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HELEN POLTIMÄE

The distributional and behavioural effects of Estonian environmental taxes

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The distributional and behavioural effects of Estonian environmental taxes

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4. **Paas, T. Poltimäe, H.** (2009) Analysis of national innovation performance: how innovative are the Baltic states in the international context. Taylor, W., Sepp, J., Frear, D. (Eds). The Key Factors of Business and Socio-Economic Development During the Global Crisis. USA, Wilkes-Barre: Congress of Political Economists International, pp. 255–275

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2. **Paas, T.; Poltimäe, H.** (2010) A comparative analysis of the national innovation performance: the Baltic states in the EU context. 11th Bi-Annual Conference of European Association for Comparative Economic Studies "Comparing Responses to Global Instability", Tartu, Estonia; August 26–28
3. **Võrk, A.; Poltimäe, H.** (2010) How should social protection system respond to the ecological tax reform in Estonia? The Conference of Estonian Economics Association, Narva, Estonia, January 28–29
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ABBREVIATIONS

2 SLS	two-stage least squares
AIDS	almost ideal demand system
CO_2	carbon dioxide
COICOP	classification of individual consumption according to purpose
CPA	statistical classification of products by activity
E3ME	Energy–Environment–Economy Model of Europe
EEA	European Environmental Agency
ETR	ecological tax reform
HBS	Household Budget Survey
IO	input–output table
NACE	statistical classification of economic activities
OLS	ordinary least squares
SUR	seemingly unrelated regression
VAT	value added tax

INTRODUCTION

Motivation for the research

Environmental taxes have been in place for decades, but their economic and social implications are still in dispute. Environmental taxes belong to the group of economic instruments used to tackle environmental problems; in addition, regulatory and voluntary instruments are also applied. This thesis focuses only on environmental taxes. The aim of environmental taxes is to change the behaviour of people or enterprises with price signals; hence there should be a decrease in the consumption or production of a taxed (polluting) good and the specific environmental problem should also decrease. At the same time, taxes change the income distribution in a society and can potentially have a positive or negative effect on income inequality. As for environmental taxes, their negative effect on poorer populations has been widely used as an argument against their implementation. However, the linkage between income distribution and people's response to environmental tax can also occur in a contrary direction: consumption decrease might depend on distributional issues, as poorer populations not having enough resources, for example, to invest in new cars or house insulation. Thus, the effectiveness of environmental taxes is determined by behavioural effects and equity issues according to distributional effects. The linkages between effectiveness and equity issues of environmental taxes are shown in Figure I.

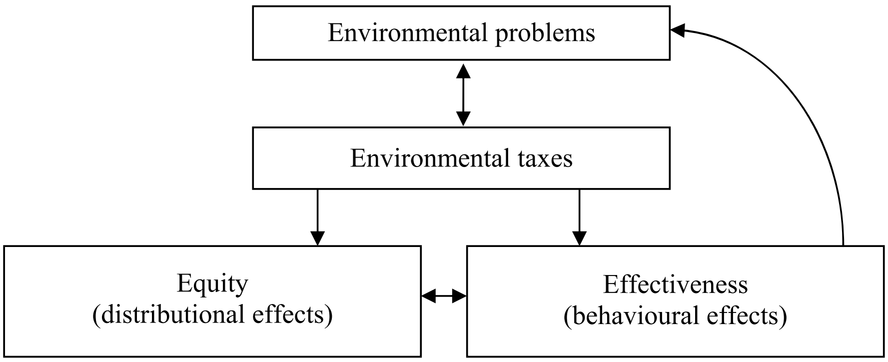

Figure I. The linkages between environmental problems, environmental taxes and their effects on households (author's figure)

Usually distributional issues are regarded as a social issue and not popular in economic research, but increasingly the direct linkage between income inequality and economic effects is recognised. For example, Levine et al. (2010) show that as a result of growing income inequality, saving rates are decreasing, and this has a clear relationship to other economic variables. Hence, the distributional issues are important to consider.

The focus of this thesis is Estonia, as a small country that has seen rapid economic and socio-political changes during recent decades. Although covering only one country, the research implications are wider and useful for other countries as well. First of all, Estonia is a good example of a country that had to introduce stringent environmental requirements for integration into the European Union, therefore the level of environmental taxes has risen very quickly. Also, Estonia has witnessed rapid economic growth during last two decades, which means that consumption patterns have changed tremendously. This means that income inequality has changed quite considerably. The Gini coefficient in Estonia has increased from 0.277 at the beginning of the 1990s to 0.396 in 1995, stayed at 0.36 for ten years and decreased in the 2000s, but increased again during the economic crisis years (Roosalu 2013). Income inequality in Estonia is one of the highest in the European Union, along with Bulgaria, Romania, Spain, Italy, Greece, the United Kingdom, Latvia and Lithuania (Eurostat 2013b).

Analysis of the effects on households in conditions of fast-growing consumption and rising environmental taxes provides grounds for elaborating policy proposals that can mitigate drawbacks for households in general or for specifically vulnerable groups. Such a development path is relevant to other emerging countries, especially when they adopt the legislation and tax policies of the European Union.

The level of environmental taxes in Estonia is not very high if compared to the main consumption tax, VAT, but during the past decade it has risen substantially: while in 2001 environmental tax revenues formed 6% of total tax revenues, this ratio increased to 9% in 2011. There is no solid trend in environmental tax revenues in Europe, but in general, environmental tax revenues are decreasing in old member states and also slightly in the new member states of Central and Southern Europe (see Figure II). However, in the Baltic countries and Poland, the environmental tax share of GDP has been increasing and for Estonia, the increase has been larger than average for this country group. Hence, similar trends of environmental tax revenues in the Baltic countries and Poland can be observed, although the tax types and levels are different.

While a large proportion of the environmental taxes are paid by enterprises, the tax load on households has also increased, as most fuel excise rates have more than doubled in the past decades and some new excise taxes have been imposed since 2008 in Estonia. There are several policy documents that stress the importance of a more resource- and energy-efficient economy, for example Europe 2020, which emphasises sustainable growth as one of the three main priorities (European Commission 2010). At the same time, the possibility of ecological tax reform (ETR) is promoted in the literature and also in real life. The aim is to increase taxes on environmental 'bads' and decrease the tax load on employment. Hence, taking into account the potential of placing more emphasis on environmental taxation, the topic of the potential effects of these taxes deserves more attention.

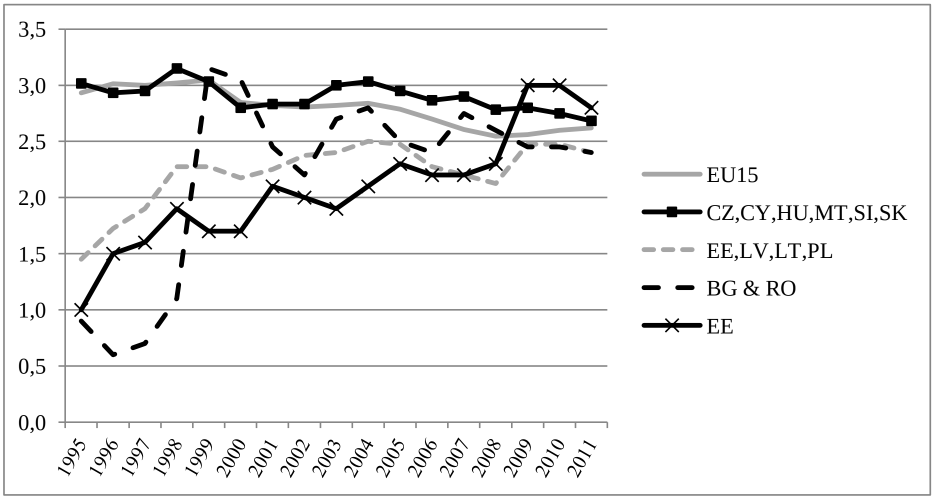

Figure II. Shares of environmental taxes in GDP in European countries, 2000–2011 (author's figure based on Eurostat 2013a)

So far in the literature, research on environmental taxes has mainly been done for developed countries, where the income levels and tax systems do not witness such rapid changes but are more or less stable, for example Denmark (Jacobsen et al. 2003, Wier et al. 2005), Germany (Bork 2006), the United Kingdom (Dresner & Ekins 2006), Italy (Tiezzi 2005), Spain (Labandeira & Labeaga 1999), Ireland (Callan et al. 2009) and the Netherlands (Kerkhof et al. 2008). Also, most research in this area has been ex ante analysis of hypothetical taxes. There are only a few ex post analyses of existing environmental taxes. An environmental tax that has deserved increasingly more attention in recent years is petrol tax and a special book has been dedicated to the distributional effects of petrol tax (Sterner 2012). However, the countries covered are developed countries (the USA and European countries like France, Germany, Spain, Sweden and the United Kingdom) and developing countries (for example, India, Indonesia, Ethiopia, Kenya, Costa Rica and Mexico).[1]

In terms of the research topic, it often seems to be a trade-off: if research is concentrated on distributional issues, the possibilities of considering behavioural issues are limited (for example, using only some household types). On the other hand, if concentrating on behavioural issues, this is a broad topic in itself and the methods applied for analysis vary depending on data availability, interests and the backgrounds of the researchers and the research emphasis. For example, the price elasticity of petrol has been estimated to vary from –0.03 for the USA (Nicol 2003) to –1.28 for Italy (Tiezzi 2005); estimates also vary for

[1] The only example from Central and Eastern European countries is the Czech Republic, but as seen from Figure II, the tendencies among Central Eastern European countries and Baltic countries could be different.

the same country: for example, for Spain from –0.11 (Labandeira et al. 2005) to –0.82 (Barros & Pietro-Rodriguez 2008) (an overview of the studies is presented in chapter 2). Hence, the estimates are not solid and it is not clear how environmental taxes affect income distribution in the longer term and how is this related to behaviour.

The novelty of the thesis comes from the two named aspects: first of all, it is an ex post analysis of environmental taxes in the context of an economy that has experienced rapid changes. Environmental tax effects on households have not received attention in such economies. Secondly, it acknowledges the linkages between distributional and behavioural effects, as these might reinforce each other, and it attempts to relate these effects to each other. Hence, the scope of these taxes' effects can also be called an original contribution of the thesis, as it covers both direct and indirect distributional issues and behavioural issues, allowing for the heterogeneity of households and relying on micro-level data.

The aim, research questions and research tasks

The aim of the thesis is to find out the distributional effects of environmental taxes and the possible linkages to their effectiveness in terms of behaviour change in the example of Estonia.

The main research questions, which also form the three parts of the thesis, are:

- What are the direct distributional effects of environmental taxes in Estonia?
- What are the indirect distributional effects resulting from price changes induced by environmental taxes in Estonia?
- Which has been households' consumption response to environmental taxes and how is this related to their socio-demographic characteristics?

Hence, the thesis concentrates on the two types of effects of environmental taxes on households. Firstly, distributional analysis concerns how environmental taxes affect income inequality in a society, and this effect can in turn be divided in two: direct effects resulting from consumption of a taxed good, and indirect effects resulting from the imposed taxes' effect on prices. Secondly, as environmental taxes are implemented to address some environmental problem, it is also important to evaluate whether these taxes have led to lower consumption of a taxed good; i.e. the behavioural effects of environmental taxes.

While the distributional effects of environmental taxes have received some research attention worldwide (a good overview is provided by Ekins et al. 2011, and for petrol taxes Sterner 2012), and there is quite good understanding of how to measure these, the behavioural effects are much more difficult to measure. The best option would be to use observation data, but unfortunately this is not available for Estonia. Hence, the thesis estimates the price and income elasticity of taxed goods to analyse households' changes in consumption, but also differentiates the effects across household types, depending on socio-demographic characteristics.

The research tasks of the thesis consist of the following:

- To provide a theoretical basis for environmental taxes, including the theoretical arguments for their application, definition, classification and development.
- To discover the linkages between environmental taxes and distributional issues based on economic theory.
- To present the empirical research results of the distributional and behavioural effects of environmental taxes from other countries.
- To set up a methodological framework for analysing the distributional and behavioural effects.
- To explore how the Estonian environmental tax system is similar to or different from those of other European Union member states.
- To find out, which distributional and behavioural effects the Estonian environmental taxes have on households.

Research object, data and methodology

The form and taxation base of environmental taxes have been altering over the past decades. Traditionally, environmental taxes were only those applied to specific pollutants or resources and thus fulfilling only environmental objectives. Environmental taxes today are mingled with other objectives, for example fiscal ones, as some environmental taxes are good sources for state budget revenues. Therefore environmental taxes today do not comprise only taxes on pollutants, but also proxies for these. A widespread definition is that an environmental tax is a tax whose tax base is a physical unit (or a proxy for it) of something that has a proven, specific negative impact on the environment (Eurostat 2001). Thus, in addition to traditional resource and pollution charges, taxes on energy, for example fuel excise, are considered environmental taxes, as they are related to CO_2 emissions. The research object of the thesis is Estonian environmentally related taxes and charges (the term “environmental taxes” is used in the thesis to refer to both categories). Of these, the thesis focuses on electricity excise, fuel excise and environmental charges (see Figure III).

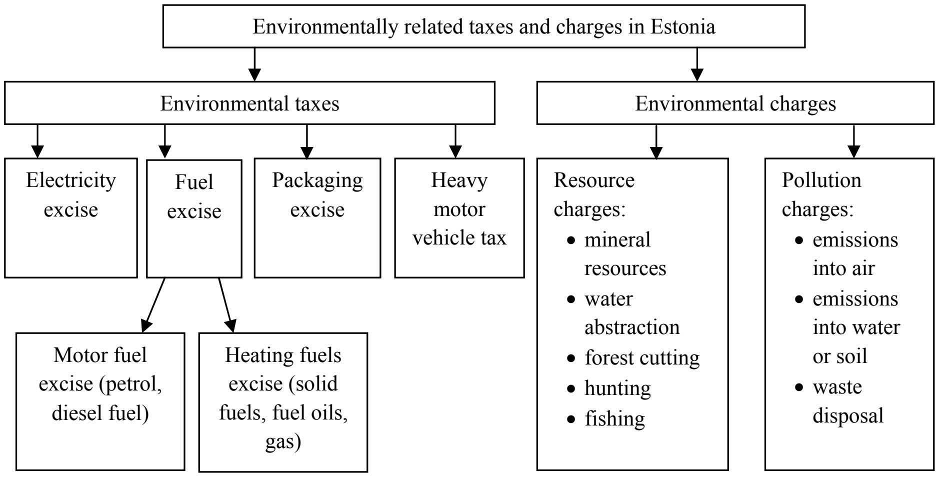

Figure III. Environmentally related taxes and charges in Estonia (author's figure)

Environmental taxes are often regarded as regressive taxes that have negative implications for income distribution. The research on distributional effects is static in that it reflects a situation only at a certain point in time. To analyse the distributional effects, the terms 'regressivity', 'progressivity' and 'neutrality' of a tax are used. Regressivity means that lower income groups pay higher tax shares than high income groups, leading to increasing inequality in a country. In addition, different inequality indices are used in the thesis (the Kakwani index, Reynolds-Smolensky index and Atkinson index). The distributional effects consist of direct and indirect environmental tax loads. The essence of direct and indirect distributional effects is elaborated on in chapter 3. As environmental taxes are aimed at changing behaviour, it is important to analyse change over time, the dynamic effects. To analyse this change over time, the third part of the thesis concentrates on changes in motor fuel consumption, as its price and tax levels have changed considerably during the observed period, while for example the electricity excise was implemented only in 2008. Microeconometric methods are implemented in this thesis to take into account the specific features of the data and problem statements.

Although the effects of environmental taxes on enterprises are equally important, households are consumers and they drive demand. This thesis analyses the effects on households.

The data used is mostly from the Household Budget Survey (HBS) of Estonia to calculate the direct environmental tax load and the consumption changes. The HBS was conducted in 2000–2007 and again since 2010, but unfortunately the crisis years 2008 and 2009 were not covered, as the methodology of the survey was altered during that period. In order to find the indirect distributional effect, different statistical data sources are used in addition to the HBS: fuel and electricity use of economic sectors, environmental charges paid by enterprises and the input–output table of the economy.

Limitations of the thesis

Although the research was triggered by sustainability concerns, i.e. the perception that economic, environmental and social objectives need to be aligned with each other, the thesis does not deal with evaluating whether the current environmental tax system is suitable for obtaining sustainability in Estonian society. Furthermore, the research deals only with the effects on households. Although there are also significant effects on enterprises, these are considered in this thesis only as far as they change the prices of consumer products. Hence, the behavioural effects of enterprises (for example, investments into cleaner technology or switching to different fuels or moving production to another country) are omitted. There are several reasons for this. First of all, the excise taxes that form the largest part of the Estonian environmental tax revenues are intended to give price signals to consumers, who should change their behaviour accordingly. A majority of taxed fuels are consumed by the household sector. A more pragmatic reason is that there is no good data available on company levels or, if it is available, it is at a highly aggregated level.

This thesis concentrates only on so-called micro-level analysis, i.e. households. It does not reflect state-level macroeconomic effects, for example, effects on state budgets, the administration burden, environmental effectiveness, efficiency of the use of earmarked revenues, etc. As the focus of the thesis is income inequality, it is important to allow as much heterogeneity as possible and hence, micro-data and microeconometric analysis have been applied here.

Structure of the thesis

The first chapter provides a theoretical framework for environmental taxes, which come from the existence of externalities. The chapter also gives a definition of environmental taxes and the different classification principles and presents their development during past decades. The second half of the chapter provides an overview of how environmental taxes are linked to distributional issues and what concepts and measures could be used to measure these effects.

The second chapter gives an overview of different studies on the distributional and behavioural effects of environmental taxes in the scientific literature; this includes analysis of both hypothetical taxes and actual taxes, but most of the research is focused on the hypothetical carbon tax to tackle climate change.

The third chapter presents the research questions, propositions of the thesis, methodological framework, data and the specific methods used.

In the fourth chapter, the Estonian environmental tax system is placed in the context of the EU to discover similarities and differences. After that, empirical findings are presented regarding the direct and indirect distributional effect on households and also the behavioural effects, which are then linked to distributional issues.

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All and any mistakes or errors in this dissertation are the responsibility of the author.

I. THEORETICAL BASIS OF THE THESIS

1.1. The theoretical background of environmental taxation

The main reasoning for environmental taxes comes from market failure, especially externalities. Externalities consist of the costs or benefits felt beyond or 'external to' those causing the effects. As Baumol and Oates (1995) state: "An externality is present whenever some individual's (say A's) *utility* or *production* relationships include real (that is, nonmonetary) variables, whose values are chosen by others (persons, corporations, governments) without particular attention to the effects on A's welfare". Additionally, the one whose activity affects others' utility levels does not pay compensation for this activity equalling the costs imposed on others in the case of a negative externality and does not receive any payment equal to the benefits in the case of a positive externality.

A negative externality like air pollution or noise creates a marginal external cost that causes the social marginal cost and the private marginal cost to differ. When considering only private costs, the price is lower than when considering externalities. Additionally, the quantity produced is larger in the case of market equilibrium without considering externalities. In the presence of an externality, the system will produce an allocation which is higher than allocative efficiency requires. Different alternatives are available to make those causing the externality bear the social marginal cost of their behaviour, for example, property rights, Pigou taxes and private negotiations. This thesis concentrates on environmental taxes, whose original idea has to a large extent been based on the so-called Pigou tax.

The Pigou tax is named after economist Arthur C. Pigou, who was the first to suggest taxes to bring private and social costs into line. As Pigou (1920) put it: "It is plain that divergences between private and social net product of the kinds we have so far been considering cannot, [...], be mitigated by a modification of the contractual relation between any two contracting parties, because the divergence arises out of a service or disservice rendered to persons other than the contracting parties. It is, however, possible for the State, if it so chooses, to remove the divergence in any field by "extraordinary encouragements" or "extraordinary restraints" upon investments in that field. The most obvious forms which these encouragements and restraints may assume are, of course, those of bounties and taxes." If polluters paid taxes equal to the marginal external cost of resulting pollution, they would feel or 'internalise' these costs. This principle goes along with the 'polluter pays principle', which has a long history in environmental policy.

In the 1970s and 1980s, environmental policy was to a great extent conducted by regulatory instruments, also known as command-and-control instruments (Böcher 2012, Common & Stagl 2005). However, during the 1980s it was recognised that traditional regulatory environmental policy was not able to fulfil the objectives of preventing environmental damage (Ekins 1999). The idea of the free market as a solution to different problems was also gaining ground: if individuals respond in a rational way to sets of benefits and burdens in order to maximise their welfare, then an incentives-based approach should lead to the desired targets (Fitzpatrick 2011b). Economic instruments are acknowledged for static and dynamic efficiency, meaning that investments are made by those for whom it is cheaper and the incentive motivates enterprises constantly, not just up to certain limits the way that regulatory measures do.

Historically, the first environmental taxes applied were cost-covering charges, typically wastewater or waste-disposal charges. In the 1980s, taxes that were not directly linked to some services were applied, but their revenues were earmarked for environmental projects (EEA 2000). In the 1990s, fiscal environmental taxes were applied, whose revenues went to general state budgets. Also in the 1990s, the concept of environmental tax reform (ETR) started to spread. The development of the concept has been attributed to Tullock, who wrote already in 1967 that "[...] there must be at least some taxes which, far from imposing an excess burden, produce an excess benefit. If some activity imposes an external cost, then a properly calculated tax on it will reduce the total output of the private sector by less than the revenue received by the government." He also suggested that taxes on water resources and air pollution would provide considerable revenues, while also enhancing welfare (Tullock 1967). The idea of ETR is to shift taxation from factors of production, such as labour and capital, to pollution and the use of natural resources. Terms like 'ecological tax reform', 'green tax reform', 'environmental fiscal reform', 'green tax swap' and 'green tax shifting' have been used for the same concept (Bosquet 2000).

The appropriate level of environmental taxes has been in dispute for a long time. Baumol and Oates (1995) argue that environmental tax levels should be set so that environmental objectives are obtained. They also stress that it should be the least-cost method for obtaining these objectives. Still, several problems occur. First, the information is not available to set appropriate objectives, as some negative implications are only revealed over long time periods and the processes are so interrelated that one cannot accurately predict what the appropriate environmental objective is. Secondly, the tax level actually applied is usually the result of both scientific research and political negotiations. Specifically, those authors who represent the discipline of ecological economics (for example, Common & Stagl 2005), argue that the arbitrary standards often adopted do not necessarily guarantee sustainability and it is uncertain whether a tax imposed will achieve the standard aimed at. Hence, the environmental taxes that have been imposed lead the world towards sustainability, but they do not express the 'right prices'.

