Education	0.226	0.170	1.328	0.185	-0.108	0.560
Adj R^2	0.19					

The table shows that the size of the amount that people are willing to pay is statistically dependent on age and income. In correspondence to the previous regression result, age has a negative sign. Income is positively correlated to the sum that people are willing to pay, which is what we expect from theory. The influence from gender and education are not statistically significant. The goodness of fit (Adj. R^2) is relatively low. However, a low R^2 is common in cross-sectional data.

4.3. Estimation of consumer surplus

In order to estimate the loss of the benefits from the closure of Lake Ülemiste we need to find the consumer surplus of the foregone recreational value. There are several different ways to calculate the consumer surplus. The open-ended WTP question that asks for the actual amount of willingness to pay allow us to calculate the consumer surplus by multiplying the average or median WTP obtained from the sample with the relevant population. However, such calculations tend to be inexact as they either overestimate or underestimate the consumer surplus and we decided to find the consumer surplus by fitting a demand curve. The construction of an aggregated demand curve for the adult population of Tallinn is based on the actual distribution of WTP amounts obtained in the survey. The results are generalized to the proportion of the population with positive WTP, which is 47.4 per cent i.e. about 155,800 persons 18 years of age or older in Tallinn on January 1st, 2010.

Based on the distribution of WTP, the exponential model is the most appropriate functional form, for presenting the demand curve, see equation (4)

$$WTP = ae^{-bX} \tag{4}$$

where *WTP* is the amount of willingness to pay, *X* is the number expressed in thousands of people willing to pay, and *a*, *b* the parameters under estimation. The results of the estimation, using the least squares method are shown in Table 6. The value of R^2=0.96 indicates a very high goodness of fit. In addition, both parameters are statistically significant.

Table 6. Parameter estimates of the demand curve, OLS regression (R^2=0.96)

	Coeff	S.E.	t-ratio	Significant	95% conf. interval	
A	69.01	0.640	108.12	0.000	67.76	70.26
B	0.038	0.000	76.821	0.000	0.039	0.037

Based on the estimation we can substitute *a* and *b* into equation (4) and obtain:

$$WTP = 69.01e^{-0.038X} \tag{5}$$

Figure 2 shows the graph of equation (5).

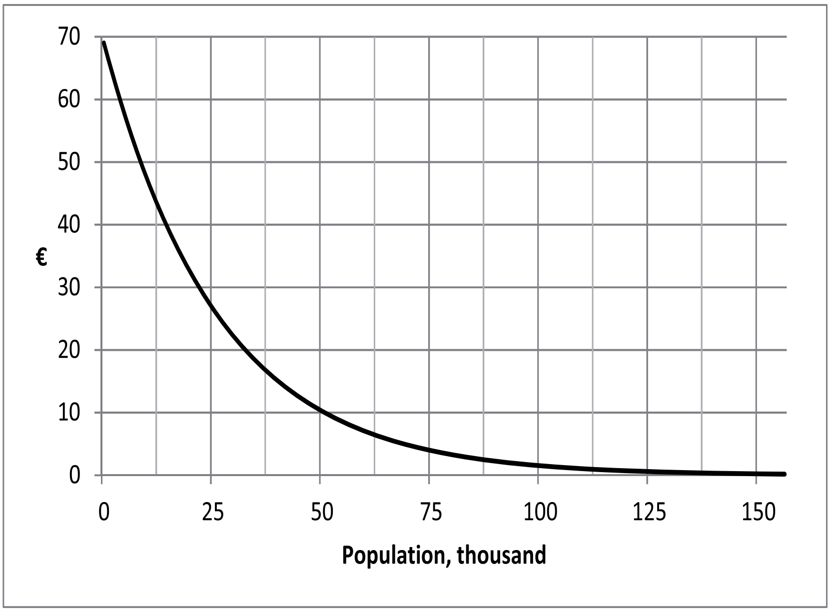

Figure 2. The demand curve of Tallinn population 18+ for getting access to Lake Ülemiste for recreational purposes.

The consumer surplus (CS) is the area below the demand curve. For this purpose we integrate the demand curve, see equation (6).

$$CS = \int_{X_1}^{x_2} WTP = \int_{x_1}^{x_2} ae^{-bx} = -\frac{a}{b}\left(e^{-bx_2} - e^{-bx_1}\right)dx \cong \frac{a}{b} \tag{6}$$

where x_1=0 and x_2 is the number of people with positive WTP (155.8 thousand). Replacing the values of the parameters *a* and *b* we find that the estimated consumer surplus is about 1.8 million Euro.

$$CS \cong \frac{a}{b} = \frac{69.01}{0.038} = 1816.05 \cong 1.8 \textit{ million } € \tag{7}$$

The interpretation is that the closure of Lake Ülemiste entails an annual loss in welfare of the Tallinn population, which amounts to about 1.8 million Euros.