As discussed above and elaborated on in the next chapter, contemporary environmental taxation is not just about externalities, but also raising funds for state budgets. As discussed by Fullerton and Wolverton (2005), the tax can be separated into an externality-correcting component and a revenue-raising component. For example, Lin and Prince (2009) calculate that the optimal gasoline (petrol) tax in California should be more than three times higher than it currently is, derived mostly from the Pigouvian tax part (marginal environmental damage), but the Ramsey tax part is also quite high, reflecting that petrol consumption is quite inelastic. The term 'Ramsey tax' is used to denote the taxation principle of Ramsey to tax goods for which demand is the most inelastic (1927). However, as for motor taxes, Ramsey shows that in such cases, firstly the part equal to the damage to the roads must be separated. In the contemporary state of knowledge, where the negative environmental and health effects of transport have been acknowledged, this would mean that this damage cost should be separated, but this would require calculating the external cost of transportation, which is way out of the scope of this thesis. Hence, the optimal level of environmental taxes is not discussed in this thesis, but rather the possible effects of current taxes, which also gives background information for future changes in the level of environmental taxes.

Bovenberg and Mooij (1994) show that environmental tax levels depend on existing distortionary taxes (specifically labour taxes). They show that the welfare effects of a revenue-neutral ETR can be expressed as the following (Bovenberg & Mooij 1994):

$$(1) \quad \frac{dU}{\lambda} = ht_L dL + \left[t_D - N\frac{\partial U}{\partial E}\left(-\frac{dE}{d(ND)} \right) / \lambda \right] dD$$

where:
λ – marginal utility of income;
h – labour productivity;
t_D – pollution tax on dirty consumption;
D – dirty consumption;
t_L – tax rate on labour income;
E – environmental quality;
N – number of households.

In the 'first-best' case, where there is no distortionary labour tax ($t_L=0$), the optimal value of a pollution tax would be the Pigou tax, at which level the benefits from environmental improvement would exactly offset the adverse welfare effects due to erosion of the tax base:

$$(2) \quad t_D = N\frac{\partial U}{\partial E}\left(-\frac{dE}{d(ND)} \right) / \lambda$$

In the presence of a distortionary tax on labour ($t_L>0$), the optimal environmental tax depends on the response of employment to a change in the tax mix. The argumentation behind this is based on the fact that a lower tax rate on labour income does not fully compensate workers for the adverse effect of the pollution levy on their real after-tax wage, which is due to the erosion of the base of the environmental tax, which in turn comes from the behavioural effect. To obtain the strong form of a double dividend (i.e. both economic and environmental effects), the initial tax system has to be greatly sub-optimal. Different authors have shown that labour taxes are indeed more distortionary in Europe compared to the USA, Canada and Japan (an overview is presented by Chiroleu-Assouline & Fodha 2010). As this thesis is about environmental taxes and their linkages to distributional issues, and not about the optimal tax level, it focuses only on environmental taxes, not the issue of labour taxes.

An overview of the suggested environmental tax levels proposed by different authors is presented in Table 1.1.1.

Table 1.1.1. The environmental tax levels suggested by different authors

Author	**Suggested tax level**
Pigou (1920)	The marginal cost of environmental damage
Ramsey (1927)	Taxation should be based on goods for which demand is inelastic.
Baumol & Oates (1995)	The pricing procedure will not lead to a Pareto optimum, but the use of unit taxes to achieve specified quality standards is the least-cost method for the achievement of these targets.
Bovenberg & Mooij (1994)	In the case of a pre-existing distorting tax on labour, the optimal environmental tax depends on the response of employment to a change in the tax mix.
Common & Stagl (2005)	Acknowledge the complexity: pollution standards are arbitrary and environmental taxes move things in the direction of sustainability, but are not expressing the 'right prices'.

Source: compiled by the author based on Pigou (1920), Ramsey (1927), Baumol & Oates (1995), Bovenberg & Mooij (1994) and Common & Stagl (2005)

Environmental taxes have been criticised for different reasons. The effect on income distribution has been a widespread argument against their implementation, and as this is an important topic for this thesis, it will be elaborated on in the next chapters. Another critique is based on moral arguments: "economic incentives may not appeal to, and may sometimes conflict with, the non-monetary values on which ecological justice could ultimately depend" (Dobson 2009, referred to via Fitzpatrick 2011b). This means that environmental taxes may be viewed as licensing selfish behaviour by "buying the right to pollute" (Fitzpatrick 2011b).

However, as Ekins (1999) puts it: environmental taxes have a high potential to integrate economic and environmental policy, as these are targeted to internalise environmental costs into prices, and in providing cost-effectiveness for environmental policy. The author of this thesis also takes the approach that environmental taxes are necessary instruments for achieving sustainability, although they are not sufficient and must be part of a wider policy mix. There is still good reason to believe that people react to price changes brought about by taxation.

Regarding the actual implications of ETRs (increasing taxes on ‘bads’ like environmental pollution and decreasing taxes on ‘goods’ like employment), the meta-analyses that have been conducted imply that the effects on environmental quality are substantial (Bosquet 2000, Patuelli et al. 2005). For economic effects, effects on employment and on GDP have been analysed. In terms of employment, both meta-analyses show that there is a positive effect on employment, but the effect is considerably smaller than the environmental effect. In terms of GDP change, the two meta-analyses show more mixed results.

There are several issues that arise along with environmental taxation. First of all, the level of application: while environmental problems often have global implications or at least wider implications than just a nation, the environmental taxes are applied at state level. This leads to problems of coordination, fairness, monitoring and compliance (Fitzpatrick 2011b). Another issue is public mistrust of government motivation in applying or increasing environmental taxes: they view taxes as means of raising revenue, not incentives in essence. People seem not to understand how taxes can increase welfare and influence behaviour (Kallbekken & Sælen 2011). Furthermore, in order to obtain the set objective, the different taxes, regulations and subsidies that are valid in the country must be in line with each other. If they are not and conflicting incentives exist, the incentive taxes do not succeed in gaining the set objective (Bailey 2002). Another interesting issue is the relationship between environmental policy (including environmental taxes) and development. It is argued specifically by poorer countries that environmental issues are something that a rich country can deal with, while for poorer countries the first objective is to attain some development level, and after that deal with environmental preservation. This is related also to the Environmental Kuznets Curve, according to which, in the early stages of economic growth, degradation and pollution increase, but beyond some level of income per capita, the trend reverses (Stern 2004). This hypothesis has been criticised for various reasons and it has been stated that perhaps the relationship is more complex: for example, the shape of the curve may differ for different pollutants. However, the general shape is believed to be an inverted U-shape, supporting the understanding that richer countries have more resources to preserve the environment.

Environmental taxation is also an issue at the individual level, as for example according to Gowdy (2005), when individuals are more secure financially, they are more likely to care about the welfare of future generations and the state of the environment. This issue is elaborated on in the next chapter, as it is related to one of the research questions of the thesis: the response to environmental taxes differs according to socio-economic factors, including income.

Before proceeding, a classification of environmental taxes needs to be presented, as this can be done in several ways. The European Environmental Agency (EEA 2000) defines the following bases:

- main objective
- field of operation
- point of application
- tax base.

Classification according to main objective

According to the main objective, environmental taxes can be classified into cost-covering charges, incentive charges and fiscal environmental charges. The shortcoming of cost-covering charges is that normally only part of the total cost is covered by polluters and the true 'environmental' cost is not covered (EEA 2000).

If an environmental charge or tax is implemented purely in order to change some environmentally damaging behaviour, not to raise revenues, then it is called an incentive charge. Environmental taxes whose objective is to raise revenue for government budgets are called fiscal environmental taxes.

It must be mentioned that these three types of environmental taxes cannot always be clearly distinguished. For example, a cost-covering charge may also be designed to change behaviour, which can also be the case for a fiscal environmental tax. However, an incentive tax which is functioning effectively cannot provide much revenue for the state budget.

Classification according to field of operation

Classifying environmental taxes according to their main field of operation involves energy taxes, transport taxes, pollution taxes and taxes on natural resources (EEA 2000). This type of classification is especially convenient for statistical purposes and Eurostat also uses such an approach. Although it is arguable whether energy and transport taxes are environmental taxes, these are included because they have an environmental impact, whether intended or not.

Classification according to point of application

Environmental taxes can also be applied to pollution, to products, to capital goods or to activities (EEA 2000). According to the theory, a tax should be imposed directly on the cause, the environmentally damaging object. However, this is not always possible if there are several different pollutants, if the pollutant affects several different environmental domains and if emissions are generated by many small, mobile sources (OECD 2001). Hence, a tax is imposed on a product or activity which is a proxy for the pollution itself. For example, fuel usage (differentiated by fuel) is considered a proxy for different polluting gases from a car. However, there are still authors who favour the tax being imposed directly on the polluting activity, even when this is a challenge (Oates 1995).

Classification according to tax base

Classification according to the tax base is in essence a more profound version of classification by point of application. For example, the OECD uses this classification, distinguishing among such bases as petrol, diesel, coals, coke (and other energy carriers), sales and registration, annual use of motor vehicles, etc (EEA 2000).

In addition to these named categories, Ekins and Dresner (2004) have created an alternative environmental tax classification:

- upstream charges on resource use or environmental emissions;
- downstream charges on resource use;
- downstream charges on environmental emissions.

'Upstream' in their system means an early production process and 'downstream' is the final consumer. The authors argue that the response to a tax depends on whether it is applied upstream or downstream. For example, a downstream response to an upstream environmental taxation is a reduction in consumption of a taxed good.

In addition to straight effects on prices induced by environmental taxes, it is important to consider the indirect effects. Usually there is also a pass-through effect on downstream goods and services (Serret & Johnstone 2006). For example, the prices of fuel and water are transferred to the prices of different consumer goods.

In this thesis, classification of environmental taxes according to field of operation (energy taxes, transport taxes, taxes on pollution and natural resources) is used, as this is also used in the statistics that the empirical section is based on. In addition, classification according to the main objective (cost-covering, incentive and fiscal taxes) is referred to in discussion of the essence of the analysed taxes.

In order to select a suitable definition of environmental tax for the thesis, the ones used by different international organizations are now provided. Eurostat (2001) defines an environmental tax as a tax whose base is a physical unit (or a proxy of it) of something that has a proven, specific negative impact. According to the European Environment Agency, environmental taxes are compulsory payments levied on tax bases deemed to be of particular environmental relevance (EEA 2002). The OECD (2001) defines environmentally related tax as any compulsory, non-refundable payment to a general government levied on tax bases deemed to be of particular environmental relevance. These definitions are similar and bring out the following important issues, which are also used in this thesis: an environmental tax is a compulsory payment and its tax base is related to a negative environmental impact. The next section gives an overview of how environmental taxes are related to distributional issues.

1.2. The theoretical background to the linkage between environmental taxes and distributional issues

Before discussing the linkages between environmental taxes and distributional issues, a brief overview of the development of economic thought regarding income distribution is provided. This is not a detailed discussion of the relevant theories, as this is not the main focus of the thesis, but rather an understanding of the context for the distributional effects evaluated in the empirical part of the thesis. After this overview, the section discusses specific linkages between environmental taxes and distributional issues.

In general, the theories of income distribution can be divided in two types: positive and normative theories. The positive theories of income distribution date back to the 1770s to the classical school of economics. The focus then was mainly on the distribution of income between the main factors of production: labour, capital and land, the income types being wages, profit and rent accordingly. The theory regarding income distribution among individuals was less advanced, as the representatives of the classical school did not deal with the distribution of ownership and believed that this was determined by historical processes, not a subject for economics (Sandmo 2013). The well-known contribution of neoclassical economics is the idea that marginal productivities determine the wage rate and interest rate and a more disaggregated view of the labour market than classical economics had held. However, it was only the later generation of neoclassical economics that made an important contribution to the theory of income distribution, explaining wage differences. The 1870s also marked the start of the spread of human capital theory explaining wage differentials by education and training that increase productivity (*ibid.*). Scholars' focus shifted from functional distribution to personal income distribution, since it was understood that wage differences contribute more to income inequality than do incomes from land and capital (Goldfarb & Leonard 2005). The shift was also enabled by the availability of new data: personal income distribution analysis is based on micro-data, which had not been available before.

From a different perspective, studies of income distribution are often related to a perception that income distribution is unequal and hence they employ a value judgement. These theories go further than just explaining the differences and represent a normative approach. These theories generally demonstrate that large income inequalities are wasteful, i.e. a lower level of welfare is produced from unequal income distribution than in cases of more equal distribution (Sandmo 2013). Surprisingly, this appears in the writings of Adam Smith, who proposed that lower prices should apply to necessities and higher ones to luxuries, but he did not elaborate further on the possible instruments for that (*ibid.*). At the same time, Adam Smith's theory of the invisible hand is very well-known, according to which a competitive market achieves an efficient allocation of resources. Efficient resource allocation is assessed according to the Pareto criterion: a decision should be implemented if it makes someone better off and no one worse off. The problem is that in real life, it is almost impossible to implement decisions that make no one worse off. Hence, the potential Pareto improvement (also known as the Kaldor-Hicks criterion) is used instead: an improvement should take place if the winners can compensate the losers, even if this does not actually take place. It is important to note that the costs and benefits are specified on an aggregate level: when the sum of the benefits of a policy is greater than the sum of the costs, then the policy is desirable, and the ranking of policies is based on this difference. This is also called the utilitarian rule, which started from Jeremy Bentham's work in 1789 and several utilitarians following (Sen 1999) and which aims to maximise the sum of the individual utilities (Sen & Foster 2003):

$$(3)\ W = \sum_{i=1}^{n} U_i(x)$$

where:
W – social welfare function;
$U_i(x)$ – utility function of an individual.

The use of the sum of utilities is based on the approach of new welfare economics, which divides economics in two parts: the first relating to production efficiency and the second to distribution. According to Kaldor (1939), for the first part the economist is on sure ground, relating increasing social welfare to an increase in aggregate efficiency in production. For the second part, the economist should not be concerned, for it is impossible to decide on economic grounds which particular pattern of income distribution maximises social welfare. Bromley (1990) states that this approach reinforced the idea that economics was about increasing the production of goods and services (commodities) and not about people and their relationships to each other. Sen and Foster (2003) point out that not everyone has the same utility function (for example, due to some disability) but even if they did, this would not take into account the inequality of the utility levels of different individuals.

Nyborg (2012) notes two problems with the Kaldor-Hicks criterion: firstly, it is difficult to separate efficiency and distribution; ex post redistribution is complicated and expensive and so the redistribution should already be targeted at project design. But if this is the case, then the project is already about Pareto improvement, not potential Pareto improvement. The second problem is that the Kaldor-Hicks criterion gives same weight to everyone's welfare; as shown by Nyborg (2012), this places more emphasis on those interests that value money less, i.e. the rich.

A widely known and stringent approach is suggested by Rawls (1971): "All social values – liberty and opportunity, income and wealth, and the bases of self-respect – are to be distributed equally unless an unequal distribution of any, or all, of these values is to everyone's advantage." He suggests that welfare maximisation means maximisation of the income of the poorest person, without regard for the incomes of the others. However, Rawls' approach to equity is not widely applied in actual policy decisions as a rule.

Another problematic issue is the individualistic nature of social welfare, which means that social welfare is a function of individual utilities, where U_i stands for the utility function of the individual i:

(4) $W(x)=F(U_1(x), \ldots, U_n(x)$

Assuming that W increases with any U_i given the set of utilities of all other individuals, the maximisation of social welfare carries the essence of Pareto optimality. But the idea of the social welfare function is to go beyond Pareto optimality. According to Sen and Foster (2003): "It seems reasonable, therefore, to argue that if the approach of social welfare functions is to give us any substantial help in measuring inequality, or in evaluating alternative measures of inequality, then the framework must be broadened to include interpersonal comparisons of welfare."

To understand this, the utility function deserves closer attention. Traditionally, the utility function has been specified as an ordinal utility function, i.e. individuals are able to rank alternative states of the world. However, as shown by Arrow's impossibility theorem, even if one person prefers one state to another and everyone else has the opposite preference, the two states must be declared equal from the societal point of view (Sen & Foster 2003).

If we want to analyse distributional implications and decide whether a loss for one person is more important than a gain for another, we need the cardinal utility concept, i.e. a utility concept saying not only whether something is preferred to something else, but how strongly it is preferred (Nyborg 2012). Furthermore, we must assume that cardinal utility is comparable between individuals.

As a proxy for social wellbeing or welfare, income is often used: the utility function U(y) means that utility depends on an individual's disposable income y. The reason is often pragmatic: there is not enough information on individual utility functions (Sen & Foster 2003). It is widely accepted that utility functions U(y) are concave, meaning that the marginal utility of income falls as income increases. Hence the social welfare function, which is a function of individual utility functions, shows a society's aversion to inequality: for example, the utilitarian function is linear, as it is concerned with total welfare, regardless of whether the income receiver is rich or poor. An alternative is concave (see Figure 1.2.1): if an amount Δy is taken from a rich person with income y_2 and transferred to a poor person with income y_1, the increase in social welfare (d_2) is greater than the decrease in social welfare (d_1).

Although distributional analysis is usually limited to income distribution, various authors stress that this might not give an adequate picture. For example, Gowdy (2005) stresses, based on a literature review, that income is a poor measure of welfare and discusses the following relationships between income and happiness: (1) people in wealthier countries are generally happier than people in poorer countries; (2) beyond a certain stage of development, increasing incomes do not lead to greater happiness; (3) security seems to be a key element of happiness; (4) mental health is a crucial factor in happiness; and (5) richer social relationships generally make people happier. Kriström (2006) suggests using environmental quality or wealth as a proxy for quality of life, which includes real, human and social capital.

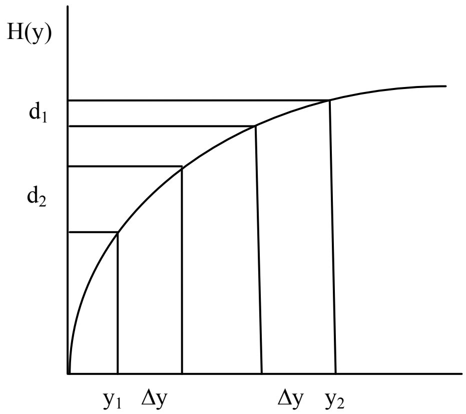

Figure 1.2.1. Social welfare and concavity (Creedy 1996)

To be more specific and include the environmental aspects of utility function, utility does not depend equally on all goods, but can be separated according to the environmental effects:[2]

(5) $U_i = u_i(c_i, d_i, nd)$

where:
c_i – a clean good;
d_i – a dirty good;
nd – amount of aggregate consumption of dirty goods.

Utility depends positively on c_i and d_i, and negatively on *nd*. While the consumption of some goods can be chosen independently by an individual (although some restrictions exist also in this case – for example, limited information or acting as an agent), the level of environmental quality (expressed as *nd* in the utility function) cannot be independently decided on, as this depends on the consumption of the others. The goods that generate external effects negatively affect the utility function and hence the environmental taxes that are designed to internalise these externalities have different implications: positive implications for the ones affected by externalities and negative implications for the ones producing these. But, as discussed in the previous section, although the

2 Source: Fullerton and Wolverton (2005) with the author's modifications

objective is to internalise any externalities in price, this cannot necessarily be fully accomplished as the instruments are second-best.

Furthermore, as environmental taxes are also taxes, the criteria that are applied according to tax policy are different from the ones for environmental effectiveness. For example, according to Mirrlees et al. (2011), the objectives of a tax system are the following:

- minimised negative effects of the tax system on welfare and economic efficiency;
- low administration and compliance costs;
- fairness other than in the distributional sense (fairness of procedure, avoidance of discrimination, etc);
- transparency.

Stiglitz (1988) defines the properties of a good tax system as economic efficiency, administrative simplicity, flexibility, political responsibility and equity. Hence a tax should be easy to administer and it should not hinder the efficient allocation of resources according to good tax policy criteria.

But for environmental taxes, the criterion of environmental effectiveness should apply as well, as their theoretical background is based on externalities. Environmental effectiveness means to what extent it achieves the environmental objective set (i.e. how much it reduces externalities in the form of environmental pollution). One might argue that distributional issues should not be of concern for environmental policy, which is aimed at environmental quality improvement. But as they are taxes, the (re)distributional issues are inevitably involved as well. It is important to note that since poor people might not have enough resources to respond to environmental taxes (as also noted by Fitzpatrick 2011a), these disadvantages might reinforce each other. It is also the case that a policy is more readily implemented and gains more support if it is not perceived as unfair (Baumol & Oates 1995, Serret & Johnston 2006).

According to Serret and Johnston (2006), the linkages between environmental policy and distribution include two different aspects:

- the distribution of environmental quality;
- the distribution of financial effects resulting from environmental policy.

The first dimension is concerned with the relative quality of the environment that people experience. The starting point of this discussion is that all groups in society should live in and have access to a good quality environment (Pye et al. 2008). There is some evidence that environmental quality is regressively distributed across socio-economic groups, meaning that low-income households experience higher environmental risks than high-income households. This research, however, is mostly from the USA and not so much from Europe (Pearce 2006). There are many issues involved, which complicates the assessment; for example, such choices may be intentional: low-income households might have a lower demand for environmental quality than high-income groups, or alternatively, higher environmental risks may have associated benefits, for example, lower property prices.

Although these issues important to consider, the problem is that usually there is no good data for analysing the issue of environmental quality. Hence this linkage between environmental taxes and environmental quality is not covered in this thesis. This thesis concentrates on the second dimension mentioned above, the distribution of the financial effects of a given policy.

If the objective of an environmental tax is to address sustainability issues, it is not enough to look at the distributional issues among living generations; we must also take into account future generations. It has been argued that the heaviest load of an environmental tax should fall on the generations which decide on it (an overview of such studies is provided in Chiroleu-Assouline & Fodha 2010). However, the intergenerational issue is out of the scope of the current thesis.

It is also questionable whether analysing only some taxes gives an adequate picture, as in the end the broad impact of different taxes, subsidies and other measures is important (see, for example, Creedy 1998 for discussion). However, concentrating on a specific type of taxes can give useful information about the effects of the specific policy and is helpful in addressing the undesirable effects.

In a way, distributional analysis of a certain policy is also in line with the approach proposed by Nyborg (2012), who does cost–benefit analysis of environmental projects. She suggests that using aggregate cost or aggregate benefit indicators for the approval of certain project is not enough and should be complemented by background information about groups, such as income level, factors associated with vulnerability (age) or wellbeing (health), etc.

The linkage between income and the tax burden in economic theory is based on the concepts of horizontal and vertical equity. Horizontal equity means that people in equal position should pay equal amounts of tax. Vertical equity means that tax payments should be differentiated based on the ability to pay: those who are able should pay more. The issue is related to the type of taxed good. As also discussed by Sterner (2012), the progressivity or regressivity of taxation is related to the consumption pattern of a taxed good (Figure 1.2.2). The relationship between the consumption of a good and income might not always be linear (as in the case of product C in Figure 1.2.2) but decelerating, as in the case of product B, or accelerating, as in the case of product A. Product A is also called a luxury good and using demand elasticity, this is expressed by an elasticity higher than unity.

Hence the income elasticity of a good is related to the potential progressivity or regressivity of a tax: if the demand elasticity is higher than unity, the taxed good is a luxury and taxation affects richer people more. If it is lower than unity, the taxed good is a necessity and taxation might affect poorer people more, depending on the consumption pattern: if the proportion of a taxed good in the budget decreases with income, this might indicate a regressive pattern.