5. Costs of additional measures

The estimate of the consumer surplus indicates that the benefit foregone is relatively high according to the willingness to pay of the Tallinn population. Although Ülemiste water purification plant uses up-to-date technology for drinking water production (Tallinna Vesi 2010), we cannot exclude that recreational use affects raw water quality. If Lake Ülemiste is opened to the public, there might be a need to invest in additional water purification measures. In order to determine whether such investments are needed and their range requires further investigation and in-depth studies. However, such studies are out of the scope of the current paper. As a proxy of possible investments, we will report about the current plans in Gothenburg and estimate whether similar investments would pass the cost benefit criteria for Tallinn.

Starting from the annual benefits of opening Lake Ülemiste of 1.8 million Euro, we calculate the benefits during the assessment period. According to the EU guidance on cost-benefit analysis during the programming period 2007-2013 investments in water and sewage plants should be evaluated during a 30 year period (European Commission, 2006). The same document suggests a discount rate of 5.5 per cent. Assuming that Lake Ülemiste will be opened in 2012, the sum of the net present value of benefits will be about 26 million Euro.

In Gothenburg there are plans to extend the Lackarebäck water purification plant to a capacity of 171,000 m^3 water per day. Currently this plant has the same maximum capacity as Ülemiste water purification plant, i.e. about 120,000 m^3 water per day. (Göteborgs stad, 2010 and Tallinna Vesi, 2010). Besides aiming at increasing the production capacity at Lackarebäck, Gothenburg will upgrade the water purification process. Investment costs of ultra-filters at Lackarebäck water purification plant have been estimated by the City of Gothenburg to approximately 70 per cent of 700 MSEK in 2009 value. The remaining i.e. 210 MSEK stands for the additional coal filter investments. The annual running costs are expected to be 9 and 1 MSEK respectively (Göteborgs stad, 2010, WSP 2010). The investment is calculated into Euros using the annual average SEK to Euro exchange rate, which was 10.62 in 2009 (Sveriges Riksbank, 2011).

Assuming that investments will be undertaken in Tallinn at the same pace as at Lackarebäck water purification plant, but scaled down to 70 per cent in order to consider the capacity increase, a provisional cost benefit analysis is carried out. Using the same assumptions as for benefits, we find that the sum of the net present value of ultra-filter investments and its running costs will be about 33 million Euros and the corresponding for coal filters about 15 million Euros, see Table 7. Assuming that Lake Ülemiste will be opened in 2012, the sum of the net present value of benefits will cover the investments and running costs of coal filters. Multiplying benefits by 0.65 thus taking into account non-responses, we receive a low level estimate. Using this low level benefit estimate will not alter the result.

Table 7. Provisional cost-benefit analysis, present values, Euro millions

Year	Discount factor	Investment		Running cost		Benefits
		Ultra	Coal	Ultra	Coal	
2011	1.000	2.234	13.840			
2012	0.948	5.753			0.062	1.706
2013	0.898	4.559		0.133	0.059	1.617
2014	0.852	0.393		0.126	0.056	1.533
2015	0.807	3.724		0.120	0.053	1.453
2016	0.765	3.530		0.227	0.050	1.377
2017	0.725	6.692		0.215	0.048	1.305
2018	0.687			0.408	0.045	1.237
2019	0.652			0.386	0.043	1.173
2020	0.618			0.366	0.041	1.112
2021	0.585			0.347	0.039	1.054
2022	0.555			0.329	0.037	0.999
2023	0.526			0.312	0.035	0.947
2024	0.499			0.296	0.033	0.897
2025	0.473			0.280	0.031	0.851
2026	0.448			0.266	0.030	0.806
2027	0.425			0.252	0.028	0.764
2028	0.402			0.239	0.027	0.724
2029	0.381			0.226	0.025	0.687
2030	0.362			0.214	0.024	0.651
2031	0.343			0.203	0.023	0.617
2032	0.325			0.193	0.021	0.585
2033	0.308			0.183	0.020	0.554
2034	0.292			0.173	0.019	0.525
2035	0.277			0.164	0.018	0.498
2036	0.262			0.156	0.017	0.472
2037	0.249			0.147	0.016	0.447
2038	0.236			0.140	0.016	0.424
2039	0.223			0.132	0.015	0.402
2040	0.212			0.126	0.014	0.381
Sum		26.885	13.840	6.360	0.945	25.800

6. Conclusions

Provision of recreational areas and safe drinking water are municipal tasks. In Tallinn, Lake Ülemiste, located only 2 kilometres from the city centre, is closed to the public in order to protect the city's main fresh water reservoir. In this paper we have reported on a contingent valuation (CVM) survey that was undertaken in order to estimate the foregone benefits of lake closure. According to the analysis of data about 1.8 million Euros are foregone annually. Another way to express this is that the costs of drinking water production are 1.8 million Euros higher than measured by the annual costs of water treatment and its distribution to households.