The different indices used to analyse the distributional impacts of taxation can be classified into two broad groups: descriptive measures and normative measures of inequality. Descriptive measures of income distribution are much more widespread in analysing the distributional effects of tax policies. These measures are mostly based on the Lorenz curve developed by Max O. Lorenz in 1905, which shows the deviation of each person's income from perfect equality (Kakwani 2010). Although Lorenz himself criticised the use of numerical calculations and suggested using the graph only for descriptive purposes, various authors following him, specifically Corrado Gini, stabilised the form of the curve and made use of numerical calculations (Derobert & Thieriot 2003). The Lorenz curve has also been criticised as it makes distributional judgements under the assumption that the two distributions have the same mean incomes, but in reality this situation is usually not the case (Kakwani 2010).

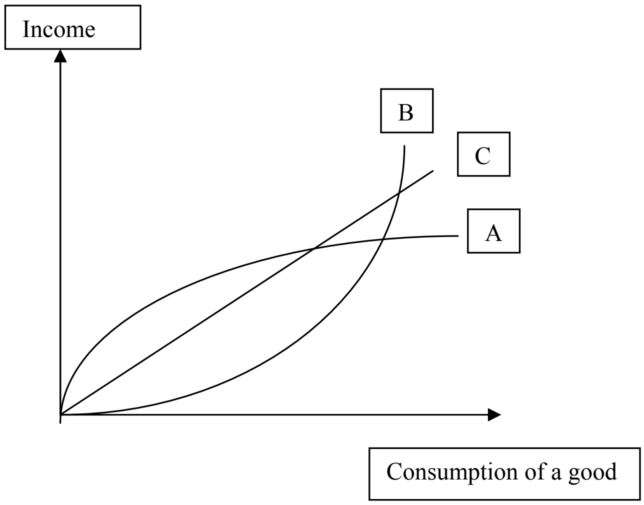

Figure 1.2.2 The relationship between income and demand for different goods

Hence descriptive measures usually rely on some form of the Gini index (pre- and post-tax) or concentration curves, for example, the Reynolds-Smolensky index and Kakwani index (Creedy 1999). These measures are summary measures and their limitations are clear. Some authors propose that instead of evaluating the summary statistics, the detailed changes between two distributions should be looked at, for example, the ranking issue – the location of families within the distributions (Atkinson & Stiglitz 1980). In this thesis, microsimulation is also applied to discover distributional effects according to income groups and various household characteristics, to find out how different household types are affected. A detailed overview of the methodological issues of the thesis is presented in chapter 3.

The other approach to analysing changes in inequality is normative. The distinction is not very clear-cut, as the Lorenz curve can also be used from a normative perspective. This interpretation was done by Hugh Dalton in 1920;

he made a utilitarian assumption that "the economic welfare of different persons is additive, that the relation of income to economic welfare is the same for all members of the community, and that, for each individual, marginal economic welfare diminishes as income increases" (Dalton 1920, cited in Derobert & Thieriot 2003). This allowed him to conclude that maximum economic welfare is guaranteed when all incomes are equal and the equality line is a situation that societies should move towards (Derobert & Thieriot 2003).

A well-known normative measure in inequality measurement is the Atkinson index, developed in 1970, which is based on the concept of the equally distributed equivalent level of income (Atkinson 1970). The formula includes the degree of inequality-aversion or the relative sensitivity to transfers at different income levels. Kaplow (2005) argues that normative measures of inequality like the Atkinson index are not very useful, since one must first undertake a complete analysis of social welfare as a prerequisite to measuring inequality. This means that one must choose a specific social welfare function in order to define such an inequality index. In this thesis also, the emphasis is on descriptive inequality measures, but in addition the normative measure of the Atkinson index has been applied to demonstrate how sensitive the distributional implications are to a change in inequality-aversion in a society. A detailed overview of the applied inequality measures is provided in the methodology chapter, together with the formulas for calculation.

A well-known issue in the distributional analysis of different policies is the tax incidence question: who actually bears the tax burden, the producer or the consumer? A thorough history of tax incidence analysis is given by Metcalf and Fullerton (2002). The authors show that the main assumption in partial and general equilibrium analysis of tax incidence is that the side of a market that is relatively price inelastic bears a greater burden of the tax than the more price-elastic side. Thus it is dependent on the demand elasticity but also the market power of a producer/seller. In environmental tax incidence, it is assumed that the costs will be shifted to consumers, as most environmental taxes apply to the energy sector, which is usually monopolistic or oligopolistic (Bork 2006), and energy consumption is considered rather inelastic, at least in the short term.

Hence, the consumption of environmentally related goods affects welfare differently depending on whether this is individual or total consumption level: although it has a positive effect for the one consuming, it might have a negative effect in society due to externalities. Furthermore, if externalities are internalised via taxes, the criteria for a good tax might be different than from an environmental policy standpoint. Taxes might have redistributional effects but also adverse effect on poorer households, as specifically exemplified by environmental taxes. It is not clear-cut how to measure welfare changes: should this be based on descriptive or normative measures, are summary measures sufficient and what is a good proxy for welfare? In empirical research of the distributional analysis of environmental taxes, usually income level is used. A review of such studies is presented in the next chapter.

2. EMPIRICAL STUDIES ON THE DISTRIBUTIONAL AND BEHAVIOURAL EFFECTS OF ENVIRONMENTAL TAXES

2.1. Empirical studies on direct distributional effects

The issue of the distribution of the environmental tax load has been of interest for decades. Poterba (1991) studied the distributional implications of gasoline (petrol) tax in the USA. He argues that expenditure data should be used instead of income to analyse distributional effects, as it is a more reliable variable to reflect the actual living standard of a household and which smoothes lifetime consumption. He finds that while the distributional pattern of gasoline tax as a ratio to income is regressive, it is not so if measured as a ratio to expenditure: in that case, the highest load of gasoline tax falls on middle-class households.

In Europe, the empirical literature on the direct distributional effects of environmental taxes can be traced back to 1991, when Pearson and Smith estimated the distributional impact of a potential carbon tax in seven European countries (France, Germany, Italy, the Netherlands, Spain, the UK and Ireland). In the first five they found that the burden of the carbon tax payment was only weakly related to income, if at all, but in the UK and Ireland there was evidence of a significantly regressive pattern. The work of Pearson and Smith was upgraded by Barker and Köhler in 1998 using the European energy–environment–economy model (E3ME). The countries covered were Belgium, Spain, the Netherlands, Ireland, Italy, Luxembourg, Portugal, France, Germany, Great Britain and Denmark. The researchers analysed the impact of of revenue-neutral ETR and found that the taxation of fuels used for domestic heating was regressive, but if only transport fuels were taxed, the tax reform would be progressive in most of the studied countries (Barker & Köhler 1998).

The most recent research about the distributional implications of a carbon tax in Europe has been done by Ekins et al. (2011) with the E3ME model. This research is based on the simplifying assumption that the consumption function is identical for all groups and there are no behavioural effects. The results show that there is an increase in real income due to the ETR in all groups in all of the analysed scenarios in 2020 at the EU level. The logic is that there is a need for more aggressive carbon reduction, requiring higher carbon taxes, which then yield higher revenues that are available for reducing income taxes, which results in greater increases in real income. Regarding different socio-economic groups, the unemployed and inactive groups experience the smallest increases in real income. Urban households see a larger increase in real income than rural households. It is also evident that there is much more difference in the changes in income between countries than within countries. In some countries, it is the third income quintile that benefits least from the policy and this is the case for the EU as a whole. But in some countries (Greece, Spain, Ireland and Hungary) a fall or no change in real income is reported in the lowest income quintile and the highest income group experiences the largest increase in real income, which suggests that income distribution becomes more unequal due to an ETR, even if this is revenue-neutral.

In addition, there have been several studies of the distributional impact of specific environmental taxes in single countries. In general, the conclusion seems to be that if the tax is posed on fuel used for domestic heating, the lower income groups bear a proportionally higher tax burden than high income groups. Such evidence has been found for Germany (Bork 2006), Great Britain (Dresner & Ekins 2006) and Ireland (Callan et al. 2009). In the case of Germany, taxes on electricity, natural gas and heating oil are clearly regressive. For taxes on motor fuels, the result is different: for the first four income classes defined in the study the tax load increases, but it decreases afterwards (Bork 2006). So the highest environmental tax load is born by middle-income groups and the lowest tax load by the highest income groups, which means that the general pattern of the motor fuel tax is regressive. The reduction in social security contribution rates also strengthens the regressive effect. It must be taken into consideration that other tax reforms have taken place together with the ETR in Germany, income tax reform and child benefits increase, which have neutralised the burden of the ecological taxes in most cases. Only some pensioners and a few households of unemployed, students and other non-employed people bear on average a higher burden after all the reforms (*ibid.*).

In the case of Great Britain, the carbon tax imposed on gas and electricity is regressive, but when compensation schemes are used then the tax system on average is progressive. However, as the variation of the tax burden in low-income deciles is very large, there is still a significant proportion of low-income households that are losing from the reform (Dresner & Ekins 2006).

In Ireland, the carbon tax is regressive, but a modest increase in welfare payments (pensions, unemployment compensation, etc) and tax credit increases would offset the negative effects of the tax in the lower half of the income distribution (Callan et al. 2009). The regressivity of the carbon tax mainly comes from the consumption of electricity and heating fuels, as the tax load resulting from motor fuel consumption increases with income.

No evidence of regressivity of carbon tax has been found in a study of Italy (Tiezzi 2005), which can probably be explained by the tax's greater impact on motor fuels and lesser impact on domestic heating fuels. A study of Denmark considered more taxes than just carbon and energy taxes and shows that the result depends on the variable used: according to disposable income, environmental taxes are regressive, but they are progressive according to expenditure (Jacobsen et al. 2003). The pattern also differs according to tax type: for example, transport-related taxes are progressive, while energy taxes and other green taxes are proportional or mildly regressive (*ibid.*).

The studies also investigated the distributional effect of environmental taxes in different socio-demographic groups and have found some important relationships, for example, according to family type. Bork (2006) finds that the tax load for single people without children is the lowest and for couples with children it is the highest. Jacobsen et al. (2003) find that the groups paying the highest share of environmental taxes are the self-employed and the early-retired, with the exception of electricity and water, where students pay the highest share. Dresner and Ekins (2006) find households with children and pensioners to be the most vulnerable groups. Callan et al. (2009) conclude that the carbon tax would hit people in smaller households harder.

Most of the studies also state that in cities, the environmental tax load is lower if compared to rural areas (Bork 2006, Callan et al. 2009, Jacobsen et al. 2003).

Recently, a specific piece of literature studying the distributional impacts of petrol tax has emerged (Sterner 2012). This book provides several chapters for developing countries from Latin America, Asia and Africa, in addition to more traditional country coverage like the USA and some developed European countries. Sterner and Carlsson (2012) have assessed the distributional impact of motor fuel taxes in seven European countries: France, Germany, Italy, Serbia, Spain, Sweden and the United Kingdom, and found that the distributional patterns of fuel taxation are mixed: if using expenditure levels, the motor fuel tax is slightly progressive or proportional in most of the studied countries, but regressive for example in Italy. The book includes a special chapter on the distributional effects of fuel taxes in the Czech Republic; this analysis shows that fuel taxation is somewhat regressive, but not as regressive as taxation on energy as a whole, and that taxation on public transport services is more regressive than taxation on fuels, showing that personal vehicles are more used by high-income households (Ščasnŷ 2012). Regarding the general conclusions for the countries analysed in Sterner (2012), the authors conclude that in most cases fuel taxation is progressive; regressive patterns are found only in Italy, the USA and Mexico (Sterner et al. 2012).

An overview of the mentioned studies is provided in Table 2.1.1. Most of the named studies of direct distributional effects use a national household expenditure survey, which is also called a family expenditure survey, household budget survey, etc. Regardless, this is data collected by national statistical offices to record household characteristics, income and expenditure. In some cases additional data has been used; for example, Dresner and Ekins (2006) also use a house condition survey. In Jacobsen et al. (2003), a special "law model" was used, which was created for the Danish government to examine the distributional aspects and revenue consequences of both existing and proposed legislation.

Table 2.1.1. Studies of direct distributional effects of environmental taxes

Author, year	Country(ies)	Tax(es) analysed	Methodology	Main result regarding regressivity/progressivity
Poterba (1991)	USA	Gasoline tax	Microsimulation based on HBS	Regressive according to income, falling mostly on middle-income classes according to expenditure
Pearson & Smith (1991)	France, Germany, Italy, the Netherlands, Spain, the UK and Ireland	Carbon tax (hypothetical)	Microsimulation based on Eurostat data	Generally no clear relation, but for UK and Ireland significantly regressive pattern
Barker & Köhler (1998)	Belgium, Spain, the Netherlands, Ireland, Italy, Luxembourg, Portugal, France, Germany, Great Britain and Denmark	Carbon tax in a framework of a revenue-neutral ETR (hypothetical)	E3ME	Taxation of fuels used for domestic heating is regressive, but if only transport fuels were taxed, the tax reform would be progressive
Jacobsen et al. (2003)	Denmark	Different environmental taxes (implemented)	Microsimulation based on HBS + “law model”	Regressive according to income, progressive according to expenditure
Tiezzi (2005)	Italy	Carbon tax (ex-ante assessment of implemented tax)	AIDS (almost ideal demand system)	Progressive
Bork (2006)	Germany	Taxes on energy (implemented via ETR)	Microsimulation based on HBS	Taxes on fuels used for domestic heating are regressive, for motor fuels the highest tax load is on middle-income groups
Dresner & Ekins (2006)	The UK	Carbon tax (hypothetical)	Microsimulation based on HBS + house condition survey	Taxes on gas and electricity are regressive, but together with compensation schemes are progressive in general
Callan et al. (2009)	Ireland	Carbon tax (hypothetical)	Microsimulation based on HBS	Regressive, but welfare payments would offset the negative effects
Ekins (2011)	All 27 EU member states	Carbon tax (hypothetical)	E3ME	In most countries not regressive, but regressive in Greece, Spain, Ireland and Hungary

Author, year	Country(ies)	Tax(es) analysed	Methodology	Main result regarding regressivity/progressivity
Sterner & Carlsson (2012)	France, Germany, Italy, Serbia, Spain, Sweden, the UK	Petrol and diesel excise tax (implemented)	Microsimulation based on HBS	Regressive when using income data. Mostly proportional or progressive when using expenditure data, but stays regressive for Italy
Ščasnŷ (2012)	Czech Republic	Different existing taxes and policy scenarios	Microsimulation based on HBS	Fuel excise is regressive, but less regressive than public transport taxation and VAT on energy
Sterner et al. (2012)	Different countries from Europe, Latin America, Asia and Africa	Existing fuel taxes	Concluding chapter based on country cases using different methods	Mostly neutral or slightly regressive in developed countries (turned into progressive via revenue recycling), progressive in developing countries

Source: compiled by the author

The most common methodology used in assessing the direct distributional effects of environmental taxes is microsimulation, but if analysing several countries, computable general equilibrium models are also used.

Most of the studies on direct distributional effects use the percentage of tax load in income/expenditure as the measurement unit, except for Callan et al. (2009), who compare tax payments in euros across income deciles. The study by Jacobsen et al. (2003) also discovers a change in the Gini coefficient as a result of different taxes. The authors conclude that three of the environmental taxes applied in Denmark are reducing inequality: tax on insecticides and herbicides, registration duty and petrol tax. As a category, transport-related taxes are reducing inequality, but energy taxes are increasing inequality. Considering all the environmental taxes together, these increase inequality, but less than for example VAT or excises on alcohol and tobacco (Jacobsen et al. 2006).

The application of normative inequality measures in environmental policy is rather rare, but done for example by Schlör et al. (2012), who study energy expenditures and inequality in Germany based on the Atkinson index.

While some of the studies use actual taxes or ETR, for example, Jacobsen et al. (2003), Tiezzi (2005) and Bork (2006), some use only hypothetical carbon taxes: Dresner and Ekins (2006) and Callan et al. (2009). The issue of the fair distribution of the environmental tax load has evolved to a large extent with the issue of climate change and the international conventions to mitigate this. The exception, in the sense that not only carbon or energy taxes are analysed but all existing environmental taxes, is the work of Jacobsen et al. (2006).

An important methodological question that arises is which income measure to use in distributional analysis. For example, Jacobsen et al. (2003) show that the regressivity of environmental taxes appears when using income data, but not if expenditure data is used. As discussed in Sterner (2012), expenditure level is believed to be a better proxy for lifetime income and it has been suggested that expenditure level should be used in assessing the distributional implications of consumption taxes (Creedy 1998). It is assumed that the distribution of lifetime income displays less inequality than one based on annual income (Fullerton & Rogers 1993).

To summarise the research findings on direct distributional effects, it is usually found that poorer people pay a greater proportion of environmental taxes in relation to their income level; this is shown by Jacobsen et al. (2003), Bork (2006) and Dresner and Ekins (2006). However, Jacobsen et al. (2003) find that environmental taxes are not less regressive than the consumption tax VAT. Different authors have also demonstrated that with the help of compensation schemes, the regressive effect can be reversed or minimised.

As for specific household types vulnerable to environmental taxes, different authors have shown that for households living in cities, the environmental tax load is lower than for households living in rural areas (Bork 2006, Callan et al. 2009, Jacobsen et al. 2003). The household types defined as vulnerable are households with children (Bork 2006, Dresner & Ekins 2006), the self-employed (Jacobsen et al. 2003) and retired people (Dresner & Ekins 2006).

However, the distributional effects also depend on the type of tax. In the case of taxes on heating fuels and electricity, a regressive effect has been shown, for example by Barker and Köhler (1998), Jacobsen et al. (2003), Bork (2006), Dresner and Ekins (2006) and Callan et al. (2009). Contrarily, taxes on vehicles are found to be progressive (Jacobsen et al. 2003). The evidence of the distributional impact of motor fuels taxes is mixed: it has been found to be progressive (Jacobsen et al. 2003, Tiezzi 2005), falling mostly on middle-income groups (Bork 2006) and even regressive (in the case of Italy in Sterner et al. 2012). Sterner et al. (2012) have been found that it is more likely to be slightly regressive or neutral in developed countries and to be progressive in developing countries. Hence, there is no consensus in the literature regarding the distributional effects of motor fuel taxes. Furthermore, if looking at environmental taxes more broadly (other than petrol taxes), the issue has not been studied at all in the new member states of the EU.

2.2. Empirical studies on indirect distributional effects

The indirect distributional effects of environmental taxes mean that taxes on production processes or intermediate consumption are transferred to consumer prices and then the distributional effect on households is assessed. As a meta-analysis of ETR effectiveness has shown, almost all studies have shown higher price levels compared with the base scenarios (Patuelli et al. 2005). This is called an inflationary spiral, which comes from an increase in the price of energy, which in turn raises the price of products. Therefore it is also important to study the effects of price increases on households.

In this approach, input–output tables are used together with the micro-simulation method. The examples are studies conducted for Canada (Hamilton & Cameron 1994), the UK (Symons et al. 1994), Australia (Cornwell & Creedy 1996), Spain (Labandeira & Labeaga 1999), Denmark (Wier et al. 2005), the Netherlands (Kerkhof et al. 2008) and the USA (Grainger & Kolstad 2010).

Most of these papers deal with distributional issues of hypothetical carbon taxes. Again, the only exception is the paper by Wier et al. (2005) for Denmark, which analyses the actual CO_2 tax. Also a common feature of these studies is that different methods have been used within a single paper to answer different research questions. All of the studies use input–output tables to analyse the effects of environmental taxes on prices. The distributional effect is then calculated with microsimulation models. Three of the studies also analyse the behavioural effects (Cornwell & Creedy 1996, Labandeira & Labeaga 1999, Symons et al. 1994). These studies use some form of AIDS (almost ideal demand system) to include the behavioural effects.

As can be seen from Table 2.2.1, the results of the studies vary greatly because the actual or hypothetical level of tax is very different; this comes from the approach used: in some studies, an actual damage estimate is used (Labandeira & Labeaga 1999), which is considerably lower than in those studies that calculate the CO_2 tax level necessary to achieve the specified target (Cornwell & Creedy 1996, Symons et al. 1994). This is why in Hamilton and Cameron (1994) the applied level of carbon tax is USD$27.70 per tonne of CO_2 but in Kerkhof et al. (2008) the tax level necessary to achieve the Kyoto Protocol target was calculated to be €91 per tonne of CO_2 (about USD$129). In the study by Grainger and Colstad (2010), the price of CO_2 is USD$15. Wier et al. (2005) use the actual CO_2 tax level, for which the standard rate at the time of the study was €81 for households and €13.5 for the business sector, although reduced rates applied to energy-intensive sectors.

The aggregation levels are also very different: the earlier studies have around 30 sectors in their input–output tables (Cornwell & Creedy 1996, Symons et al. 1994), while in the latest studies there are more than 100 sectors (Kerkhof et al. 2008, Wier et al, 2005).

Due to the differences in aggregation and tax levels, the obtained results are also very different. In Hamilton and Cameron (1994), the price increase induced by a carbon tax is 1% in the primary and manufacturing sector, 0.2% in construction, 2.2% in transportation and 1.5% in services. Symons et al. (1994) show the carbon tax has the greatest effect on household energy with a price increase 79%; the price of motor fuel also increased substantially (34.7%). The largest indirect price effect occurred in ceramic ware (8.5%), transport (5.2%) and food (2.9%), caused by the energy intensity of these sectors.

According to Cornwell and Creedy (1996), in Australia the carbon tax has had the greatest impact on the fuel and power sector (price increase 1.3%), but also on food (0.1%), and alcoholic beverages and tobacco (0.1% for both).

In Spain, the effect of the carbon tax has been assessed by Labandeira and Labeaga (1999), who demonstrate the highest price increase for electricity (3.8%), natural and manufactured gas (3.2%) and fuel for private transport (2.7%).

Grainger and Colstad (2010) calculate the largest cost increase for lime manufacturing (14.8%), power generation and supply (11.2%) and cement manufacturing (8.3%) in the USA.

In the paper by Wier et al. (2005), the distributional effect of the Danish CO_2 tax has been assessed. Direct household tax payments are associated with energy commodities and electricity is the most taxed energy type. The commodities with the highest indirect CO_2 tax liabilities are water (0.39%), package holidays (0.38%) and dairy products (0.26%). Looking at the distributional implications of the CO_2 tax, the authors conclude that CO_2 taxes are more regressive than other consumption taxes (for example VAT) and direct CO_2 taxes are more regressive than indirect CO_2 taxes. However, petrol tax is found to be progressive.

Kerkhof et al. (2008) find that in the Netherlands the highest price increase occurs in product groups of other costs: heating and lighting (101.5%), electricity (49.4%), gas including solid and liquid fuels (35.4%) and petrol and oil (28.3%). In addition, high price increases also occur for fish (11.4%), caused by fuel combustion on the ships used for fishing, gardens and flowers (6.8%) and vegetables (5.6%) due to fossil fuel use in glasshouses.

Table 2.2.1. Overview of studies of indirect distributional effects of carbon taxes

Author, year	**Country**	**Methodology**	**Tax level (USD per ton of carbon)[3]**	**Sectors which are affected most**	**Distribu-tional impact**
Hamilton & Cameron (1994)	Canada	Input–output model + microsimulation	102	Transportation	Regressive
Symons et al. (1994)	The UK	Input–output model + AIDS	411	Household energy, petrol, durables: ceramic ware	Regressive
Cornwell & Creedy (1996)	Australia	Input–output model + AIDS	306	Fuel & power, food, alcoholic beverages, tobacco	Regressive
Labandeira & Labeaga (1999)	Spain	Input–output model + microsimulation	20.3	Electricity, natural and manufactured gas, fuel for private transport	Proportional
Wier et al. 2005	Denmark	Input–output model + microsimulation	66	Water, package holidays, dairy products	Regressive
Kerkhof et al. 2008	The Netherlands	Input–output model + microsimulation	473	Product groups: heating, electricity, petrol but also fish, gardens and flowers, vegetables	Regressive
Grainger & Colstad (2010)	The USA	Input–output model + microsimulation	55	Lime manufacturing, power supply, cement manufacturing	Regressive

Source: compiled by the author

[3] Calculated in USD by the author

Most of the studies find that the carbon tax is regressive: such results are confirmed by Hamilton and Cameron (1994), Symons et al. (1994), Cornwell and Creedy (1996), Wier et al. (2005), Kerkhof et al. (2008) and Grainger and Colstad (2010). Most of the studies use the household as the unit for calculating the progressivity or regressivity of a carbon tax. Grainger and Colstad (2010) show that per capita incidence, using equivalence scales, shows much more regressivity than calculations at the household level. The same is concluded by Wier et al. (2005).

Symons et al. (1994) find that if revenue-neutral tax reform is used, the carbon tax is progressive and decreases inequality. Only Labandeira and Labeaga (1999) in Spain find the carbon tax slightly progressive.

The results are also dependent on the energy intensity of the economy or sectors; for example, Yusuf and Resosudarmo (2007) show that for developing countries, a carbon tax need not be regressive, as a carbon tax has a greater impact on energy and capital-intensive sectors, but in the case of Indonesia, poorer people are typically living in rural areas and employed in the agricultural sector, which is not so much affected by the carbon tax. Hence, the overall impact of the carbon tax is not regressive.

In sum, the results of previous studies vary depending on the country, tax level, methodology applied, data sources etc. It can be concluded that the sectors of energy production and transportation bear the greatest load of the carbon tax, but also other energy-intensive sectors, for example, food and beverages (Cornwell & Creedy 1996), water and package holidays (Wier et al. 2005) and lime manufacturing (Grainger & Colstad 2010). As low-income households spend a higher share of their income on goods and services that are believed to be energy-intensive, i.e. home heating and electricity, this leads to adverse indirect distributional effects on poorer households. A regressive indirect distributional effect has been detected for all empirical studies presented in section 2.2 except for that of Labandeira and Labeaga (1999), who find that the effect is proportional, and also that of Yusuf and Resosudarmo (2007), who assess the impact in Indonesia.

Regarding the proportions of indirect and direct tax loads, the only authors who have discussed this are Wier et al. (2005); according to their analysis, the indirect tax payment equals about one-third of the direct tax payment.

To sum up the implications of the empirical research on the direct and indirect distributional effects, it seems more likely that if a tax is placed on fuels used for domestic heating, the direct distributional effect is regressive, but there is no consensus on regressivity or progressivity if motor fuels are taxed. Even if the direct effect is progressive, the indirect effect induced by environmental taxes is regressive. In addition, different household types may react differently to taxes, which is a topic for the next section.

2.3. Empirical studies on behavioural effects

Regarding the behavioural effects, it is hard to find literature assessing the pure effects of environmental tax on household behaviour. A question might be posed as to whether the reaction of a household to a price increase is different if it is known to be for good reason (environmental arguments) and not just a price change. This has been addressed by Ghalwash (2005), who has estimated whether the consumer reaction to a price change due to an environmental tax is different from one to a producer price change. The author divides the consumer price into producer price and tax and uses AIDS (almost ideal demand system) to assess the price and income elasticities of different product groups. The research was done with Swedish aggregate data and the author finds that consumers are more sensitive to energy taxes than to producer prices for most energy goods except petrol and public and other transport. He also finds that the tax elasticities for electricity, oil and district heating are in absolute value higher than unity, which means that for these goods, higher energy taxes will lead to relatively large reductions in consumption but also decreases in the budget share. On the other hand, the tax elasticities for petrol and public and other transport are less than unity, meaning that energy taxes are perhaps less effective in reducing pollution from transport than from heating and electricity.

As this work about the signalling effect of environmental taxes was done based on aggregate data, it has important limitations. Aggregate data does not enable differentiation of the effects according to some household characteristics, for example, income level, place of residence, household size, etc. Also, the price changes of energy products might differ in size; for example, the price of electricity is perhaps not as volatile as the price of petrol, which might explain some of the different price elasticities.

The following literature review is based on undifferentiated elasticities of energy products. As households' budgets are restricted and we expect them to optimise their welfare, then increasing prices cause changes in consumption patterns, regardless of the reasons behind the price change. The following review covers only motor fuels and does not include energy products used for domestic heating. The reason is that as electricity and gas excise were imposed in Estonia only in 2008 and several household heating fuels are not taxed with the excise, the focus is set on motor fuel taxes, which have been in place for a long time and have been raised considerably in the implementation period.

Behavioural effects should characterise the way people respond to price changes, so they can be measured by the price elasticity of a taxed good. There is a considerable amount of literature estimating the price and income elasticity of motor fuel. Historically, such analysis has been based on aggregate data showing that income elasticities are higher than price elasticities (Basso & Oum 2007). Meta-analysis of elasticities of road traffic and fuel consumption by Goodwin et al. (2004) finds that if the real price of fuel rises by 10%, the volume of fuel consumed will fall by about 2.5% within a year and by 6% in the longer run. If real income goes up by 10%, the total amount of fuel consumed will rise by nearly 4% within about a year and by more than 10% in the longer run. It is a general tendency that long-run elasticities are substantially higher than short-run effects, mostly by factors of 2–3, and income elasticities are greater than price elasticities, mostly by factors of 1.5–3. The review of Goodwin et al. (2004) also shows that the USA has lower income and price elasticities of fuel consumption than Europe.

Another review of road traffic demand elasticities has been done by Graham and Glaister (2004), who find that the mean short-run income elasticity of fuel demand is 0.47 and it is 0.93 in the long run. The mean short-run price elasticity is –0.25 and it is –0.77 in the long run. A similar result is found in the meta-analysis of Brons et al. (2008): the mean short-run price elasticity is –0.34 and it is –0.84 in the long run.

However, as Basso and Oum (2007) argue, because analysis based on aggregate data has dominated, this also dominates the mean values of meta-studies. They argue that income elasticity is overestimated when demographic effects are ignored and that estimations based on disaggregate data show many other aspects, other than price and income, that influence petrol consumption.

Hence this thesis concentrates on studies that use data at household level. One of the popular methods for studying petrol demand elasticities based on disaggregate data is some form of two-part model, where firstly the probability of owning a car has been estimated, followed by use of a regression model to analyse the positive amounts of spending. Such approaches are used, for example, by Kayser (2000), Asensio et al. (2002) and Sardianou (2008).

Kayser (2000) uses a Heckman selection model to analyse gasoline (petrol) demand and car choice in the USA. The results of the estimation show that households headed by a woman consume 30% less gasoline than households headed by a man and households with non-white heads consume 11.6% less gasoline than their white counterparts. Also, living around good public transportation tends to lower gasoline consumption. The level of education and the number of children in the household do not appear to significantly affect gasoline consumption. The short-run income elasticity found in the study is 0.48 and the price elasticity is –0.23. Kayser (2000) also finds that the interaction between price and income is significant.

Asensio et al. (2002) studied petrol expenditure in Spain, estimating firstly the probability of owning a car and secondly petrol expenditure according to the number of cars owned. Their results show that households living in the countryside have a higher probability of owning a car than households living in cities. Income elasticities are higher than unity for the households with lower income levels and lower than unity for richer households.

Sardianou (2008) uses the Heckman model to study car fuel consumption in Greece and finds that demographic, economic and attitude characteristics and the quality of public transport services explain the differences in car fuel demand. The mean income elasticity of the study is 0.52. The study also applies quantile regression to analyse whether the estimated coefficients are similar in different quantiles, finding that income elasticity is higher in the first decile of petrol consumption, but also in the highest petrol consumption group.

A widely used method of analysing elasticities of car fuel demand is AIDS (almost ideal demand system). While two-part models can capture socio-demographic aspects as well, the AIDS method concentrates only on own and cross-elasticities. Examples of such studies are those of Nicol (2003), Brännlund and Nordström (2004), Labandeira et al. (2005) and Barros and Prieto-Rodriguez (2008). Brännlund and Nordström (2004) analyse changes in consumer behaviour due to the carbon tax in Sweden and conclude, interestingly, that price elasticity is similar in different regions and income groups. In the study of Labandeira et al. (2005) based on Spanish household expenditure data, it is found that car fuel is a luxury and there is a significant relationship between car fuel expenditure and place of residence.

Barros and Prieto-Rodriguez (2008) analyse a revenue-neutral tax reform intended to increase demand for public transport services. They find that fuel and public transport services conform to the profile of luxuries, since their expenditure elasticity is greater than unity: for car fuel it is 1.25. Hence, car fuel is viewed as a luxury good in their study. They also find that demand for these goods is very sensitive to price changes: the own-price elasticity of fuel is –0.817, of private transport services is –1.853 and of public transport it is –1.003.

There is also a study that uses the instrument variable method for analysing the expenditure elasticity of different transportation goods, Aasness and Larsen (2003). In general, it is believed that for efficient and fair tax policy, a broad tax base is preferable, but Aasness and Larsen argue for a differentiated tax scheme in the case of environmental taxes: if a society wants to apply the vertical equity principle, then the products with high elasticity should be taxed more and the products with low elasticity should be taxed less. According to their estimates, one should be careful when taxing car fuel, because its Engel elasticity is quite low (0.7). Luxury goods are, for example, air flights (2.0), road tolls (2.0), taxi rides (1.74) and car purchases (1.6). Tax differentiation is also supported by Albrecht (2006), who argues that taxes should be differentiated according to their environmental impact. However, instead of imposing new environmental taxes, he is in favour of differentiating existing consumption taxes.

The above studies mostly reflect short-run effects. However, there is no clear distinction between short-run and long-run effects, as different studies use different definitions. As a general rule, anything shorter than one year is considered short term (Graham & Glaister 2004). In this overview, a study that has used data from less than five years is short term and from more than five years is long term.

A different methodological approach has been taken by Wadud et al. (2009) to estimate the price and income elasticity of gasoline demand in the USA based on data from 1984–2003, hence this is the longest time perspective among the studies reviewed in this thesis. The authors find that price elasticity follows a U-pattern: for the first income quintile it is high, but then it starts to decrease, being at minimum for the third quintile and increasing again after that. Wadud et al. (2009) show also that gasoline consumption does not depend on income changes in the lowest and highest income groups.

We see that the studies presented have been mostly for developed countries. The studies used in the meta-analysis of Goodwin et al (2004) are also mostly from developed countries: (the USA, Great Britain, Canada, France, Germany, Belgium and others). One of the few studies that has captured very different countries to analyse motor fuel prices and income elasticity is that of Dahl (2012). The author finds a pattern that lower-income countries are less price responsive than higher-income countries and explains this by two alternative hypotheses: in poorer countries, only rich people have personal vehicles and they may be less responsive to price changes. Alternatively, poorer countries tend to have higher capital costs and people keep their vehicles for longer. The price elasticities found in Dahl (2012) are rather low and the income elasticities rather high (mostly above unity). The high income elasticities can partly be explained by the nature of the data: aggregate country level. Hence, as stated before, the income elasticities might be overestimated. As for the price elasticity, Dahl (2012) finds the petrol price elasticity of –0.32 for Estonia and the same price elasticity for most of the EU new member states except Bulgaria and Romania, which have petrol price elasticity of –0.26. The income elasticity of Estonia is found to be 1.11.

Table 2.3.1 summarises the different studies presented above. We see that the methodologies for analysing energy demand elasticities are very different, from two-part models to several-equation demand systems. The country coverage is limited, focusing on developed countries. We also notice that even for a specific country, different researchers have obtained very different results; for example in the case of Spain, the estimated income elasticity ranges from 0.51 to 1.79. Hence, according to some studies motor fuel can be classified as a necessity (Asensio et al. 2002, Kayser 2000, Sardianou 2008), while according to other studies it is a luxury good (Barros & Pietro-Rodriguez 2008, Labandeira et al. 2005).

Some part of the differences might come from the time perspective; as also shown by meta-analyses, long-term elasticities are higher than short-term elasticities. But even then the differences are very large: for example, the Labandeira et al. (2005) price elasticity is –0.11, while the data used is long-term. Also, the long-term estimations of Wadud et al. (2009) are lower than expected. It seems reasonable to state that as a general rule, long-term elasticities are higher than short-term ones, but for the USA these are lower than for European countries. It seems also that the elasticities obtained via the AIDS methodology are higher than the ones obtained via two-part models, which is also in line with the conclusions of Basso and Oum (2007), who state that income elasticity is overestimated when demographic effects are ignored. Demographic factors that are found to be important in the studies presented above (specifically, Asensio et al. 2002, Kayser 2000 and Sardianou 2008) are: settlement type, age, gender, race, education, employment, number of children.

Table 2.3.1. Overview of income and price elasticities of motor fuel estimated by different authors

Author, year	Country	Methodology	Data used, short or long term	Income elasticity	Price elasticity
Goodwin (2004)		Meta-analysis	Long-term	1.08	–0.64
			Short-term	0.39	–0.25
Graham & Glaister (2004)		Meta-analysis	Long-term	0.93	–0.77
			Short-term	0.47	–0.25
Brons et al. (2008)		Meta-analysis	Long-term		–0.84
			Short-term		–0.34
Dahl (2012)	Estonia	Meta-analysis	Long-term	1.11	–0.32
Kayser (2000)	The USA	Heckman	Micro-data, short-term	0.48	–0.23
Asensio et al. (2002)	Spain	Ordered probit+OLS	Micro-data, short-term	0.51	
Nicol (2003)	The USA	AIDS	Micro-data, short-term	0.56	–0.03
	Canada	AIDS	Micro-data, short-term	0.44	–0.58
Aasness & Larsen (2003)	Norway	2 SLS	Micro-data, long-term	0.70	
Brännlund & Nordström (2004)	Sweden	AIDS	Micro and macro-data, long-term		–1.18
Labandeira et al. (2005)	Spain	AIDS	Micro-data, long-term	1.79	–0.11
Barros & Pietro-Rodriguez (2008)	Spain	AIDS	Micro-data, long-term	1.25	–0.82
Sardianou (2008)	Greece	Probit+OLS	Micro-data, short-term	0.52	
Tiezzi (2005)	Italy	AIDS	Micro-data, long-term		–1.28
Wadud et al. (2009)[4]	The USA	OLS, SUR	Micro-data, long-term	0.38	–0.20

Source: compiled by the author

The distributional effects of environmental taxes are related to the behavioural effects. People in higher income classes have more options for adapting to a specific policy, for example by moving to another location or investing in environmentally friendly equipment or a different heating system. Traditionally it is assumed that elasticities are the same for everyone (Kriström 2006), as the average elasticity is measured. However, there might be important differences

[4] If multiple elasticities are found in the studies, the ones of the middle income quintile are used.

according to income level or demographic characteristics. For example, Wadud et al. (2009) show that price elasticity is highest for the first and the last income quintiles, but lowest for the third income quintile. As for income elasticity, Asensio et al. (2002) find that it is higher for low-income households and lower for richer households. Wadud et al. (2009) show that gasoline consumption does not depend on income in the lowest or in the highest income groups.

As seen from this literature overview of the behavioural effects, the country coverage is limited to developed countries, as data is more readily available there and also the problems of transportation stand out more sharply. In order to address the equity issues of environmental taxation, the income elasticities of motor fuel consumption are assessed together with related socio-demographic factors. In order to assess the effectiveness of environmental taxation in terms of changed consumption, price elasticity is assessed. The next chapter provides the methodological framework for the empirical part of the thesis.

3. METHODOLOGICAL FRAMEWORK

3.1. Research questions and propositions

To fulfil the objective of the thesis, three main research questions are formed, restated in Figure 3.1.1. The distributional effects of environmental taxes are covered by the first two research questions, as they originate from two sources: firstly, the direct effects, which can be attributed to the consumption of goods that are taxed and for which the amounts paid by consumers can be calculated (for example, excise tax on petrol bought for driving); and secondly, the indirect effects, as different consumption goods contain an environmental tax share, but its amount is unknown to the consumer (for example, the excise tax share in food prices). In addition, distributional effects are related to behavioural effects, which also reflects the effectiveness of a tax, and hence the third research question is about households' responses to environmental taxes. Here, tax effectiveness means whether the applied tax has decreased the consumption of a taxed good, which should be the primary objective of an environmental tax. The linkage between tax effectiveness and distributional issues can be bidirectional: while changes in the consumption of a taxed good have an effect on income distribution, concurrently the income distribution and inequality might restrict households' responses (for example, low-income households cannot afford new energy-efficient cars).

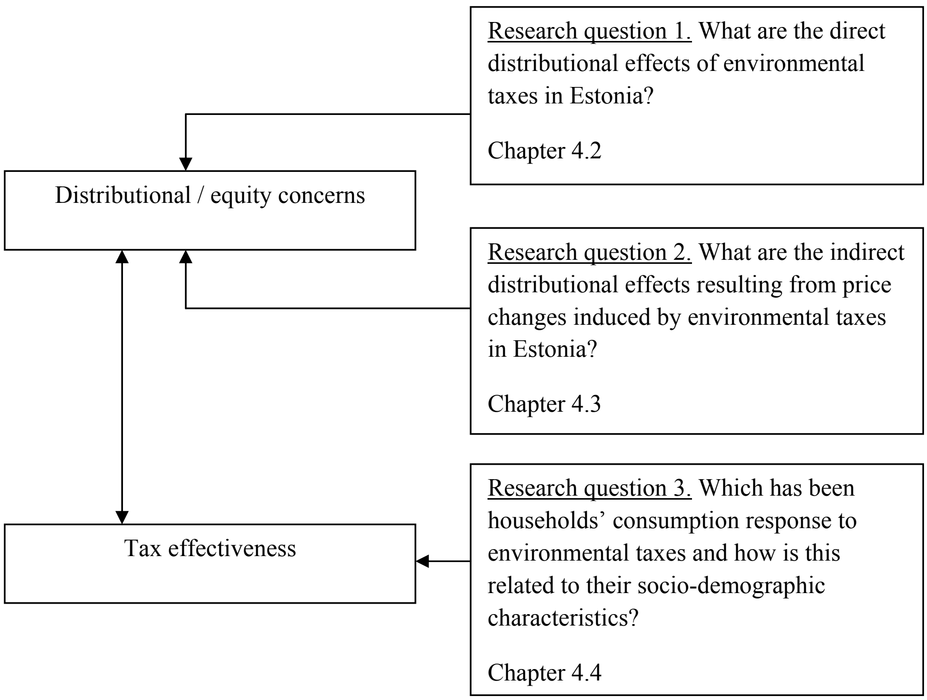

Figure 3.1.1. The research questions and structure of the empirical research of the thesis (author's figure)

Ekins et al. (2011) distinguish between the following distributional effects of ETR:

1) those due to the environmental taxes themselves;
2) those due to any tax reductions or revenue distributions associated with the ETR;
3) those that arise from the broader economic and environmental effects of the ETR, including price changes of goods and services and macro-economic effects such as effects on employment levels;
4) those due to exemptions and other specific provisions that have been made in the tax design for various purposes (e.g. competitiveness, social concerns or environmental considerations);
5) the distribution of the environmental improvements brought about by the ETR.

The distributional effects analysed in this thesis are connected to the first and third abovementioned aspects. As there are no exemptions in the Estonian environmental tax system as regards households, the fourth aspect is not relevant. The revenue distribution aspect and the distribution of the environmental improvements are not covered by this thesis. Although these are important as well, the thesis is focused on the pure effects of environmental taxes, leaving out other taxes. For environmental improvement and its distribution, there are no good data sources to use in Estonia.

The addition of this thesis to the abovementioned list is its focus on the interlinkages between the distributional and behavioural effects, as both are important for environmental taxes. As discussed in the theoretical part of the thesis, environmental taxation is motivated by internalising the external costs of certain activities. At the same time, some environmental taxes have proven to be good sources for state budgets, in accordance with the Ramsey taxation principle: the goods whose demand has low elasticity should have higher tax rates. Hence, there seems to be a trade-off: if goods with negative environmental effects have low price elasticity, they are good sources for state budgets, but then the price increase does not help to address the environmental problem involved. At the same time, the response to price changes might be different according to income levels or other demographic aspects, and hence the behavioural and distributional effects are related.

Below, the research questions and thesis propositions are discussed in detail.

Research question 1. What are the direct distributional effects of environmental taxes in Estonia?

Proposition 1. Environmental taxes in general are regressive and hence increase the income inequality in a country.

The suspected negative distributional effect on poorer populations has been one of the barriers to environmental tax implementation. A regressive distributional effect resulting from the direct consumption of environmentally taxed goods has been demonstrated, for example, by Jacobsen et al. (2003), Bork (2006) and Dresner and Ekins (2006). However, Jacobsen et al. (2003) find that environmental taxes are less regressive than VAT and alcohol and tobacco excises. Different authors show that with the help of compensation schemes, the regressive effect can be reversed. An overview of the empirical research regarding direct distributional effects of environmental taxes is provided in section 2.1.

Some authors suggest that taking only some taxes for analysis does not give an adequate picture, as the broad impact of different taxes, subsidies and other measures is important (see, for example, Creedy 1998 for discussion). However, as this thesis is justified by the increasing role of environmental taxes in state policy, then concentration on a specific tax type provides useful information about the effects of the policy, the vulnerable groups involved and ways to address the undesirable effects.

<u>Proposition 2.</u> *The adverse effect of environmental taxes appears specifically for certain taxes (on heating fuels) and certain household types (retired people, households with children, households living in rural areas).*

This proposition comes from empirical studies done elsewhere. The regressive effects of fuels used for heating and/or electricity have been demonstrated, for example, in Barker and Köhler (1998), Jacobsen et al. (2003) Bork (2006), Dresner and Ekins (2006) and Callan et al. (2009). Only Jacobsen et al. (2003) and Bork (2006) have analysed the effects of existing environmental taxes and have shown that while the taxes on heating fuels are regressive, there is also a progressive component of environmental taxation, which originates from taxing vehicles. The distributional pattern of motor fuels has been more mixed; for example, Jacobsen et al (2003) and Tiezzi (2005) find it progressive, Bork (2006) finds it mostly falling on middle-income groups, while Sterner et al. (2012) conclude that it is more likely neutral or slightly regressive in developed countries and progressive in developing countries. The Estonian environmental tax system related to households is different in the way that there is no transportation tax implemented; only the fuels used for transport are taxed in a form of fuel excise. During past years, the role of taxes on heating fuels has increased and excise on electricity has been implemented in Estonia. Hence it can be expected that the general pattern of environmental taxes in Estonia is regressive.