The attitude questions of the CVM survey showed that about one in three supports the idea of allowing recreational access to Lake Ülemiste, more than one half oppose to the idea while the remaining respondents did not express any opinion. Out of those who filled in the willingness to pay question, 47 per cent were willing to pay for making Lake Ülemiste accessible to the public. The analysis showed that age is the only statistically significant variable determining whether a person chooses to state a zero or a positive willingness to pay. A potential explanation to why young people are more likely to contribute could be that young people have grown up in a free Estonia and are more conscious of their rights. The analysis of the determinants of the size of the amount that people are willing to pay showed that young people and people with higher income are prepared to pay more.

According to the city's practices there is no trade-off between drinking water provision and recreation use. This is in contrast to water protection policies elsewhere. A change in views would make possible public access to a valuable recreation area within the immediate neighbourhood of the city centre. Apart from the benefits that we have identified here, there may be additional positive impacts from public access to the Ülemiste area since provision of recreational areas in an urban setting have significant positive health impacts on the urban population. However, before taking such a decision further investigation and in-depth studies will be needed in order to determine whether opening the lake would require additional investments into water purification measures. The paper has shown that the discounted benefits amount to almost 26 million Euros. This sum significantly exceeds the investment cost of coal filters in Gothenburg.

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Appendix 7.

ELULOOKIRJELDUS

1. Isikuandmed

Ees- ja perekonnanimi	Sirje Ilona Pädam
Sünniaeg ja –koht	21.06.1962 Stockholm (Rootsi)
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2. Kontaktandmed

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3. Hariduskäik

Õppeasutus (nimetus lõpetamise ajal)	Lõpetamise aeg	Haridus (eriala/kraad)
Tallinna Tehnikaülikool, majandusteaduskond	2003	magister
Stockholmi Ülikool, rahvamajandus	1989	bakalaureus, rahvamajandus
Östra Real (Stockholm)	1981	gümnaasium

4. Keelteoskus (alg-, kesk- või kõrgtase)

Keel	Tase
Inglise	Kõrgtase
Rootsi	Kõrgtase
Soome	Kesktase
Eesti	Kõrgtase

5. Täiendusõpe

Õppimise aeg	Täiendusõppe teema ja õppe korraldaja
2000	Projekti juhtimine, Wenell Management AB, Stockholm

6. Teenistuskäik

Töötamise aeg	Tööandja nimetus	Ametikoht
2007-…	WSP Sverige AB	Majandusanalüütik
2011- ...	Tallinna Tehnikaülikool, Majandusteaduskond, Avaliku sektori majanduse instituut, Keskkonnaökonoomika õppetool;	Lektor
2006-2007	WSP International,	Majandusekspert, peaökonomist
2008 – 2009	Tallinna Tehnikaülikool, Majandusteaduskond, Majandusuuringute teaduskeskus;	Erakorraline teadur
2005-2006	Tallinna Tehnikaülikool, Majandusteaduskond, Avaliku sektori majanduse instituut, Avaliku sektori ökonoomika õppetool;	Teadur
2001-2003	Tallinna Tehnikaülikool, Majandusteaduskond, Avaliku sektori majanduse instituut, Avaliku sektori ökonoomika õppetool;	Assistent
1997-2006.	Konsultatsioonifirma Inregia AB, Stockholm	Majandusanalüütik
1996	Rootsi tööturuministeerium	valitsuse komitee abi-sekretär töötu abiraha uuringus
1991-1992	Tallinna Pedagoogikaülikool,	Rootsi keele õppejõud
1991	Rootsi Välisministeerium,	Balti majandusuuringute assistent
1990	Rootsi Energiaamet,	Konsultant
1987-1990 1993-1995	Stokholmi ülikool, rahvamajanduse õppetool	Lektori abi Teadur

7. Teadustegevus:

Tegevus teaduseksperdina:
2008-2009 WSP Sweden, Suure väina liiklusühenduse kava koostamine, majandusekspert

2008-2008 Stockholmi regiooni plaan (RUFS 2010), majandusekspert ja kaastoimetaja
2007-2008 Regionaalplaani koostamine Tripoli regioonile, vastutav majandusekspert
2006-2006 Rootsi keskkonnaamet, Majanduskasv versus keskkond, Hndamise meetodi väljatöötamine konfliktsete arengueesmärkide väljaselgitamiseks regionaalplaneeringus, majandusekspert

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Juhendatud teadustööd:

Kristiina Kaasik, magistritöö, (juhendaja) **Sirje Pädam**, "Keskkonnajuhtimissüsteemi rakendamise uuring: Eesti ettevõtete näitel". Tallinna Tehnikaülikool 2003.