Based on the literature overview, the household types that are more vulnerable to environmental tax increases are households living in rural areas, households with children, and self-employed and retired people. An overview of the relevant studies and results is provided in section 2.1.

<u>Research question 2.</u> What are the indirect distributional effects resulting from price changes induced by environmental taxes in Estonia?

Proposition 3. Households in low-income groups consume more goods for which the environmental tax load is higher.

The studies done elsewhere on indirect distributional effects have found that the sectors of energy production and transportation bear the greatest load of the carbon tax, followed by other energy-intensive sectors, for example, food and beverages (Cornwell & Creedy 1996), water and package holidays (Wier et al. 2005) and lime manufacturing (Grainger & Colstad 2010). As low-income households spend a higher share of their income on goods and services that are believed to be energy-intensive (home heating and electricity), this leads to adverse indirect distributional effects on poorer households. A regressive indirect distributional effect has been detected for all empirical studies presented in section 2.2 except for that of Labandeira and Labeaga (1999), who find that the effect is proportional. Hence, it is assumed that in Estonia the indirect distributional effect of environmental taxes is also regressive.

Proposition 4. The direct and indirect distributional effects of environmental taxes are increasing income inequality.

If the heating fuels tax is prevailing in environmental taxes paid by households and if the share of energy-intensive consumption goods is higher among poorer populations, then the result is increasing inequality in a society. Income inequality as a general trend in Estonia has increased during the past two decades, although there have been periods of ups and downs. The proposition is motivated by the assumption that the rise in environmental taxes has contributed to increasing inequality in a country.

As for the proportion between direct and indirect environmental tax loads, from the reviewed studies only Wier et al. (2005) discuss this: the indirect carbon tax load forms one-third of the direct one. Hence we expect that also in case of Estonia the indirect environmental tax load is lower than the direct tax load.

Research question 3. Which has been households' consumption response to environmental taxes and how is this related to their socio-demographic characteristics?

Proposition 5. On average, households' response to environmental taxes (specifically motor fuel excise) has been low and consumption of a taxed good has not decreased significantly.

This research question and relevant propositions are limited to changes in motor fuel consumption, as the price increase induced by environmental taxes needs to be in place for some years in order to analyse the behavioural responses, and this is the case for motor fuel excise. For other fuels and electricity, the time series is too short (imposed since 2008) or the data is limited (for example, only a small proportion of households is using heavy fuel oil for heating).

From the literature overview on the price and income elasticities of motor fuel presented in the previous chapter, it can be seen that the assessed elasticities are very varied, from low responses to price increases to very high responses. According to income elasticity, some studies classify motor fuel as a necessity and some as a luxury good (a detailed overview is presented in section 2.3). The income level in Estonia has increased in the 2000s but so has the price of motor fuels: the motor fuel excise has increased by 100% and the motor fuel price has increased even more than just through the excise tax (see annex 15). The consumption of motor fuel in Estonia has increased during the growth period of the 2000s (by 20% for petrol and by 100% for diesel) and decreased in the years of crisis for petrol, which still has not reached its pre-crisis consumption level, while for diesel the consumption level is higher than ever before (Eurostat 2013b). Hence it is expected that the consumption of motor fuel has rather low price elasticity.

Proposition 6. The reaction to price changes differs by specific socio-demographic characteristics.

As discussed earlier, the distributional effects are related to the behavioural effects. People in higher income classes have more options for adapting to a specific policy, for example by moving to another location or investing in environmentally friendly equipment. The elasticity figure is usually an average response, but there might be important differences according to income level or demographic characteristics, which has been also shown by different authors. Wadud et al. (2009) show that the price elasticity of gasoline demand is highest for the first and the last income quintiles, but lowest for the third income quintile. As for income elasticity, Asensio et al. (2002) find that it is higher for low-income households and lower for richer households. Wadud et al. (2009) also show that gasoline consumption does not depend on income in the lowest and highest income groups.

Demographic factors that are found to be important in the studies presented in section 2.3 are: settlement type, number of children, age, gender, race, education and employment of the household head. The corresponding socio-demographic characteristics used from HBS data in this thesis are provided in annex 13, but in sum the demographic variables tested in the thesis are the same listed above except for race, which is not relevant in the Estonian context, and in addition to the number of children, household size is also tested.

3.2. The data and methods

3.2.1. The data and the main steps of the analysis

As most of the studies on environmental taxes' distributional effects, which include direct as well as indirect effects, are about the carbon tax and are specifically interested in the changes in economic structure, the analysis method normally used is the general equilibrium model. This, however, does not allow for very specific analysis at household level, but typically some household classes or groups are selected to illustrate the impact. However, it has been shown that variation within a specific household category is important and can significantly affect the results of the analysis (see Yusuf & Resosudarmo 2007 for an overview). This thesis is specifically interested in the effects on households and differentiating these according to socio-demographic characteristics, and thus micro-data at the household level has been used. It has been argued that the unit of distributional analysis should be the individual (Atkinson & Stiglitz 1980). In empirical terms, only data at the household level is available. The household data can be transformed into individual data with the help of equivalence scales, which is also a common practice in assessing the distributional effects of environmental taxes. Equivalence scales take into account the economies of scale in consumption in a household, for example, in the case of durables. The equivalence scale used most often in this research area, and also in this thesis, is the one from the OECD (1; 0.5; 0.3), which means that the first household member is assigned a value of one, additional adult household members a value of 0.5 and children a value of 0.3.

The data used in this thesis originates from the following sources:

- Household Budget Surveys (HBS) 2000–2007 and 2011 from Statistics Estonia (altogether 50,320 observations);
- fuel excise rates 2000–2011 from Ministry of Finance (2013);
- fuel price data from the following sources:
  - electricity and gas for 2000–2011: Eurostat (2013b);
  - diesel and petrol for 2000–2004: Statoil (2008), 2005–2011: European Commission (2013b);
  - light fuel oil: Statistics Estonia (2013);
- average prices for energy in enterprises of Estonia from Statistics Estonia (2013);
- production of heat energy by energy source in Estonia, 2000–2011: Statistics Estonia (2013);
- input–output table of 2005 as the most recent one from Statistics Estonia (2012c), sectors detailed by NACE 2 digits and for some sectors 3 digits;
- fuel consumption data by economic sector (sectors detailed by NACE 2 digits) from 2000–2007 and 2010 from Statistics Estonia (2013);
- electricity consumption in enterprises in 2007 (latest available) (Eurostat 2013b);

- amounts of environmental charges differentiated by charge type (resource charge, water extraction charge, air pollution, water pollution and waste disposal charge) paid by economic sectors (sectors detailed by NACE 4 digits) from Ministry of the Environment (2011);
- correspondence tables of COICOP (classification of individual consumption according to purpose) and CPA (classification of products by activity) from Statistics Estonia.

The main data source is HBS, collected by Statistics Estonia and including monthly after-tax income and household consumption expenditure. This survey was carried out annually up to 2007 and again since 2010 with slightly altered methodology. One of the reasons for the methodology change was the decreasing response rates, part of which was attributed to the complicated and sizeable materials. Hence, instead of the four different materials used in previous years (inquiry about household characteristics and background data, post-inquiry, food expenditure diary, and income and expenditure diary), only two remain since 2010 (one inquiry and one diary) and the diary is filled in for two weeks instead of one month (Statistics Estonia 2012a). As the fuel and electricity excises are related to quantities, but the HBS includes only expenditures in monetary terms, the average prices of commodities are used to calculate quantities.

This thesis uses expenditure as a proxy for lifetime income, regarding it as a better indicator for distributional analysis than income, as also discussed before. To allow comparison with other researchers' results, at some points the tax load as a ratio to income is also presented. As income data is not available from the HBS after 2010, it is imputed by the author for recent years based on the proportion of income to expenditure by income decile as an average for 2000–2007 (annex 11).

The environmentally related taxes and charges covered by this thesis are electricity and fuel excises and environmental charges (see Figure II in the Introduction). Environmental charges are not taxes in the classical sense, but duties imposed specifically on enterprises for environmental pollution or resource use, and the revenues are earmarked for environmental purposes. Hence their effect on households cannot be directly assessed, but the indirect tax load can be calculated, which comes from households' consumption of goods produced by enterprises that pay the charges (see the next section for a description of the thesis methodology).

Motor fuel excise is also included under environmentally related taxes and charges in this thesis, although its environmental effect can be questionable: 75% of motor fuel excise is spent on road building and maintenance. However, motor fuel excise is a typical energy tax and energy use is closely related to emissions. It is also shown by Sterner (2007) that the effect of petrol taxes on carbon emissions makes it a significant instrument of climate policy.

Although district heating as such is not taxed by excise, if a full tax shifting is assumed, a tax share for district heating consumers can also be calculated.

Regarding the sectoral data used for indirect distributional effects, an obstacle is the different classifications – in the case of fuel consumption NACE rev. 1.1 are used, but for the input–output tables NACE rev. 2 is used. Hence correspondence tables are used to give results in the form of NACE rev. 2. The resulting list of sectors is provided in annex 7.

The assessment of distributional and behavioural effects has been done according to the following steps:

1. calculation of the direct distributional effect on households originating from fuel and electricity excise;
2. calculation of the indirect distributional effect on households originating from fuel excise, electricity excise, pollution and resource charges;
3. calculation of the behavioural effect on households of the petrol excise based on income and price elasticity;
4. synthesis of the different effects.

In accordance with microeconometric terminology about taxation effects, distributional effects are referred to as 'static effects' and behavioural effects as 'dynamic effects'. This means that although the distributional pattern is observed over time, it still denotes a certain point in time reflecting how much environmental tax a household pays. We cannot observe, based on this kind of data, whether a household actually changes its consumption behaviour as a response to a tax or not. The same holds for the indirect distributional effect, where the tax load for a household is calculated via the input–output table, which again reflects a certain point in time. It might also be that enterprises change their behaviour due to tax, but this cannot be observed with such a static approach. The change in consumption is called a behavioural effect and measured by the demand elasticity of Estonian households in this thesis. Although the original idea of the author was to discuss the behavioural effects of all the environmental taxes applied in Estonia, due to substance and data considerations, the behavioural part is limited to petrol excise. Firstly, switching heating fuel or additional investments into the system is costly and takes time. At the same time, rapid change in the taxing of domestic heating fuels and electricity has taken place since 2008, which leaves too short a time period to consider changes in consumption. Hence, the analysis is limited to motor fuel excise, consisting of petrol and diesel excise. However, the number of households reporting diesel consumption is very limited, for which reason the author has chosen to limit the behavioural analysis to petrol consumption.

The analysed taxes, used data and methods are presented in Figure 3.2.1. The methods and models used in the thesis are described in more detail in the next section.

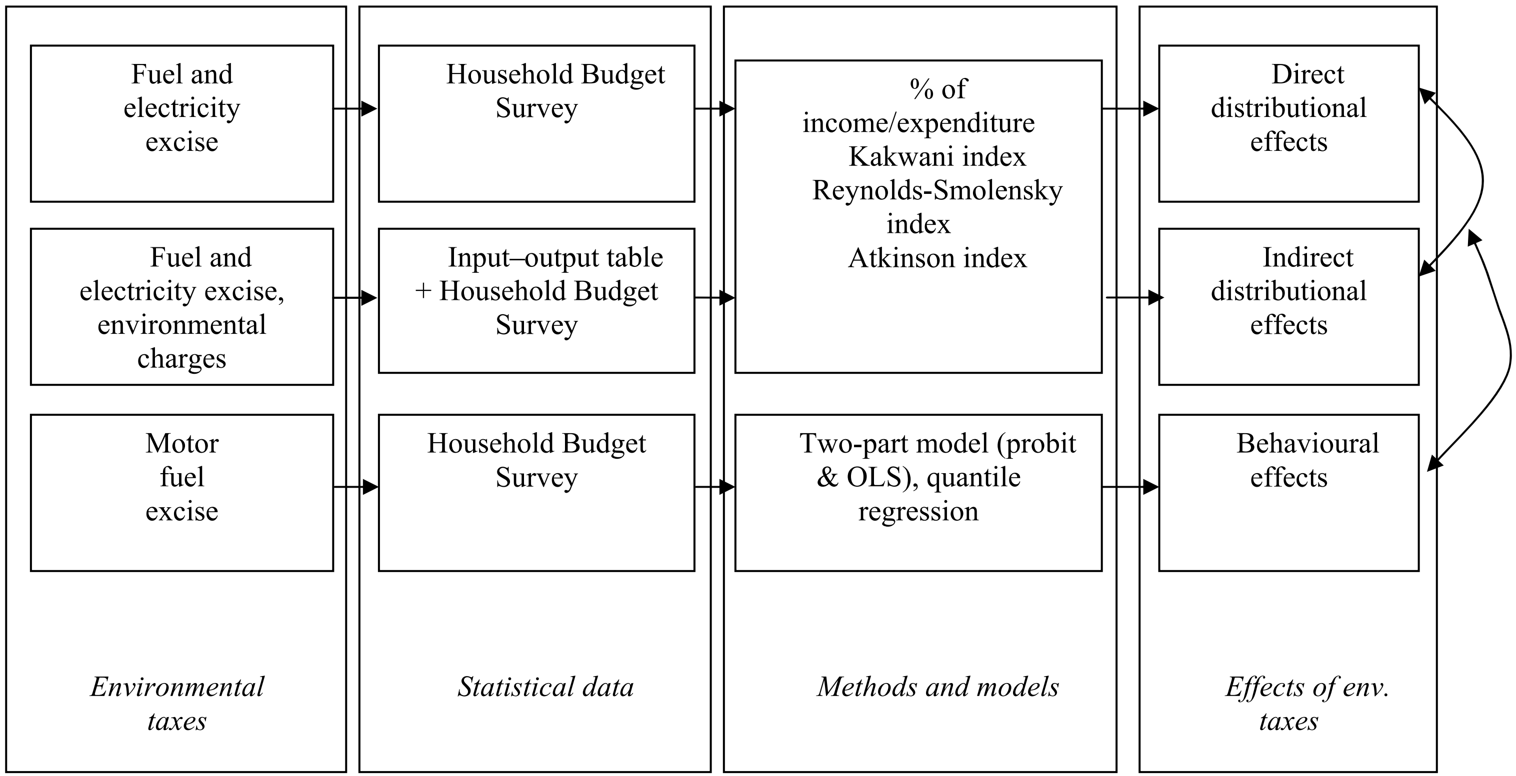

Figure 3.2.1. Methodology of the thesis (compiled by the author)

3.2.2. The methods and models applied

To assess the distributional effects of environmental taxes, the microsimulation method is used in this thesis. This method simulates policy effects on a sample of economic agents. The idea of microsimulation originates from Orcutt (1957), who criticised the economic models in use at that time for high levels of aggregation and limited predictive usefulness. He proposed a new type of model on a disaggregated level, for example on individual, household or firm levels. However, it was only in the 1980s that the method became widely used, mostly due to the availability of detailed datasets and increases in computing power (Bourguignon & Spadaro 2006). Microsimulation models can be divided into two categories: static and dynamic models. The difference is that dynamic models include the behavioural responses of individuals or households. Static microsimulation models calculate tax or benefit payments to identify winners or losers from any reform (*ibid.*). However, the microsimulation method has not been widely used in assessing the effects of environmental taxes.

This thesis applies the microsimulation method using the HBS data as described in the previous section to discover who bears the highest load of environmental taxes in Estonia. The distributional measures used in the thesis are descriptive ones (the Kakwani index, Reynolds-Smolensky index) and normative ones (the Atkinson index). Empirical studies of the direct distributional effect of environmental taxes also apply some descriptive measures like the Gini coefficient, Suites index and Kakwani index in addition to demonstrating tax share as a proportion of income or expenditure (an overview of these studies is provided in chapter 2). The Kakwani and Suites indices are similar in idea, both relying on the concept of the Gini index. According to Sterner (2012), the Kakwani and Suites indices are the two most common indices used to measure the progressivity of taxes. In this thesis, the Kakwani and Reynolds-Smolensky indices are used to find out whether different indices demonstrate similar distributional implications. In addition, the normative Atkinson measure is applied in the thesis to find out the sensitivity of the distributional effects evaluated by the descriptive indices to different levels of inequality aversion in a society.

The Kakwani index is based on the Gini index, which in turn is based on the Lorenz curve (Figure 3.2.2). The Lorenz curve ranks people according to income and plots the percentage of income enjoyed by specific proportions of the population.

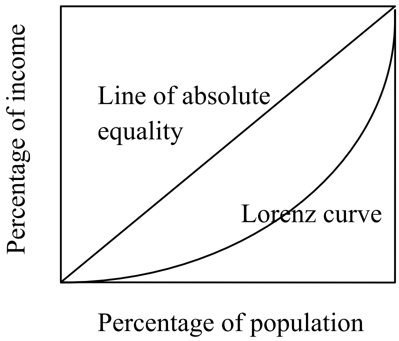

Figure 3.2.2. The Lorenz curve (Sen & Foster 2003)

The Gini coefficient is the ratio of the difference between the line of absolute equality and the Lorenz curve to the whole area under the line of absolute equality (see Figure 3.2.2). Hence the larger the coefficient, the greater the inequality in a society. The Gini index is the percentage equivalent of the Gini coefficient. Computationally, the Gini coefficient is calculated as the arithmetic average of the absolute values of the differences between all pairs of income values (Sen & Foster 2003):

$$(6)\ G = \left(1/2n^2\mu\right)\sum_{i=1}^{n}\sum_{j=1}^{n}\left|y_i - y_j\right|$$

Alternatively, the Gini index can be computed by using the covariance between the income values and their ranks (Xu 2003):

$$(7)\ G = \frac{2\operatorname{cov}(y_i, i)}{n\mu_y}$$

The Kakwani index (Kakwani 1977) is calculated by comparing the Lorenz curve for income and the concentration curve for the taxes, i.e.

$$(8)\ P = \left(C - G\right)$$

where:
C – concentration index of taxes;
G – the Gini index of the before-tax income.

The concentration index of taxes is also based on the Lorenz curve, but instead of income the amount of taxes paid is used on the vertical axis. A positive value of P implies a progressive tax system and a negative value of P implies a regressive tax system.

The Reynolds-Smolensky index measures the difference between the pre-tax Gini index (G_x) and the post-tax Gini index (G_y) (Lambert 1993):

(9) $L=G_x-G_y$

If the Reynolds-Smolensky index is positive, this means that the inequality after taxation has decreased, as the post-tax Gini index is smaller (closer to equality), and a negative index value demonstrates an increase in inequality.

A normative measure applied in this thesis is the Atkinson index, which was developed in 1970 and is calculated based on the following formula (Atkinson 1970):

$$(10)\ I = 1 - \left[\sum_i \left(\frac{y_i}{\mu} \right)^{1-\varepsilon} f(y_i) \right]^{\frac{1}{1-\varepsilon}}$$

where:
y_i – individual income;
μ – mean income;
ε – degree of inequality aversion.
An important part of the formula is the degree of inequality aversion, or the relative sensitivity to transfers at different income levels. As ε rises, more weight is attached to transfers at the lower end of the distribution and less weight to transfers at the top. At one extreme, where ε=0, this means the linear utility function is ranking distributions solely according to total income. The author of the thesis is aware of the limitations of the Atkinson index noted, for example, by Sen and Foster (2003), but applies the concept to find out how sensitive the distributional effect is to different levels of inequality aversion.

As explained earlier, a distinction is made between direct and indirect environmental tax payments, and the total environmental tax burden for households is the sum of both. For fuel excise, both can be calculated: direct and indirect tax burden. The direct effect comes from fuel consumption in households resulting from consumption of motor fuel by cars and fuels used for heating homes. But environmental charges (pollution and resource fees) can be considered only under the indirect tax burden for households, as the tax is applied to enterprises based on some polluting substance which cannot be directly associated with any unit of consumer product. The differentiation of environmental charges by charge types and sectors has only been done since 2009, as the earlier administration system was not able to provide it. The methodology applied in this thesis to calculate the indirect environmental tax load is similar to the one used by Wier et al. (2005) and Kerkhof (2008). This is done with the help of an input–output table, which allows consideration of the intersectoral linkages: for example, a sector uses some fuel for production, but also buys some parts from other sectors, which also consume fuel for production. The input–output table enables us to consider this indirect consumption as well. Hence, the indirect environmental tax load is calculated as follows:

$$(11)\ TAX^{indirect} = T(I - A)^{-1}$$

where:
$TAX^{indirect}$ – total indirect environmental tax payments by sectors;
T – 1x18 vector with environmental charges and taxes per unit of output;
$(I\text{-}A)^{-1}$ – 18x18 Leontief inverse matrix, A being coefficients based on intersectoral commodity flows.

Although the input–output table obtained from Statistics Estonia includes 80 sectors, due to the aggregation of tax data the input–output table has to be aggregated as well. The aggregation of tax data in turn originates from a high level of aggregation of fuel use data; it is not possible to obtain more detailed data about the sectors due to the small sample size.

As in this thesis actual taxes are analysed, not hypothetical ones, we cannot speak of price increases but rather a share of tax in the price of goods. Hence the indirect tax amount paid by a relevant sector reflects its own payment of environmental taxes, but also payments due to buying goods or services from other sectors which also pay the taxes. The taxes analysed in this section are fuel excises (calculated from data of fuel consumption by fuel and sector) and resource and pollution charges. The indirect price effects are transferred to consumer goods with COICOP–CPA coefficients obtained from Statistics Estonia. Subsequently, the indirect price effect on consumers is evaluated with the help of HBS data. As this data source does not enable us to assess the share of imported goods consumed by different consumer groups, it is assumed that the share of imported goods does not vary between socio-economic groups or products.

This indirect price effect is a very rough estimate and a very static one. First of all, different commodities belong to one commodity group and due to aggregation problems, this is amplified. For example, food contains bread, meat, milk, fruit and other products for which the importance of fuel use and pollution charges might differ significantly and hence the result is an average estimate. Also, it is static in the sense that fuel use and environmental charge payments are expected to be stable, but in reality enterprises' behaviour might change: they respond to taxes, change the fuel used or invest in better technologies, reducing fuel or resource use or pollution. However, if the indirect price effect is different across sectors, this might lead to different distributional effects as well.