Ave Matsalu, magistritöö, (juhendaja) **Sirje Pädam**, "Ummikumaksu mõju Tallinna liiklusele". Tallinna Tehnikaülikool 2011.

8. Kaitstud lõputööd

Magistritöö:
Sirje Pädam, *Sotsiaalmajanduslik tasuvusanalüüs ja jätkusuutlik areng transpordi näitel*. Juhendaja prof. Alari Purju. Tallinna Tehnikaülikool, 2003.
Bakalaureusetöö:
Sirje Pädam, *Bättre miljö till lägre kostnad*. [Parem keskkond vähema kulu eest] Juhendaja Lars Vahtrik. Stockholmi ülikooli majandusteaduskond. 1989.

9. Teadustöö põhisuunad

- Majandusteadus (Keskkonna- ja transpordiökonoomika)
- Regionaalmajandus

10. Teised uurimisprojektid

Biogaas transpordikütusena (2011-2012)
Tasuvusuuring (CBA) ühistranspordi teenuse pakkumise parandamiseks Lääne-Harjumaal ja Läänemaal (2011)
Regulaarse sõiduplaani kõrvalekalde kulu (2010-2011)
Raudteetunnelite ohutuse tagamise sotsiaalmajanduslik väärtus (2009-2010)

Appendix 8.

CURRICULUM VITAE

1. Personal data

Name	Sirje Ilona Pädam
Date and place of birth	June 21st, 1962, Stockholm

2. Contact information

Address	Müürivahe 33-2 10140 Tallinn
Phone	+372 -6443343
E-mail	Sirje.Padam@tseba.ttu.ee

3. Education

Educational institution	Graduation year	Education (field of study/degree)
Tallinn University of Technology	2003	Master's Degree, Economics
Stockholm University	1989	Economics/Degree of Bachelor of Social Sciences
Östra Real (Stockholm)	1981	Secondary School

4. Language competence/skills (fluent, average, basic skills)

Language	Level
English	Fluent
Swedish	Fluent
Finnish	Average
Estonian	Fluent

5. Special Courses

Period	Educational or other organisation
2000	Project management, Wenell Management AB, Stockholm

6. Professional Employment

Period	Organisation	Position
2007-…	WSP Sverige AB	Economist
2011- ...	Tallinn University of Technology, School of Economics and Business Administration, Department of Public Economy, Chair of Environmental Economics;	Lecturer (0.25)
2006-2007	WSP International	Senior Economist
2008 - 2009	Tallinn University of Technology, School of Economics and Business Administration, Centre for Economic Research;	Extraordinary Researcher
2003-2007	Tallinn University of Technology, School of Economics and Business Administration, Department of Public Economy, Chair of Public Economy;	Researcher
2001-2003	Tallinn University of Technology, School of Economics and Business Administration, Department of Public Economy, Chair of Public Economy;	Assistant
1997-2006	Inregia AB, Stockholm	Economist
1996	Swedish Ministry of Labour Market, Government Committee	Secretary Assistant, Unemployment benefit study
1991-1992	Tallinn State Pedagogical University	Lecturer Swedish
1991	Ministry of Foreign Affairs, Sweden	Assistant in Baltic studies
1990	Swedish Energy Administration	Consultant
1987-1990 1993-1995	Stockholm University, Department of Economics	Assistant lecturer Researcher

7. Scientific work

Work as scientific expert:
2008 - 2009 WSP Sweden, Suur Strait transport connection preparation of strategic development plan, key expert economics

2008 - 2008 Stockholm regional plan (RUFS 2010), expert in economics and co-editor
2007 - 2008 Regional Plan, Tripoli Region, Senior Economist
2006 - 2006 Economic development versus the Environment, Development of an assessment tool for identifying conflicts of goals in regional development policy planning, expert in economics

Publications:

Reimann, M.; Ehrlich, Ü.; Pädam, S. (2011). Non-use Value of the Natterjack Toad (Bufo calamita) in Estonia: a Contingent Valuation Study. *In: Recent Researches in Chemistry, Biology, Environment and Culture: Recent Researches in Chemistry, Biology, Environment and Culture, Montreux, Switzerland, December 29-31, 2011. (Toim.) Vincenzo Niola, Ka-Lok Ng.* WSEAS Press, 2011, 202 - 206.

Pädam, S.; Ehrlich, Ü. (2011). The foregone recreation value of Lake Ülemiste. Sulev Mäeltsemees, Janno Reiljan (Toim.). Eesti majanduspoliitilised väitlused (135 - 138). Berlin, Tallinn: Berliner Wissenchafts-Verlag, Mattimar

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Pädam, S.; Ehrlich, Ü. (2009). Life-Cycle Based Approach to Identifying Environmental and Economic Impacts of Biofuels: the Example of Stockholm. Jüri Sepp, Dean Frear, Wagiha Taylor (Toim.). The Key-Factors of Business and Socio-Economic Development (501 - 515). USA: COPE International

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Pädam, S. (2007). Transformation from a centrally planned to a market economy: developments in Estonia, 1989-2003. Istvan Kiglics (Toim.). Rebuilding the Market Economy in Central-East Europe and the Baltic Countries (255 - 317). Budapest: Hungarian Academy of Sciences

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Supervision:
Kristiina Kaasik, Master's-thesis (supervisor **Sirje Pädam)**, "Keskkonnajuhtimissüsteemi rakendamise uuring: Eesti ettevõtete näitel". Tallinna Tehnikaülikool. 2003.

Ave Matsalu, Master's-thesis, (supervisor **Sirje Pädam)**, "Ummikumaksu mõju Tallinna liiklusele., Tallinna Tehnikaülikool. 2011.

8. Defended theses

Master's Degree:
Sirje Pädam, Sotsiaalmajanduslik tasuvusanalüüs ja jätkusuutlik areng transpordi näitel [Social cost benefit analyses and sustainable development in transport], (supervisor Alari Purju) Tallinn University of Technology, 2003.

Bachelor of Science:
Sirje Pädam "Bättre miljö till lägre kostnad" Stockholm University. 1989.

9. Main areas of scientific work/Current research topics

- Environmental Economics, Transport Economics and Regional Economics

10. Other research projects

Biomethane as a Transport Fuel (2011-2012)
Cost-Benefit Analysis of Improvement of Public Transport Service in Lääne-Harjumaa and Läänemaa County (2011)
The Cost of Deviation of Regular Train Time Tables (2010-2011)
The Value of Improvements in Railway Tunnel Safety (2009-2010)

KOKKUVÕTE

Keskkonnapoliitika majanduslik aspekt: keskkonnareguleerimise kulud ja tulud Eestis

Üks majandusanalüüsi oluline valdkond on keskkonnaga seonduvad teemad, kuna turumajandus ei suuda tagada otsuseid, mis soodustaks jätkusuutlikku arengut. Suur osa keskkonnahüvistest on turuvälised ja selle tagajärjel ei anna turg tõest informatsiooni keskkonnahüviste tegelikust väärtusest. Lõppkokkuvõttes põhjustab see ebaefektiivse ressursikasutuse. Paradoksaalne on ka asjaolu, et isegi nõudluse olemasolu korral keskkonna kvaliteedi parandamiseks ei toodaks turud seda ikkagi piisavalt, sest keskkonna kvaliteedi parandamine on avalik kaup, mida üksikisikud saavad üldjuhul kasutada tasuta. Omaette probleem on, et keskkonna kvaliteedi halvenemine pole otseselt seotud keskkonnakasutusega, vaid ilmneb mingi muu inimtegevuse kõrvalmõjuna. Kuna praktikas turud, kas puuduva omandiõiguse või ebatäieliku informatsiooni tõttu, selliste kõrvalmõjudega ei arvesta, on nende vähendamiseks vaja ühiskonna sekkumist välismõjudega tegeleva poliitika näol. Keskkonnapoliitika kujundajate ülesanne on seetõttu keskkonna parandamise vajaduse määratlemine, kvaliteetsema keskkonna nõudluse väljaselgitamine, vastavate meetmete väljatöötamine ning vajadusel elluviimise rahastamise tagamine. Nende ülesannete täitmisel on majandusanalüüsil oluline koht.

Kuigi Eestis kehtestati keskkonnaseadusandlus ja keskkonnatasud varsti pärast iseseisvuse taastamist ning et Euroopa Liiduga liitumisel jõustusid täiendavad regulatsioonid, on keskkonnapoliitikas endiselt suuri väljakutseid, kaasa arvatud Eesti energiasektori uuendamine ja vajadus muuta see keskkonnale vähem koormavaks. Eraldi probleem siinjuures on majandusliku analüüsi piiratud kasutamine suunavate poliitiliste otsuste langetamisel. Tõhus poliitika peab arvestama nii sellega, kuidas üldsus keskkonna parandamist hindab kui ka arenguga pikemas perspektiivis.

Doktoritöö eesmärk on analüüsida Eesti keskkonnapoliitikat lähtuvalt heaoluökonoomika vaatepunktist. See eesmärk hõlmab näiteks EL-i keskkonnale eraldatud rahade jätkusuutlikkuse hindamist, taastuvallikatest toodetava elektri toetuse efektiivsuse analüüsi ja veekvaliteedi tagamise ettekäändel puhkepiirkonna sulgemisega tekkivate ühiskondlike kulude suuruse tuvastamist.