Hence the direct and indirect distributional effects for households derived from environmental taxation are calculated in this thesis. Although the tax share can be followed over time, the effect is considered a static one, because we can observe only the result of taxation: how much environmental tax a household is paying. However, we cannot say, based on this information, to what degree households have changed their behaviour in terms of decreasing the consumption of taxed goods or investing in new technologies, or whether this response differs by socio-economic group. It is known from the literature that omitting demand responses makes an environmental tax appear more regressive, as the demand in higher income groups is substantially less elastic (West & Williams 2002). However, it is not an easy task to research, applying entirely different methods than distributional analysis, and researchers usually focus on one of the issues, not both. This thesis is aimed at capturing both distributional and behavioural effects. As specified before, the behavioural effect analysis is limited to households' petrol consumption.

As less than half of the respondents to the HBS are questioned in two sequent years and some fill in only one part of the survey (household picture, diary book, income data), and as less than half of the these who fill in all parts claim to expend some money on petrol, the panel data method is not used in this thesis, as the panel is very small. The options for analysing such data are the Tobit model or the two-part model, of which the first part is the Probit model and the second part is truncated regression for positive values. The Tobit model assumes the same sign for the two effects: that the household spends on petrol and the same sign for the amount of expenditure. This might not be the case in reality, as some variables might have two-way effects on the dependent variable; in order to address this, a two-part model is applied in this thesis, containing the Probit model and the OLS/truncated regression model.

The Probit model is a model of binary outcomes, where the outcome can take one of two values (Cameron & Trivedi 2009):

$$(12)\quad y = \begin{cases} 1 & \textit{with probability } p \\ 0 & \textit{with probability } 1-p \end{cases}$$

In this thesis, we are interested in whether a household spends at all on petrol or not and we believe that there is a latent variable involved which determines whether a household has a car or not. Hence, we observe the outcome variable only when the latent variable is larger than or equal to zero. The probability of an event depends on some characteristics expressed by x_i (*ibid.*):

$$(13)\quad p_i = \Pr\left[y_i = 1 | x\right] = \Phi(x_i \beta)$$

The variables x_i used in the Probit model in this thesis are the number of adults, number of children, gender of the household head, age of the household head, education level of the household head, type of residence, household social type and expenditure of the household.

The marginal effects of the Probit model can be interpreted as how much the dependent variable will change if the independent variable changes by one unit:

$$(14)\ \frac{\partial p_i}{\partial x_{ij}} = \phi(x_i\beta)\beta_j$$

In order to assess the factors affecting the amount of petrol consumed in a household (taking into account only those households in which petrol consumption is higher than zero), the OLS for positive amounts is assessed. The setup of the model is thus the following:

$$(15)\ \ln(y_i) = \beta_0 + \beta_1 \ln(x_{1i}) + \beta_2 x_{2i} + \beta x_i^{'} + \varepsilon_i,$$

where:
y_i – quantity of petrol consumed in a household;
x_1 – household's total expenditure;
x_2 – petrol price;
$x_i^{'}$ – different demographic characteristics of a household (number of adults, number of children, gender of household head, age of household head, education level of household head, type of residence, household social type).

As the emphasis of the thesis is to consider as much heterogeneity of the data as possible, an additional method applied is quantile regression. Standard methods express the average relationship between the dependent variable and independent variables based on the conditional mean, but quantile regression shows the relationship between those at different points in the conditional distribution of the dependent variable (Cameron & Trivedi 2009). The quantile means that every percentile can be used in the analysis. While OLS minimises the sum of the residuals, in quantile regression the sum of asymmetrically weighted absolute residuals is minimised (Koenker & Hallock 2001). The benefit of quantile regression is that while OLS estimates apply to the conditional mean, quantile regression estimates models for conditional quantile functions. The sample qth quantile μ_q can be expressed as the solution to the optimisation problem of minimising with respect to β (Cameron & Trivedi 2009):

$$(16)\ \sum_{i:y_i \geq \beta}^{N} q|y_i - \beta| + \sum_{i:y_i < \beta}^{N} (1-q)|y_i - \beta|$$

Quantile regression has been conducted only for positive values of petrol consumption. The data analysis software used in the thesis is Stata (version 10 together with a Distributive Analysis Stata Package), applied for direct distributional analysis and behavioural analysis, and MS Excel, applied for input–output calculations.

4. EMPIRICAL RESEARCH

4.1. Environmental taxes of Estonia in the European context

The form and role of environmental taxes have altered during past decades and to a lesser extent this trend is also valid for Estonia. The lesser extent is related to the fact that the history of environmental taxes in Estonia is not as long as in countries with advanced environmental protection. Estonian environmentally related taxes and charges can be divided into two broad categories: environmental taxes and environmental charges (see also Figure III in the Introduction). The difference is that although environmental charges go into state budgets, their revenues are earmarked for environmental objectives and these are redistributed as project funds via the Environmental Investment Centre. Environmental charges are composed of pollution charges and resource charges, and these can be considered classical examples of economic instruments used in environmental policy, as the amounts of polluting substances or extracted resources are directly monitored and taxed. However, for state budgets the most important environmental tax is the fuel excise, which is easy to administer and whose rate increase can be justified by the requirements of the EU.[5] The latter aspect will be elaborated on below. As both of the mentioned taxes and charges are still related to the environment, the term ‘environmental taxes’ is used in this thesis to refer to all environmentally related taxes and charges of the country.

The fuel excise contributes about 80% of environmental tax revenues in Estonia (Figure 4.1.1; see also annex 2). In total, €454 million of revenue was collected from environmental taxes in Estonia in 2011, of which €361 million came from the fuel excise. Estonia is an exceptional country in the EU in the sense that there is no tax based on car ownership: a motor vehicle tax was applied until 2002 and then replaced by a heavy goods vehicle tax. However, the tax revenues are not comparable between the two taxes: in 2011, the heavy goods vehicle tax contributed only €3.7 million to the state budget (1% of environmental tax revenue), while the motor vehicle excise in 2002 was €11.2 million.

[5] The requirements are set by Council Directive 2003/96/EC restructuring the European Community framework for the taxation of energy products and electricity. As Estonian legislation uses the term ’fuel excise’ this term is also used in this thesis.

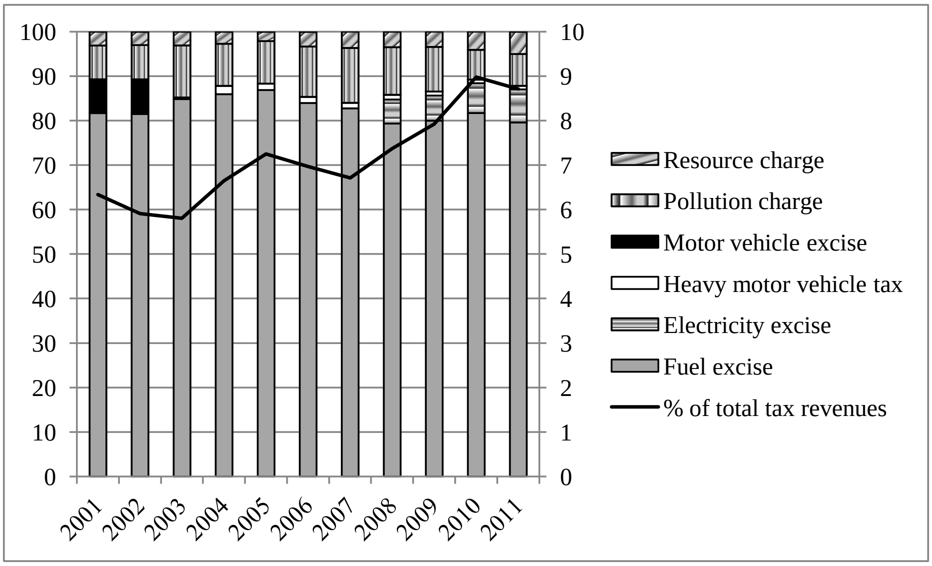

Figure 4.1.1. Environmental tax structure as a percentage of total tax revenues 2001–2011 (author's compilation based on Statistics Estonia (2013) and Ministry of the Environment (2011))

The share of pollution and resource charges in environmental tax revenues has been quite stable over the period, the pollution charge giving about 10% of environmental tax revenues, the resource charge about 3%. In recent years, the sums coming from the resource charge has increased; in 2009 €13 million, in 2011 €22 million). The sums from the pollution charge have decreased: in 2009 €39 million, in 2011 €32 million. In 2008 an electricity excise was applied, from which about €30 million of tax revenue is collected annually.

The general share of environmental taxes in total tax revenues has risen over the past decade: in 2001 it was about 6.3%, in 2011 close to 9%. At least partly this has been caused by an approach taken by the Estonian government to shift the tax burden from income taxation to the taxation of consumption, use of natural resources and environmental pollution (State Budget Strategy 2013–2016). At the same time, it is a trend similar to those in other new member states, which have been obliged to raise the fuel excise to the minimum EU level and hence the revenues of those taxes are also increasing. The specific countries where environmental tax revenue has witnessed similar trends as in Estonia are Latvia, Lithuania and Poland, where it has increased from 1.5% of GDP in 1995 to 2.5% of GDP in 2011 (see also Figure II in the Introduction). For EU15 and new member states located in Central Europe, the trend has been decreasing: from about 3% of GDP in 1995 to about 2.5% in 2011.

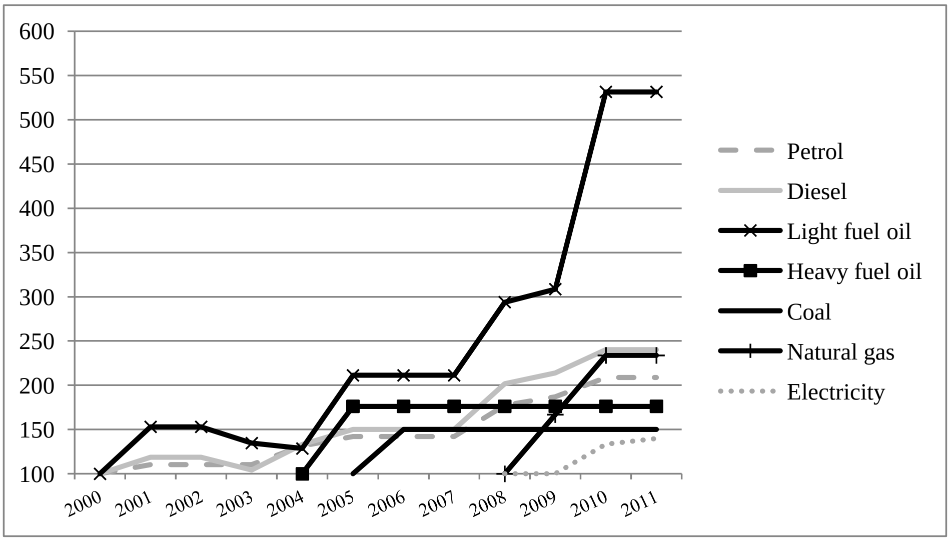

Figure 4.1.2. Index of energy tax rates in Estonia, annual averages, 2000=100, 2000–2011[6] (author's compilation)

Comparing Estonia's fuel and electricity excise rate level of 2011 to the initial level set, the biggest increase has taken place for light fuel oil: more than five times over 2000–2011 (Figure 4.1.2, the data of monetary value is in annex 1). For petrol and diesel, the increase in excise rate in 2000–2011 has been about two times. Also we can note that a large rise in excise rate took place in 2008–2009 and rates have been stable since 2010. In 2008–2009 new excises on natural gas and electricity were applied.

Setting the Estonian environmental taxes in the context of the European Union, we are reminded that the environmental policy of the EU is based on certain targets, for example regarding air and water quality, while the instruments to achieve these targets are chosen by member states, not prescribed by the EU. An important exception is the excise taxes on energy products, for which there are minimum EU-wide rates applied. The minimum excise rates are supposed to contribute to improve the functioning of internal markets as well as integrating environmental concerns into energy taxation. However, the current tax rates do not reflect the energy content or CO_2 emissions of the energy products, leading to inefficient energy use, and hence the European Commission has proposed changing the taxation of energy products to consider these two aspects (European Commission 2011). However, the proposal is still being discussed and hence the empirical analysis of this thesis has been done based on the existing excise rates for energy products.

[6] For fuels which have been taxed since later than 2000, the year of first applying the excise has been taken as the base value.

To give a perception of the tax rates in the context of other EU countries, the Estonian fuel excise rates are compared to the minimum rate required by the EU, but also to the lowest, average and highest rates applied in member states (Figure 4.1.3, more detailed data in annex 3). Estonia has imposed the minimum EU rate for all fuels, but it is lower than the EU average rate. As can be seen from the figure, for motor fuels (petrol and diesel) the excise rate is less dispersed among EU countries than for heating fuels and electricity. In the case of petrol, the Estonian excise rate is 58% of the highest EU rate (which in the case of petrol applies in the Netherlands); for heavy fuel oil the Estonian rate is only 3% of the highest (Sweden). For electricity, the Estonian excise rate forms 4% of the highest rate (Denmark). There are six countries distinguished for their high excise rates for most fuels: the Netherlands, Denmark, Sweden, Germany, Finland and the United Kingdom.

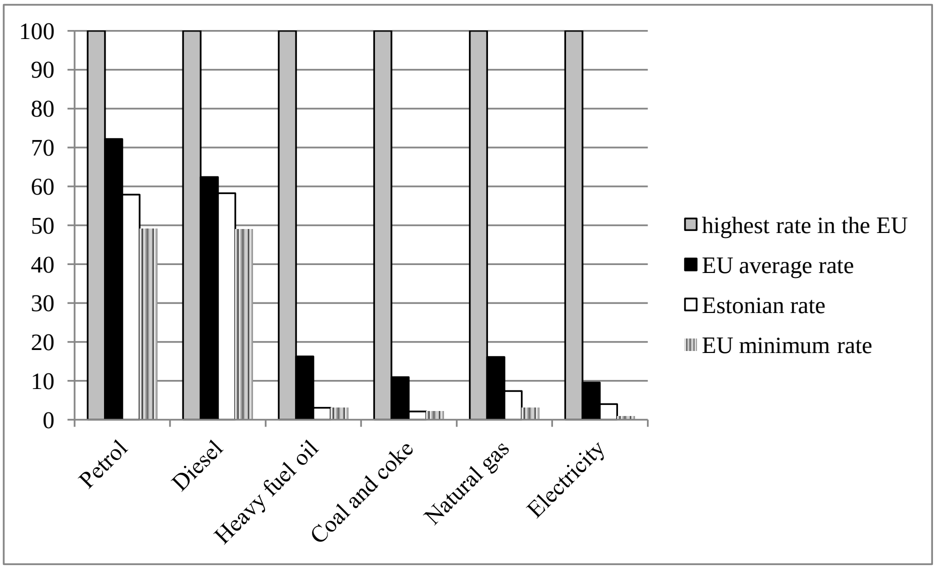

Figure 4.1.3. Excise rates of energy products and electricity in EU member states as of 1.07.2012: EU highest, average and lowest rates and the Estonian rate, EU highest rate=100 (author's compilation based on European Commission (2013a))

Most of the named countries which have excise rates of energy products much higher than required by the EU are also the countries which are known for implementing ecological tax reform: Sweden, Denmark, the Netherlands, Finland, Germany and the UK (Ekins 2012). Surprisingly, these countries are not necessarily these that collect the highest shares of tax revenue from environmental taxes (see Figure 4.1.4). The highest shares of total taxation were in 2011 collected by Bulgaria (10.6%), the Netherlands (10.2%), Malta (9.5%) and Slovenia (9.3%). Estonia collected 8.6% of total taxation from environmental taxes in 2011. Estonia is distinguished from other countries by the fact that transport taxes are almost non-existent. As said before, most Estonian environmental taxes are collected from the fuel excise, which falls under the category of energy tax (about 87% of environmental tax revenue). At the same time, the share of resource/pollution taxes is quite large in Estonia compared to other European countries: 0.9% of total taxation, when the EU average is 0.3%. For pollution and resource charges, it is not possible to compare the rates applied, as the taxation systems are very different and comparable data is not available.

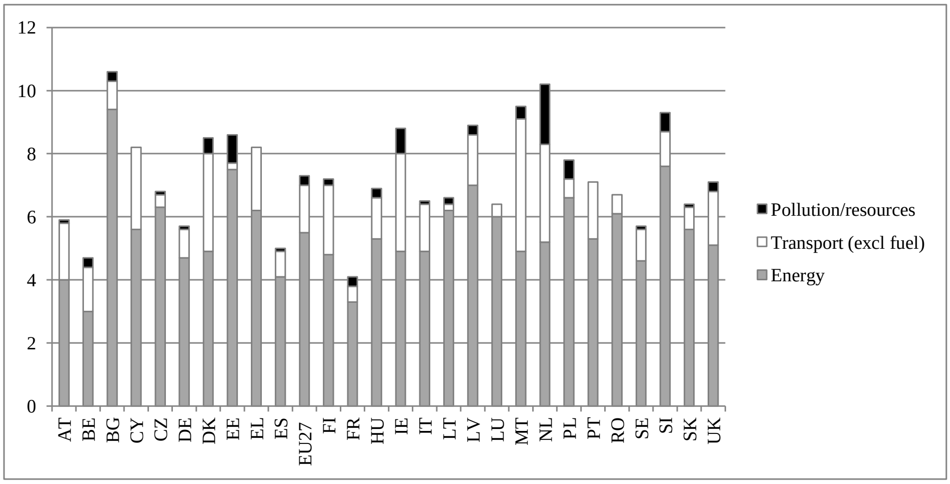

Figure 4.1.4. Environmental tax revenue as % of total taxation in 2011 (Eurostat 2013a)

Returning to the issue of the high share of energy tax, this could be for two reasons: the tax rate is high or energy consumption is high. Looking at the implicit tax rate on energy, which is the ratio between energy tax revenue and final energy consumption calculated for a calendar year (Eurostat 2013b), the Estonian tax rate (87.6 €/t oil equivalent in 2011) is about two times lower than the EU27 average (183.8). It is a general pattern that new member states have the lowest implicit tax rates on energy in the EU: lower values than in Estonia were in the Czech Republic, Hungary, Lithuania, Latvia, Bulgaria and Romania and the lowest in Slovakia (Eurostat 2013b).

To summarise the environmental taxation trends in Estonia in the context of the EU, the revenues collected from environmental taxes are above the EU27 average, but the environmental tax rates (at least those that are applied to energy products in the form of excises) are below the EU27 average. This seems to be a general trend for new member states in the EU, with some minor exceptions.

The EU has set energy and resource efficiency as one of its goals and different policies address this issue. For example, Europe 2020 sets three priorities: smart, sustainable and inclusive growth. As for sustainable growth, the topic emphasised is greenhouse gas emissions and the necessity of reducing those. The general targets of the EU are transposed to the targets at member state level, and although Estonia has not had difficulty in reaching the targets, this was not induced by specific actions but by the economic restructuring that has taken place compared to the baseline year 1990. Although pressure on the environment is not the topic of the thesis, the author believes that it gives useful background information for further analysis, which also covers behavioural change in different households. Greenhouse gas emissions are closely related to fuel combustion, although they include other sources as well. Comparing the changes in greenhouse gas emissions to changes in GDP over 2001–2011, we note that the changes are similar, although the changes in greenhouse gas emissions are steeper than for GDP. However, it seems safe to conclude that greenhouse gas emissions in the last decade have not decreased and decoupling of the two variables has not taken place.

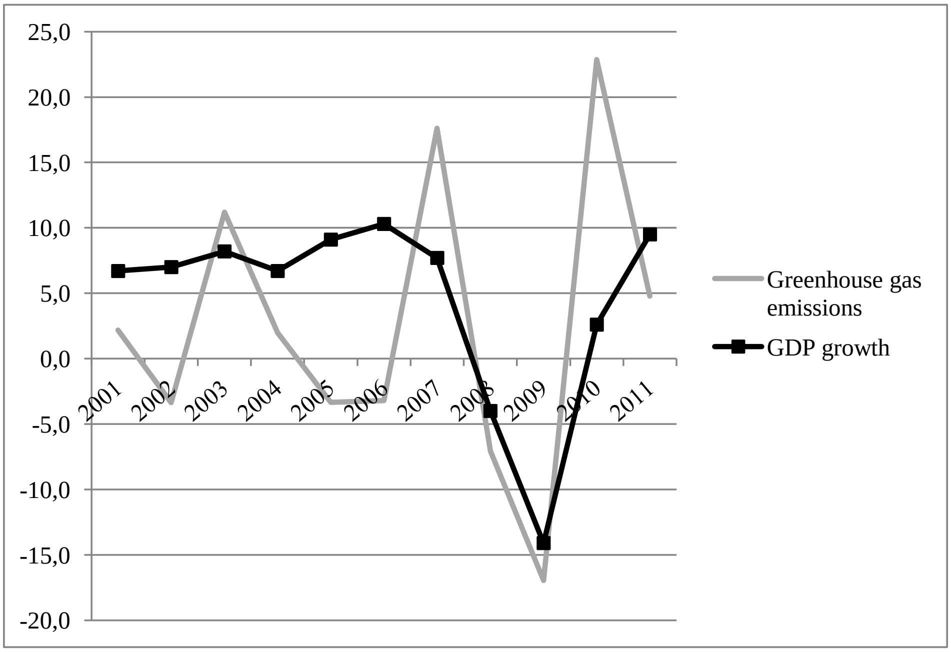

Figure 4.1.5. Change in greenhouse gas emissions and GDP, %, 2001–2011 (Source: author's calculations based on Eurostat 2013b)

Looking specifically at transportation, there is no good data available regarding the fuel used by households, as the amounts are presented under the household sector but also under different industrial and service sectors, as employers often compensate employees' fuel and car costs. However, looking at the travel behaviour of people, this indicates that the fuel used for commuting by personal car has increased over 2000–2011. It can be seen from Figure 4.1.6 that the share of employed people using private cars for travelling to work has increased from 24% to 38%. The modes of transport that have decreased the most are public transport (from 31% to 22%) and walking (from 28% to 18%). The use of the rest of the transport modes has been stable over this period.