Väitekiri koosneb kuuest iseseisvast teadusartiklist, mida seob ühine teema: keskkonnapoliitika kulud ja tulud. Artiklid käsitlevad Eesti keskkonnapoliitikaga seotud juhtumeid ning keskenduvad suures plaanis kolmele aspektile. Esimene aspekt puudutab avaliku sektori eelarve keskkonnale eraldatud kulutusi. Teine aspekt käsitleb taastuvallikatest toodetava elektri toetuste kui energiasektori keskkonnamõjude vähendamise vahendi majanduslikku efektiivsust. Kolmas aspekt tegeleb väidetud eelistuste meetodi abil turuväliste keskkonnakaupade hindamisega.

Allolev tabel 8 annab ülevaate eelnimetatud aspektide ja väitekirja teadusartiklite seostest ning neid ühendava analüütilise käsitluse iseloomustavatest tunnusjoontest.

Tabel 8 Väitekirja aspektide ja teadusartiklite analüütiline käsitlus

Aspekt	Teadus-artikkel	Sotsiaal-majanduslik tasuvusanalüüs	Majanduslik hindamine	Mitte-majanduslik hindamine
Avaliku sektori keskkonna-kulutused	1	X	Tingliku väärtustamise meetod (CVM)	Keskkonna-indikaator EPI
	2			
Taastuv-elektri-toetused	3	X	Majandusliku kahju hinnang	
Turuväliste väärtuste hindamine	4			Hoiak
	5	X	Tingliku väärtustamise meetod (CVM)	
	6	X	Tingliku väärtustamise meetod (CVM)	

Töös tehtud analüüsid lähtuvad heaoluökonoomika meetodist. Väitekirja ülalnimetatud aspektide empiirilised allikad on erinevad. Esimest aspekti käsitlevad artiklid kasutavad peamise allikana EL-i ühtekuuluvusfondide kinnitatud rahastamisplaane ja valitsemissektori kulude andmeid. Teise aspekti artikli allikad hõlmavad lisaks elektritootmise üldandmetele ka kahe koostootmisjaama finantsaruandeid. Kolmandat aspekti kajastavad teadusartiklid lähtuvad iseseisvatest elanikkonna valimi ankeetuuringutest.

Lähtudes jätkusuutliku arengu majandusteooriast soovitab esimene artikkel keskkonnarahade jätkusuutlikkuse määramisel kahesammulist hindamist. Esimese sammuna selgitatakse välja, kas rahade paigutus on sotsiaalmajanduslikult tasuv. Teise sammuga püütakse välja selgitada, kas rahastamisvaldkonnal on ebakindlusest või pöördumatutest protsessidest tingitud täiendavaid ökoloogilisi kitsendusi. Avaliku sektori keskkonnarahastamise analüüs näitab, et Euroopa Liidu fondide keskkonnale eraldatavate vahendite struktuur ja maht Eestis, Lätis ning Leedus ei lähtu jätkusuutlikkuse põhimõttest. Investeeringud vastavad ainult osaliselt sotsiaalmajandusliku tasuvuse kriteeriumile. Hindamise teine samm ei tuvastanud lisakitsendusi. Selle asemel, et efektiivselt vähendada välismõjusid ja luua avalikke hüviseid, viitab järeldus tõsiasjale, et EL-i vahendeid on vee kvaliteedi parandamise ja jäätmekäitluse tõhustamise kaudu kasutatud uute liikmesriikide elustandardi parandamiseks.

Sellest järeldub, et kui investeeringute kavade ettevalmistamisel aastateks 2007−2013 oleks kasutatud sotsiaalmajanduslikku tasuvusanalüüsi, oleks investeeringute alternatiivne jaotus parandanud nende efektiivsust ja jätkusuutlikkust. Näiteks oleks rohkem panustatud looduskeskkonna liigirikkuse taastamisele.

Valitsuse keskkonnakaitselised kulutused Eestis suurenesid finantskriisi ajal (aastail 2008−2009). Siit järeldub, et EL eelarve koos oluliste keskkonnaalaste kulutuste juurdevoolu ajastusega kriisiaastatel tasakaalustas majanduse madalat konjunktuuri. EL eelarve keskkonnakulutuste jaotuste analüüs näitab, et keskkonnakulutuste eelkõige allokatiivse funktsiooni asemel on rahastamine olnud ümberjaotav ning taganud rahanduskriisi ajal teatud stabiilsuse.

Eestis, nagu teisteski Euroopa Liidu riikides, kehtestati kvantitatiivsete eesmärkide saavutamiseks taastuvelektritoetused. Subsiidiumide ja maksude ning muude hinnakujundusmeetmete suurim puudus on see, et nende mõju ulatus kvantiteedile ei ole ennetavalt selge. Kuna eesmärgile sobivat rahalist toetust ei ole võimalik täie kindlusega valida, siis vajab taastuvelektritoetuste regulatsioon pidevalt taashindamist. Toetuse ümbervaatamine toob investoritele paratamatult kaasa ebakindluse. Seega on toetuspõhise reguleerimise puhul vaja leida kompromiss ümbervaatamisest tulenevate probleemide ja kulukate toetuste jätkuva maksmise vahel.

Eestis on positiivne see, et taastuvelektri eesmärgid on käesolevaks ajaks teostatud ja ületatud. Teisest küljest on tulemused saavutatud taastuvenergia tootjate ülekompenseerimise hinnaga. Juhtumiuuringus vaadeldud koostootmisjaamad oleksid olnud kasumis ka ilma elektritoetuseta. Taastuvelektritoetuste ja Euroopa Liidu kasvuhoonegaaside lubatud heitkogustega kauplemise skeemi (EU-ETS) koosmõju tõttu on välismõjude vähenemine tunduvalt väiksem kui loodetud. Lisaks näitas uurimistöö, et Eesti taastuvelektritoetuse kulud on kasvanud tunduvalt kiiremini kui teistes EL-i riikides ning, et need kulud on tarbijad kollektiivselt kinni maksnud, samas kui kasusaajateks on teiste hulgas suured koostootmisjaamad. Eesti näide viitab vältimatule vajadusele vaadata läbi ka Euroopa Liidu taastuvenergiatoetuste skeemid, et need oleksid ühilduvad EL-i kasvuhoonegaaside lubatud heitkogustega kauplemise skeemiga (EU-ETS).

Keskkonnaparandamise turuväliste väärtuste hindamise ja rahvusvahelise võrdlemise uuring näitas, et keskkonnaparandamise osas valitseb Eestis üldine keskkonnakaitset pooldav hoiak. Seda näitab eelkõige tõsiasi, et Eesti paigutus valmisolekus ohverdada rahalisi vahendeid keskkonna kaitsmiseks uuritud riikide pingerea keskele. Hoolimata sellest asetseb Eesti riikide pingerea alumises otsas, kui küsimuseks on elatustaseme alandamine looduskeskkonna kaitsmise nimel. Riikidevaheline statistiline baasanalüüs näitas, et hoiakud sõltuvad sissetulekust. Mida kõrgem on riigis keskmine sissetulek, seda suurem on positiivse suhtumise osakaal keskkonna kaitsmisel ja ankeedile vastanud olid valmis maksma kõrgemat hinda ning nõustusid ka elatustaseme alandamisega. Kui vastanutelt küsiti nende valmiduse kohta maksta keskkonna kaitsmise eesmärgil kõrgemaid makse, siis see valmidus ei sõltunud riigi keskmisest sissetuleku tasemest.

Loomaaialoomade heaolu väärtustamise uuring kajastas nõudlust nii kasutus- kui mittekasutusväärtuste osas. Väljaselgitatud kogunõudlus ei olnud piisav, et finantseerida Tallinna loomaaia arengukavas ette nähtud kulusid loomade elutingimuste parandamiseks. Kasutades fiskaal-föderalismi teooriat, oleks soovitav, et loomaaeda rahastatakse keskvalitsuse vahenditest. See muudaks ka uuringu järeldust. Olemasolev loomaaia rahastamise skeem eirab tõsiasja, et olukorra parandamisest kasusaajad on ka Tallinnast väljaspool elavad inimesed, samas kui loomaaeda rahastavad peamiselt Tallinnas elavad maksumaksjad.

Joogivee kvaliteedi tagamiseks Ülemiste järve sulgemisest tingitud heaolu kaotuse analüüs näitas, et kaotuse suurus Tallinna elanikele on ligikaudu 1,8 miljonit eurot aastas. Kuigi joogivee kvaliteedi pärast ei ole vajalik inimeste eemalehoidmine joogiveeks kasutatavast veekogust, ei ole Eestis väljakujunenud keskkonnapoliitika tõttu veekogust joogivee tootmine ja selle veekogu puhkealana kasutamine ühitatavad. Selline praktika erineb oluliselt Põhjamaade veekaitsepoliitikast. Arusaamade muutmine võimaldaks avalikku pääsu väärtuslikule puhkealale linnakeskuse vahetus läheduses.