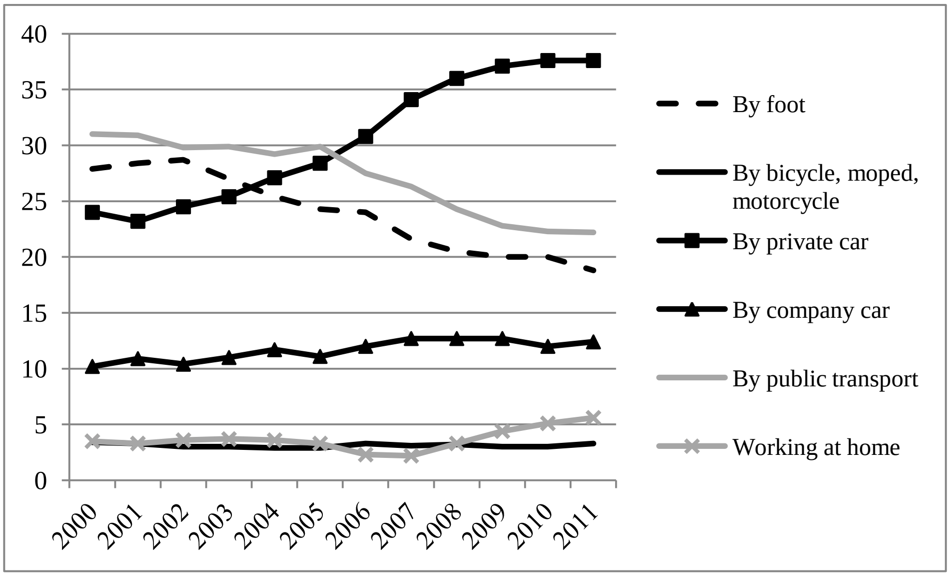

Figure 4.1.6. Means of transport used by employed people in Estonia to travel to work, 2000–2011, % of total employed (Statistics Estonia 2013)

The external costs of land transport in Estonia have been estimated to be €441 million in 2007, of which the biggest share results from accidents (€114 million), air pollution (€106 million) and noise (€80 million) (Anspal & Poltimäe 2009). By vehicle type, the largest contributors to the external costs of land transport are cars (€237 million), followed by trucks (€106 million) and vans (€59 million). At the same time, revenues from fuel excise in 2007 were €278 million, of which 75% was used on road maintenance. In addition, there are other instruments that internalise the external costs of road transport (traffic insurance, the heavy goods vehicle tax, environmental charges paid by electricity producers) and according to estimates, about 34% of external costs can be regarded as having been internalised by different policy instruments (Anspal & Poltimäe 2009).

Hence, the share of private cars in transport is increasing and so are the greenhouse gases resulting from that. According to the Principles of Economic Tax Reform adopted in 2007, the level of environmental taxes is not high enough to give appropriate signals to people and enterprises. Hence, the named principles were adopted by the Estonian government to change the tax system in order to sustainably utilise natural resources and the environment, and increase energy and resource efficiency and environmental awareness. The Principles of Economic Tax Reform raise existing environmental taxes, impose new environmental taxes (specifically related to private transport) and lower of personal income tax. In reality, only the raise of existing environmental taxes has taken place and no new taxes have been introduced. However, the rates of existing environmental taxes have been raised without investigating the effects on households or enterprises. The following sections of the thesis analyse the distributional and behavioural effects of Estonian environmental taxes on households.

4.2. Direct distributional effects of Estonian environmental taxes

Households are subject to different consumption taxes, of which the highest tax load comes from VAT: about 14% of total expenditure (Poltimäe & Võrk 2009). VAT is followed by fuel and tobacco excises, both forming up to 2% of expenditure depending on income decile, and alcohol excise (less than 1%). In the following, the amount and composition of tax load coming from environmental taxes (in this subsection, specifically fuel excise) have been analysed more closely.

The direct tax burden on households from the fuel excise in Estonia has been on average around 1% of expenditure in 2000–2003, increasing to about 1.3% in 2004 and to 1.8% in 2011 (see Figure 4.2.1; data is provided in annex 4). It can be seen from the figure that tax share increases correspond with larger tax rate increases which took place in 2004 and 2008.

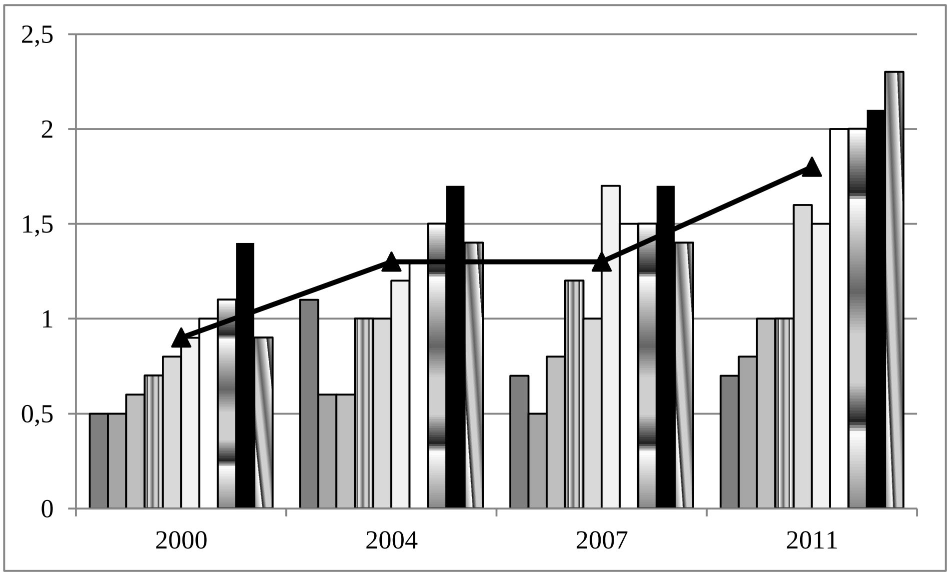

Figure 4.2.1. Direct environmental tax loads in Estonia as a percentage of expenditure by income deciles and as an average, 2000, 2004, 2007 and 2011 (author's calculations)

As can be seen from Figure 4.2.1, environmental taxes in Estonia are on average progressive, as lower income groups pay proportionally less than higher income groups. The tax load for all income groups has increased over 2000–2011; the increase has been smaller for low income deciles, although we can notice from the figure that the tax share in the mid-2000s in the lowest income decile was higher than at the beginning and the end of the 2000s. At least partly, the higher tax load for the first income decile comes from the nature of the data, as the HBS records monthly income and this is often volatile, with vacations, sickness, etc. Although expenditure data is used here, the division into deciles is based on income and hence the first income decile contains households that would actually be in higher income deciles if a longer time period was observed. But this could also be partly caused by behavioural responses to significant rises in fuel prices, especially by lower income groups

What has changed significantly during the considered time period is the composition of the environmental tax load, which in 2011 was much more varied than in previous years. Over 2000–2007 about 90% of the environmental tax load originated from the petrol excise and the remaining 10% from diesel. In 2011, the average share of the environmental tax load originating from the petrol excise was 66%, 17% from the diesel excise, 10% from electricity, 5% from district heating and about 1% from the excise imposed on gas and light fuel oil (see Figures 4.2.2 and 4.2.3; data is presented in annex 5).

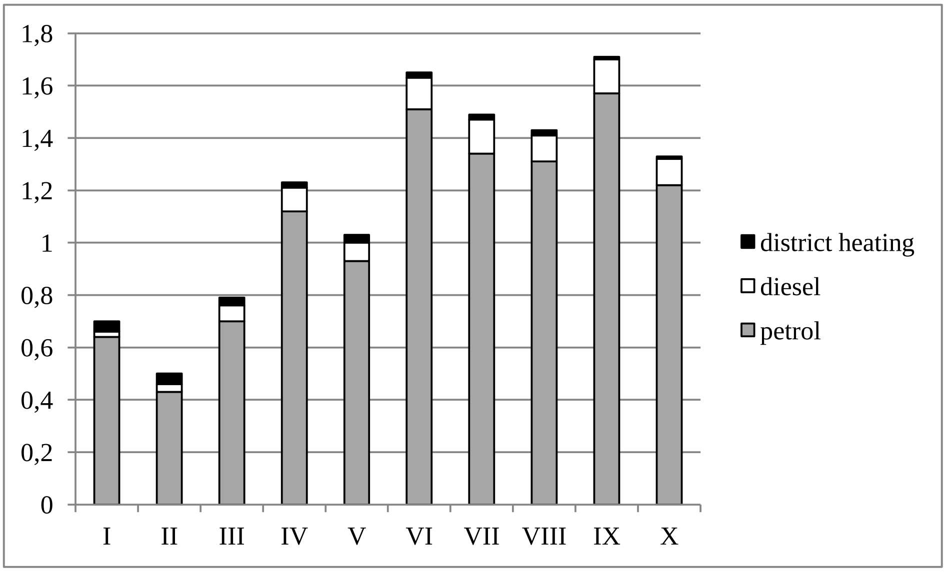

Figure 4.2.2. Direct environmental tax load in Estonia per household member in 2007 and its composition by income deciles, % of expenditure (author's calculations)

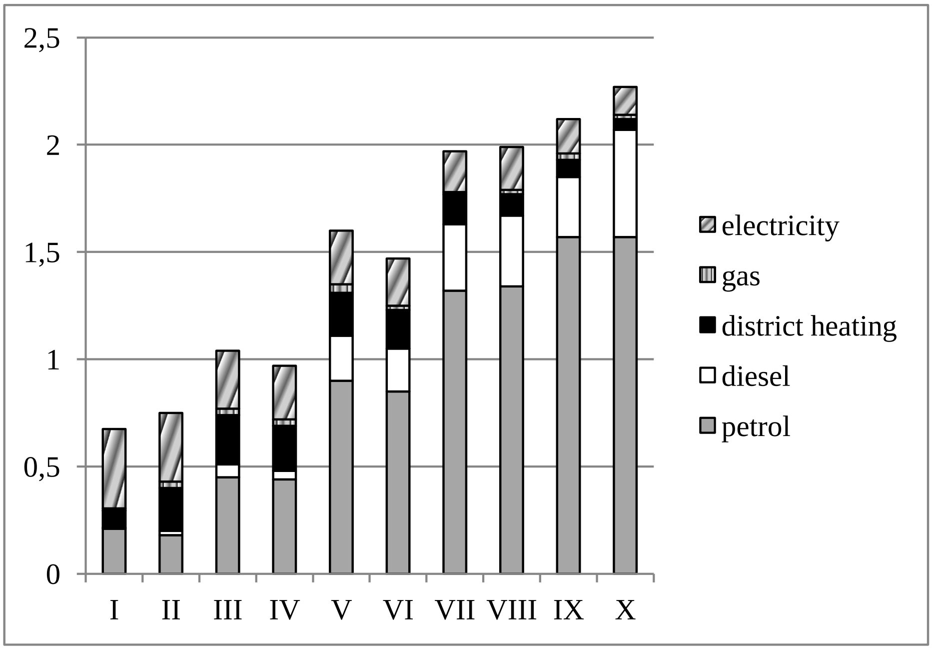

Figure 4.2.3. Direct environmental tax load in Estonia per household member in 2011 and its composition by income deciles, % of expenditure (author's calculations)

The decreasing share of petrol excise has come partly from decreased petrol consumption, as mean consumption in 2007 was about 30 litres per household and in 2011 it was 18 litres. The decreasing trend is confirmed in the aggregate data of Statistics Estonia, according to which the total petrol consumption in the household sector has decreased from 201 thousand tons in 2007 to 184 thousand tons in 2011. Meanwhile, the consumption of diesel has grown from 47 thousand tons to 52 thousand tons (Statistics Estonia 2013). So the trend of decreasing petrol consumption might be overestimated in the HBS data, but it is indisputable that the variety of the environmental tax load has increased as new fuel excises have been imposed. While motor fuel (petrol and diesel) taxation is progressive, as its share is increasing in higher income groups, the taxes based on electricity and district heating are regressive, meaning that for lower income groups, the tax load of the electricity excise and district heating excise is larger than for high income groups.

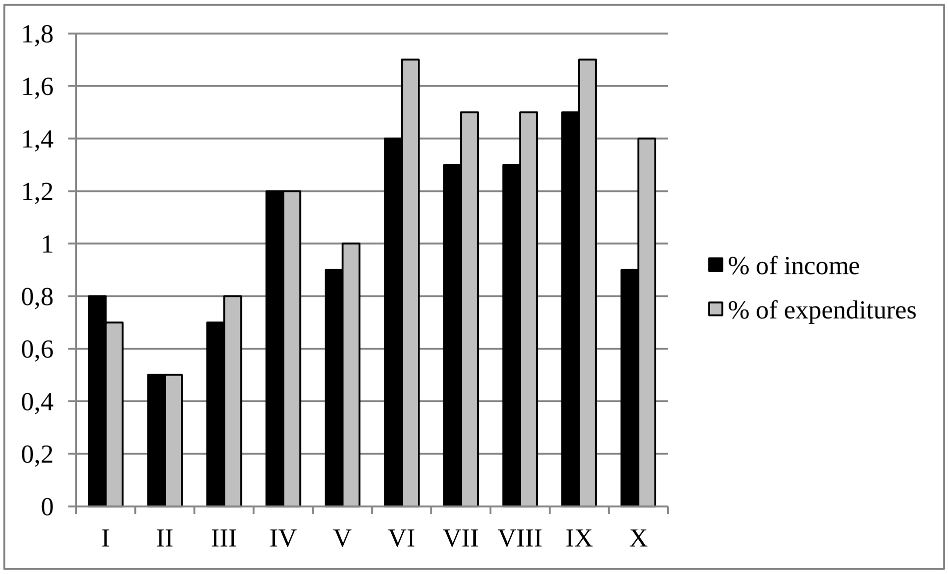

Figure 4.2.4. Direct environmental tax load in Estonia per household member as % of income and % of expenditure, 2007 (author's calculations)

To compare whether the result changes for different measures used (income versus expenditure), Figure 4.2.4 was compiled, and it turns out that even when using income data, the direct environmental tax load in Estonia is not regressive, although the progressivity appears less than when using expenditure data. This tendency is in line with results obtained in other empirical studies, for example that of Jacobsen et al. (2003), although they find energy taxes to be regressive according to income and proportional according to expenditure.

The differences between the income deciles have increased in the observed period: while in 2000 the highest income decile paid about twice as high an environmental tax load as the first income decile, in 2011 the difference increased to more than three times. The pattern remains the same in other income deciles selected (see Table 4.2.1). Hence the distributional pattern of the environmental tax load seems to be favourable for lower income groups.

Table 4.2.1. Decile ratios for a share of equalised environmental tax payments in consumption expenditures

	2000	2001	2002	2003	2004	2005	2006	2007	2011
10 / 1	1.8	1.7	1.4	1.7	1.3	0.8	1.1	2.0	3.3
10 / 2	1.8	2.0	2.5	2.5	2.3	1.2	1.7	2.8	2.9
5 / 1	1.6	1.2	1.3	2.0	0.9	1.2	1.3	1.4	2.3

Source: author's calculations

Additional measures applied in this thesis to analyse the progressivity of environmental taxes are the Kakwani and Reynols-Smolensky measures, whose results are given in Table 4.2.2. The Kakwani index has been positive for the most of the observed period, which means that the environmental tax load is progressive in Estonia, and the same trend is expressed by the Reynolds-Smolensky index. The progressivity of environmental taxes in 2011 is higher than ever before, which implies that regardless of the new imposed taxes that are regressive (specifically, the electricity excise), the general pattern has not changed, or has changed in a favourable direction for lower income groups. One of the reasons for increasing progressivity might be that low-income groups may have responded more to the fuel excise rate increases. This issue is discussed further in section 4.3.

Looking separately at taxed fuels (the Kakwani index for different fuels), the progressivity resulting from petrol and diesel excise has increased, as suspected above. The new excise duty imposed on electricity is regressive and the district heating excise is also regressive. Regarding light fuel oil, the pattern is changing from year to year, resulting perhaps from the fact that this fuel's consumption is not regular but it is purchased occasionally and for longer time periods than a month, and also the number of observations for light fuel oil is quite small.

Table 4.2.2. Progressivity/regressivity measures for Estonian environmental taxes

	2000	2001	2002	2003	2004	2005	2006	2007	2011
Kakwani index for environmental taxes	***0.03***	***0.06***	***0.01***	***0.03***	***0.05***	***-0.03***	***0.02***	***0.04***	***0.11***
Petrol	*0.04*	*0.08*	*0.01*	*0.04*	*0.04*	*–0.02*	*0.02*	*0.04*	*0.17*
Diesel	*0.03*	*–0.11*	*–0.05*	*0.03*	*0.17*	*0.05*	*0.06*	*0.06*	*0.23*
District heating	*–0.29*	*–0.27*	*–0.29*	*–0.34*	*–0.34*	*–0.32*	*–0.35*	*–0.33*	*–0.26*
Light fuel oil	*–0.64*	*0.33*	*0.33*	*0.07*	*0.31*	*–0.54*	*0.22*	*0.27*	*0.22*
Gas									*–0.10*
Electricity									*–0.19*
Reynolds-Smolensky	***0.0003***	***0.0006***	***0.0001***	***0.0003***	***0.0006***	***-0.0003***	***0.0002***	***0.0005***	***0.0019***

Source: author's calculations

According to household type, the highest environmental tax load applies to households with employed people (Figure 4.2.5; data in annex 6). However, although the general level of environmental tax load is lower for retired people, their share has increased considerably, forming 1.5% of their expenditure in 2011, while in 2000 it was only 0.6%. 65% of this tax load comes from petrol excise and 16% from electricity excise. For the households of the unemployed, the tax load increase has been modest compared to other groups: from 0.7% in 2000 to 1% in 2011.

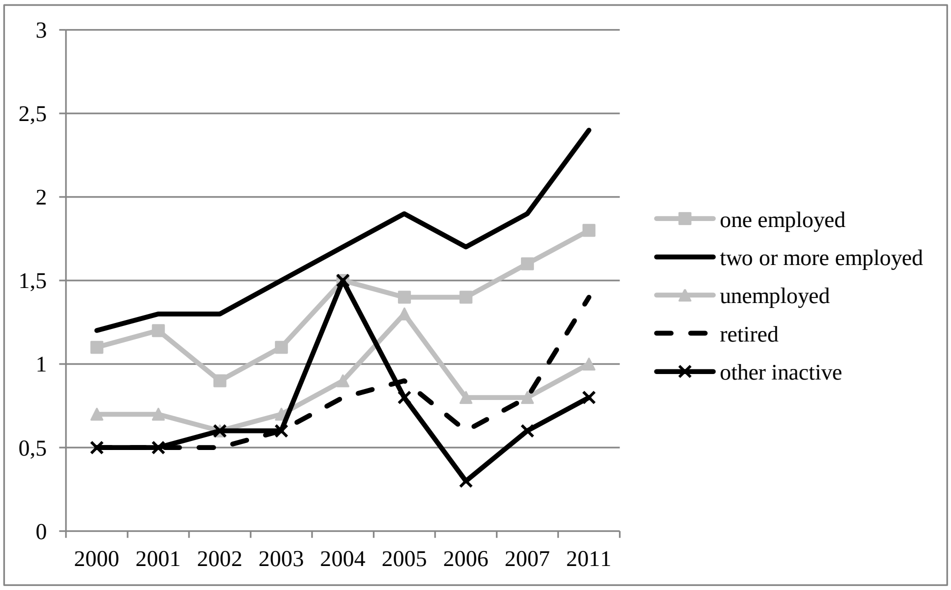

Figure 4.2.5. Direct environmental tax load in Estonia by household type 2000–2007 and 2011, % of consumption expenditure (author's calculations)

Looking at households by number of children, until 2005 the largest tax load of environmental taxes was borne by households with more than three children, but from 2006 the highest tax load fell on households with one or two children (Figure 4.2.6; data in annex 6). As a rule, households without children bear a smaller environmental tax load than households with children. In 2011, the environmental tax load for households with one or two children was 2.2% of expenditure, for households with three children 2.1% and for households without children 1.7%. As most of the environmental tax load comes from petrol excise, it can be concluded that households with children spend more on petrol.

Environmental tax differentiation according to settlement type in Estonia gives similar results to those shown by other authors: the environmental tax load is considerably higher for households living in rural areas than for households living in urban areas (Figure 4.2.7; data in annex 6). As most of the tax load results from motor fuel consumption, this is self-evident, as households living in rural areas need to rely more on private transport as distances are longer and no good alternatives are available.

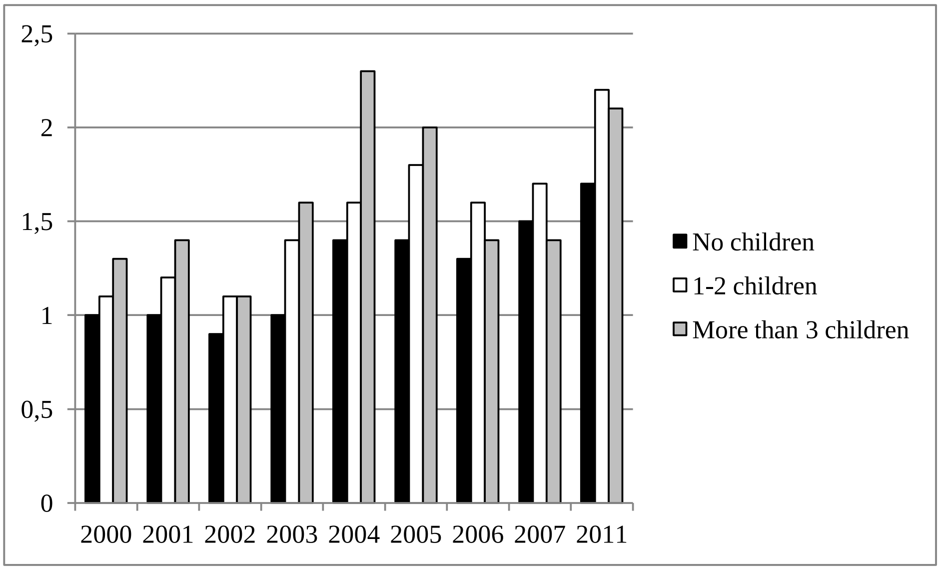

Figure 4.2.6. Direct environmental tax load in Estonia by number of children in 2000–2007 and 2011, % of consumption expenditure (author's calculations)

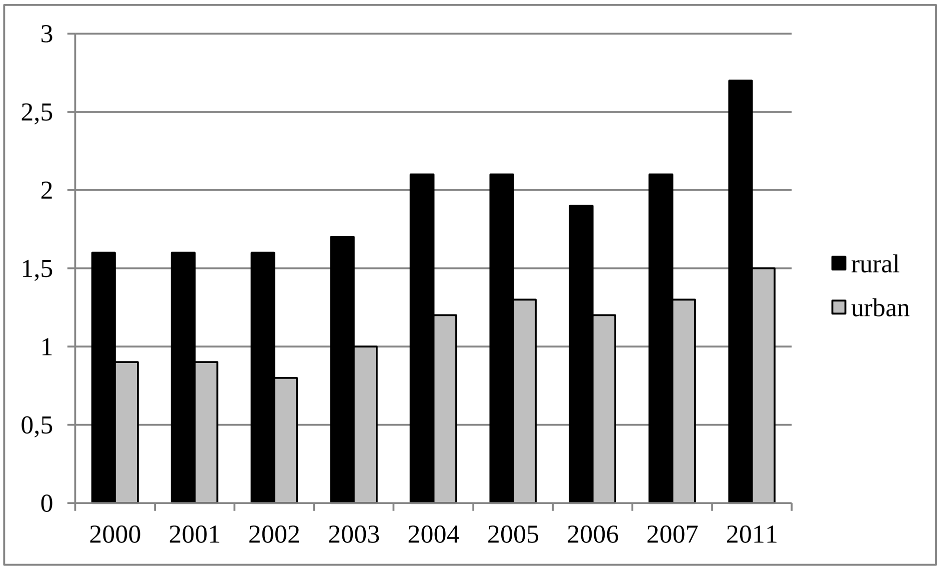

Figure 4.2.7. Direct environmental tax load in Estonia by settlement type in 2000–2007 and 2011, % of consumption expenditure (author's calculations)

To analyse the inequality resulting from environmental taxes according to normative measures, the Atkinson index was also applied. In the next table, the petrol excise amount for 2011 have been used as an example, as most of the environmental tax load originates from it. Epsilon, as explained in the methodological part of this thesis, is a measure of the degree of inequality aversion and as epsilon rises, the more weight is attached to transfers at the lower end of the distribution and less weight to transfers at the top (Atkinson 1970). Hence, where a spread of the Atkinson index across epsilon values is larger, the inequality is greater.