Väitekirja aspektide kõiki juhtumiuuringuid hõlmav kokkuvõttev järeldus on, et analüüsitud keskkonnapoliitika valdkonnad on nendest saadava kasuga võrreldes sageli liiga kulukad, samal ajal kui poliitikast väljajäävad valdkonnad vääriksid suuremat tähelepanu. Nõudlusest ja keskkonnaprobleemidest lähtudes investeeritakse liiga palju EL-i rahasid jäätmekäitlusesse ja veevarustuse infrastruktuuri. Samuti toetatakse tarbijate kulul taastuvallikatest toodetavat kallist elektrit palju suuremas mahus, kui eesmärk seda ette näeb. Kui arvestada kogu joogiveega varustamise kulu, tuleb tallinlastele joogivesi kätte, lisaks finantskuludele, linnasisesest puhkealast loobumise hinnaga. Juhtumuuringud viitavad sellele, et keskkonnapoliitika tähelepanu vääriks rohkem pöörata liigirikkuse taastamisele, loomaaia loomade elutingimuste parandamisele ja linlastele puhkealade loomisele.

ABSTRACT

The thesis is made up of six independent research papers connected by a common theme, which is the costs and benefits of environmental policies. The research papers represent different case studies mainly based on Estonia and on a general level they focus on three aspects of environmental costs and benefits. The first aspect relates to the allocation of environmental expenditure in the public budget. The second aspect is about the economic efficiency of using feed-in tariffs (FIT) as a means to reduce the environmental impact of the energy sector. The third aspect is devoted to the valuation of non-market goods with stated preference methods.

Welfare economics and cost-benefit analysis provide the basis of the analyses of the thesis. At the methodological level the empirical sources differ between the aspects of the theme of the thesis. The first aspect uses as its main source the ratified funding plans of EU cohesion funds and data about government expenditure. Energy sector statistics and the financial reports of two combined heat and power (CHP) plants are inputs to the second aspect. Surveys were carried out for empirical data collection of the cases studies included in the third aspect of the thesis.

By using economic theory of sustainable development, the assessment of the sustainability of EU funding to the environment applies a two-step approach. The first step is based on the cost-benefit criterion and the second step uses ecological information as an additional constraint. The enquiry into the allocation of environmental expenditure in the public budget showed that Estonia's, Latvia's and Lithuania's EU-fund allocation plans to the environment did not consider sustainability as priority. Only to some degree did the investments pass the cost-benefit rule. Available evidence suggests too much funds is devoted to investment in waste management and drinking water infrastructure and too little to the enhancement of biodiversity.

In Estonia central government spending on environmental protection increased during the financial crisis in 2008–2009. It thus seems as the timing of the EU budget with significant inflow of environmental spending during the crisis years has potentially been counter-cyclical. The observations about the environmental fund allocation from the EU budget thus suggest that instead of being primarily allocative, which is the expected function of environmental expenditure, the funding has been redistributive and supposedly provided stabilisation during the financial crisis.

In Estonia feed-in tariffs (FIT) were introduced in order to achieve quantity goals. On the positive side, the goals have so far been outperformed. On the other hand, this achievement has come at a high cost as the feed-in tariffs overcompensate producers of renewable electricity. By the co-existence of FITs and the EU-ETS the reduction in externalities will be much smaller than expected. Furthermore the assessment found that the costs of the Estonian FITs have increased at a more rapid rate than in other countries. The results of the

research suggest FITs are non-efficient and have adverse impacts on income distribution.

The research into non-market valuation of environmental improvement showed, according to expectations that in cross-country comparison attitudes to contribute to environmental protection increase with the level of income. This is the case when respondents consider paying higher prices and accepting cuts in the living standard. When respondents are asked about their willingness to pay much higher taxes for environmental protection there is no correlation to the country level of income. Instead a country's tax burden seems to explain the attitude towards paying higher taxes for the sake of environmental protection.

The survey of human valuation of zoo animal wellbeing showed that people express both use and non-use value for improvement of zoo animal living conditions at Tallinn Zoological Gardens. However, the total benefit from the foreseen improvements according to the zoo development plan was not sufficient to cover the costs. Using the theory of fiscal federalism suggests that the responsibility of zoo funding should be centralised. This is because the existing funding system disregards the fact that there are beneficiaries of the improvements in zoo animal wellbeing outside the territory of the City of Tallinn, while the tax payers who cover the public funding of the zoo are Tallinn residents.

The analysis of the welfare loss of the lake closure to assure drinking water quality showed that the loss to Tallinn inhabitants is about Euro 1.8 million annually. According to the city's practices there is no trade-off between drinking water provision and recreation use. This is in contrast to water protection policies elsewhere. A change in views would make possible public access to a valuable recreation area within the immediate neighbourhood of the city centre.

The general conclusion of the case studies reported in the research papers is that those areas covered by environmental policies often are too costly in relation to their benefits, while neglected areas deserve greater attention. The analyses further suggest that application of cost-benefit analysis has potential to increase the precision of current policies and thereby improve efficiency of environmental policy making.

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