Table 4.2.3. Atkinson index for petrol excise, 2011

Values of epsilon	**All households**	**HH with one working member**	**HH with two working members**	**HH of unemployed**	**HH of retired**
0.5	0.13	0.13	0.13	0.11	0.10
1	0.26	0.26	0.27	0.21	0.20
2	0.52	0.53	0.54	0.37	0.41
		HH with one member	**HH with no children**	**HH with children**	
0.5		0.11	0.13	0.14	
1		0.21	0.24	0.28	
2		0.41	0.49	0.56	

Source: author's calculations

If we compare the Atkinson index for petrol excise across different household groups, we can see that for households of unemployed and of retired people, the distribution of petrol expenditure is more equal across different epsilon values compared to households with one or several working members. Also, the inequality is greater for households with children compared to households with no children and households where there is only one member. Hence, if using the normative approach and there is high inequality aversion in the society, a higher degree of inequality is related to households with children and households with working member(s), or perhaps these categories are related. This finding implies that perhaps in these household groups, the variety is greater than for example in the case of retired people. Also, turning back to the question of inequality aversion and social welfare function, it can be concluded based on the Atkinson index that inequality is much higher if more weight is attached to low-income groups (by increasing the value of epsilon).

The different measures used in the distributional analysis tell us that the richer population in Estonia bears a higher environmental tax load than the poorer population. Hence, the tax system seems to be fair to the poorer population. As a general trend for environmental taxes, this is different from most of the studies referred to in section 2.1, but the reason comes from the different tax structure: if heating fuels are taxed, then the pattern is regressive, but if motor fuels are taxed, there is more evidence of progressivity.

As can be seen from Figure 4.2.8, the ownership and use of cars follows quite similar patterns to the motor fuel excise progressivity, with the exception of the tenth income decile, which pays a smaller percentage of the motor fuel excise than the ninth but for which the ownership of cars is the highest. The percentage of households owning or using a car has increased in all income deciles over the observed period, but specifically in the lower ones. On average, car ownership and use has increased by 42% in Estonian households over the period 2000–2011, reaching 40–50% in 2011 in the lowest four income deciles, 60–70% for deciles V–VII, 80% for deciles VIII–IX and 90% in the highest income decile. Hence there is a clear progressive pattern in the ownership of cars in Estonian households, which can partly explain the progressivity pattern of the motor fuel excise.

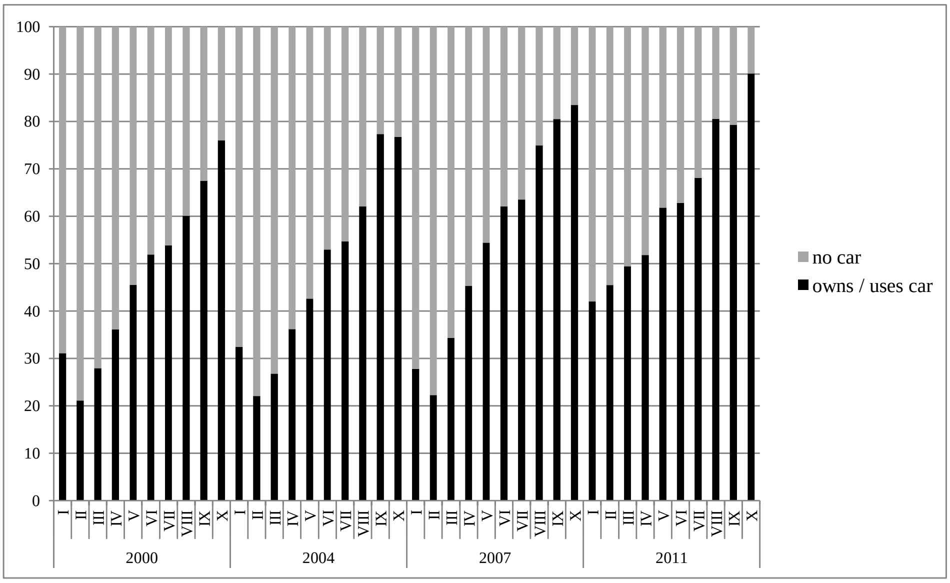

Figure 4.2.8. Ownership and use of cars by income deciles in Estonian households, 2000, 2004, 2007, 2011 (author's calculations based on Statistics Estonia HBS)

Looking at car ownership percentage by household type (Figure 4.2.9), the households of employed people have a higher car ownership percentage than others. For example, in 2011 in households of two employed people, the percentage owning cars was 88%, in households of one employed person 65%, while for unemployed, retired and other inactive people the figures were 38%, 34% and 24% respectively. Hence there is a clear relationship of car ownership to employment status. At the same time, we notice that car ownership has increased in the observed period for the last mentioned groups, which means that these now have better access to cars, although this is not necessarily related to car use. The increasing levels of car ownership imply that transport fuel use will also grow in the future and further measures to decrease greenhouse gas emissions are needed.

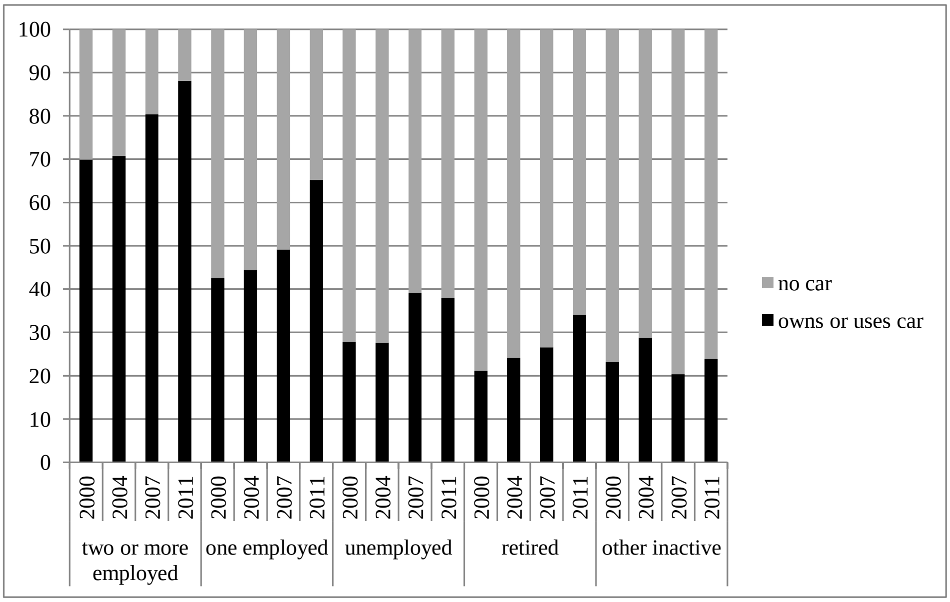

Figure 4.2.9. Car ownership and use by household type in Estonia, 2000, 2004, 2007 and 2011, % of households (author's calculations based on Statistics Estonia HBS)

In conclusion, it can be said that the direct distributional impact of environmental taxes is progressive, as Estonian fuel excises tend towards motor fuel taxation. However, the tax system development of recent years in the form of raising excises on heating fuels and electricity is unfavourable to the poorer population. It is not clear how much of this progressivity of motor fuel taxation comes from different responses by different income groups: for example, have poorer people responded more to higher taxation and decreased their car use and how does this relate to inequality? This aspect is discussed further in the section on behavioural changes.

As for specific household types vulnerable to environmental taxes, the findings of the thesis are in line with those of other studies: households living in rural areas bear a higher environmental tax load than those living in cities (also found in Bork 2006, Callan et al. 2009, Jacobsen et al. 2003), and households with children bear a higher tax load than ones without (also found in Bork 2006, Dresner & Ekins 2006). Dresner and Ekins (2006) also find retired people to be a vulnerable group, but this finding has not been confirmed in the case of Estonia: although the environmental tax load has increased substantially, it is still lower than for working people. This is again because of the nature of the main tax object, which in Estonia is motor fuel.

The direct tax load considers only taxation that is based on directly consumed goods, in this case fuels. However, fuel taxes have an effect on other goods as well, as transportation and energy use form part of the cost of almost all goods and services, although the patterns and importance might differ. The next section analyses the secondary distributional effects, i.e. distributional effects on households resulting from price changes induced by environmental taxes.

4.3. Indirect distributional effects of Estonian environmental taxes

In order to discover the indirect distributional effect, environmental charges can be considered in addition to fuel and electricity excises. Although according to revenue, the fuel excise contributes by far the most to state budgets, environmental charges are concentrated on a limited number of enterprises and specific sectors, and might pose a significant tax load for specific sectors or commodity groups.

As can be seen from Figure 4.3.1 (data in annex 8), fuel excise forms the largest share of environmental taxes for most sectors; considerable amounts are paid by the sectors of land transport (€73 million in 2009), services (€66 million) and construction (€24 million).[7] A different pattern appears for the energy sector, for which environmental charges form a majority: €47 million, i.e. 73% of the environmental tax load, of which in turn €27 million is paid as the pollution charge and €20 million as the resource charge. The fuel excise payment of the energy sector is €17 million, forming about 27% of the sector's environmental tax load. However, the named sectors differ as to high environmental taxes and charges load; for the rest of the sectors, the tax payments are less than €10 million per year. The share of the electricity excise is very small.

Figure 4.3.1 shows only how much each sector pays directly to the state budget. If we also take into account intersectoral commodity flows (how much one sector is using another sector's output as its input), the pattern changes. Figure 4.2.8 (data in annex 9) presents the share of environmental taxes and charges per production unit considering the intersectoral flows. It is highest in the sectors of land transport (7.6%), construction (4.5%), energy (4.4%) and mining and quarrying (3.2%). For agriculture and food manufacturing, the tax shares are also quite high: close to 2%. Hence we can see that for some sectors the tax share is quite high regardless of the fact that the tax payments in absolute numbers are low (e.g. agriculture), but for some sectors the opposite can be seen: the environmental tax amount in absolute numbers is high, but in relative numbers quite low, for example, services.

[7] Although transport services belong to the group of services in general, in this thesis this sector is kept separate, as fuel excise is a remarkable environmental tax in Estonia affecting transport services significantly.

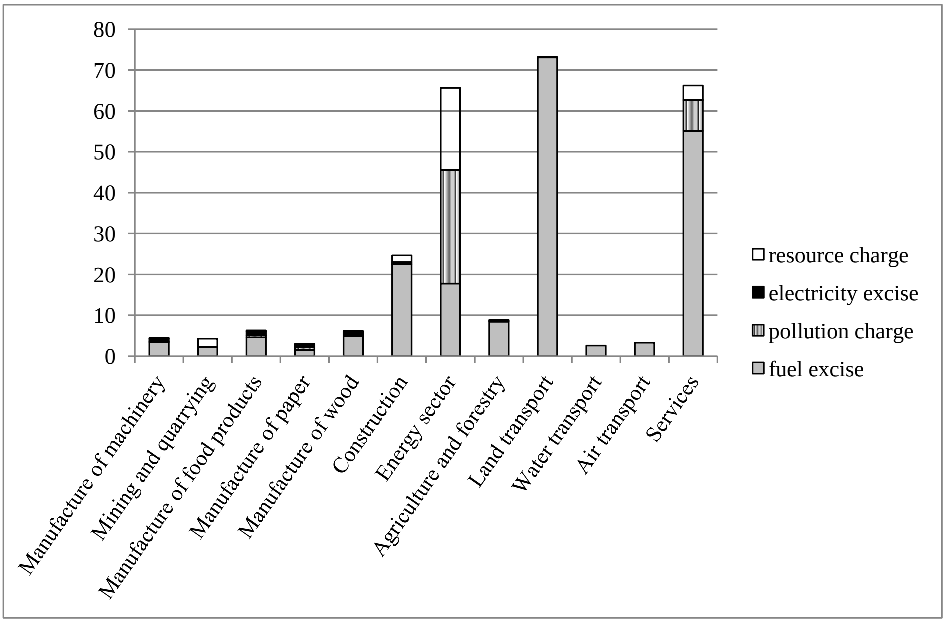

Figure 4.3.1. Environmental taxes and charges paid by economic sectors in Estonia, latest available data, millions euro[8] (author's calculations)

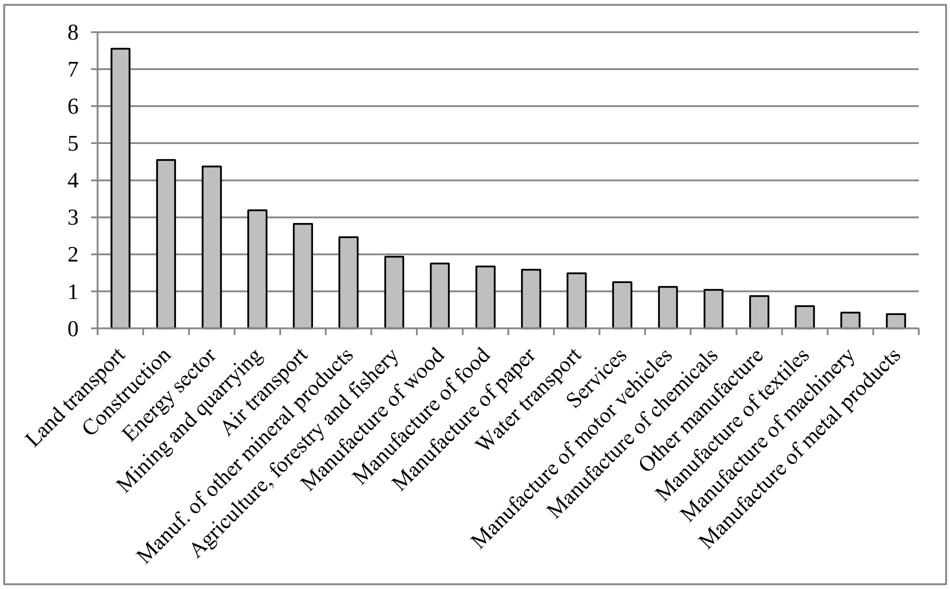

Figure 4.3.2. Environmental taxes and charges shares in sectoral output in Estonia, considering also intersectoral flows, % of output (author's calculations)

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8 Only sectors where total payment is larger than €4 million are presented in the figure. Environmental charges data is from 2009, the electricity excise is calculated based on 2007 electricity consumption data and the fuel excise is calculated based on 2009 fuel consumption data.

The indirect environmental tax loads of different sectors can be transferred to consumption categories consumed by households, which in turn allows us to assess the indirect distributional impact coming from the environmental taxation of enterprises. The highest proportion of the environmental tax share in price is found for the commodity groups of transport (3.2%) and housing (2.7%), but it is also quite high for food products, non-alcoholic beverages and alcoholic beverages (1.7%), as these sectors need significant input from the energy and transportation sector (see Figure 4.3.3; detailed data in annex 10).

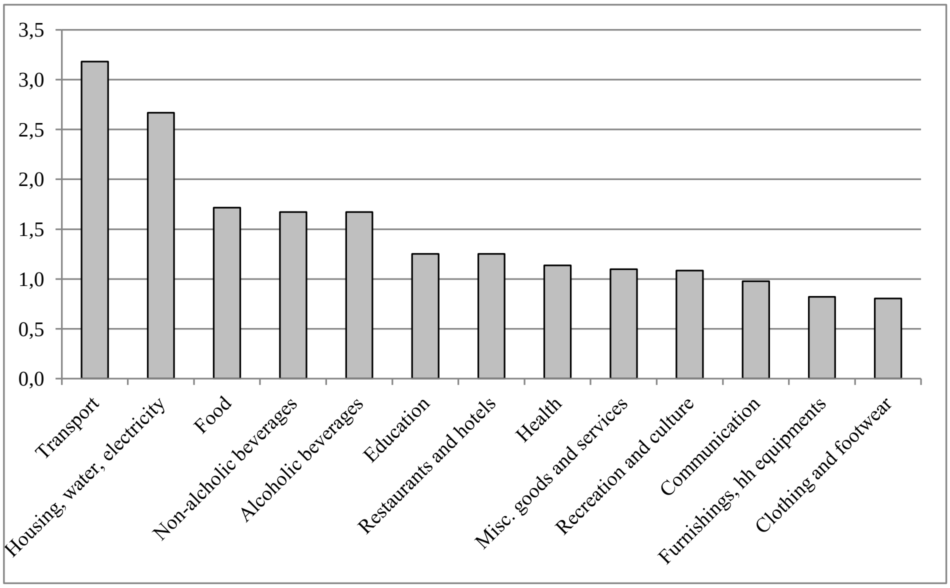

Figure 4.3.3. Indirect environmental tax load of consumption items in Estonia, 2011, % (author's calculations)

Linking the environmental tax loads of different commodity groups to household expenditure and analysing this by income decile, it can be seen that according to the share of income, the regressivity of the indirect environmental tax load appears (Figure 4.3.4; data in annex 12). According to the share of consumption expenditures, the pattern can be considered proportional or very slightly regressive. The regressivity of the indirect environmental tax load according to income can be explained by different savings proportions between income deciles: richer households save more than poorer households.

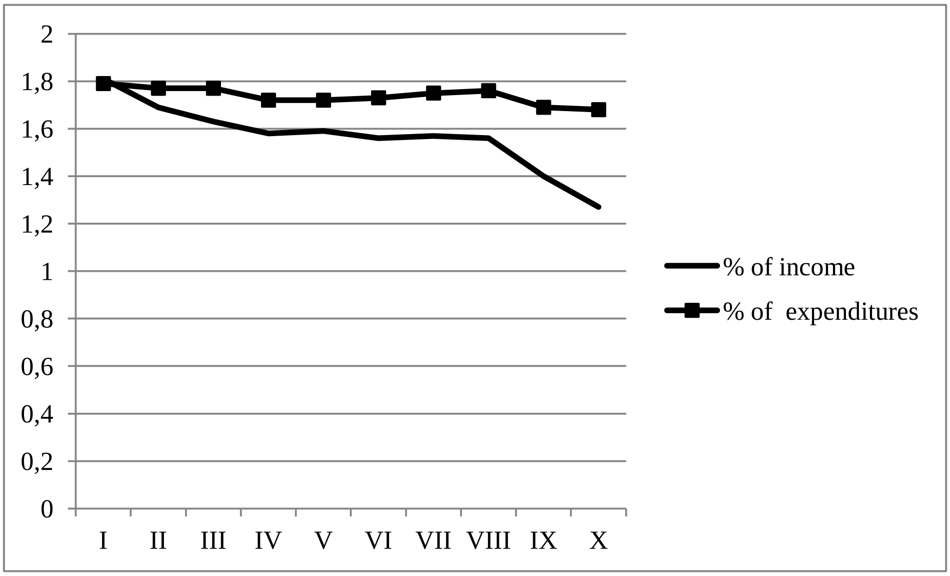

Figure 4.3.4. Indirect tax load of environmental taxes in Estonian household income and expenditure, 2011, percentage (author's calculations)

The fact that according to expenditure a slight regressivity also appears comes from the different consumption patterns of income deciles: lower income deciles spend more on food, alcohol and housing, which have higher shares of the environmental tax load than for some other consumption commodities which are consumed more by higher income deciles (clothing and footwear, culture and recreation, etc). For example, in the lowest deciles the expenditure on food, alcohol and housing forms more than 50%, but for the highest income decile the same share is 35%. As for transport, which also bears quite a high environmental tax load, the share of expenditure increases with income: for the first five income deciles, it is about 10%, while for the highest income deciles it is up to 17% (see Figure 4.3.5).

The regressive pattern of the indirect tax load of environmental taxes that has been found in this thesis is in accordance with empirical studies done in other countries, although those mostly concentrate on CO_2 taxes (the review is in section 2.2). The most affected sectors are also similar: the ones that are more energy-intensive (energy production, transport) and, as poorer households spend a higher share of their expenditure on energy-intensive products and services, this causes the regressive effect. However, in the case of Estonia, this regressivity coming from housing and heating expenditure is decreased by spending on transport, which is higher for high-income groups.

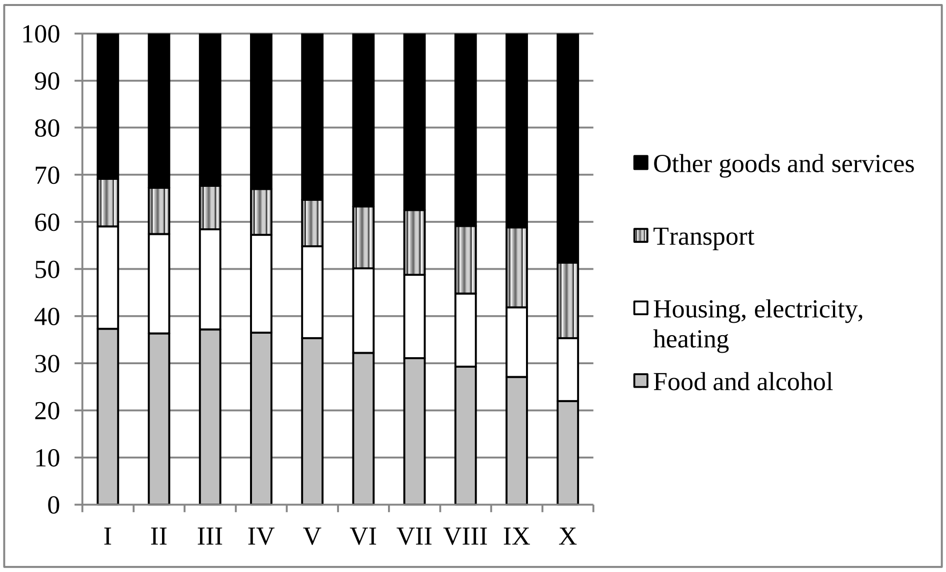

Figure 4.3.5. Estonian household expenditure composition by income decile in 2011, % (compiled by the author based on Statistics Estonia HBS)

Summing up the direct and indirect environmental tax shares for households in Estonia, we conclude that the direct distributional effect is progressive, mostly originating from the fact that high-income groups spend more on private transport and fuel, but the indirect effect resulting in large part from the same fuel excise is regressive and has a larger impact on lower income groups (see Figure 4.3.6, based on annexes 4 and 12). The proportions of both direct and indirect tax load are close to 2% on average. The proportion of the indirect tax load to the direct one is considerably higher than assessed by Wier et al. (2005) for Denmark: they find that the indirect environmental tax load in Denmark forms about one-third of the direct tax load for households. However, the Danish study concentrated only on the CO_2 tax, while here in this thesis the effects of all environmentally related taxes are analysed, including charges that cannot be considered under the direct distributional effect.

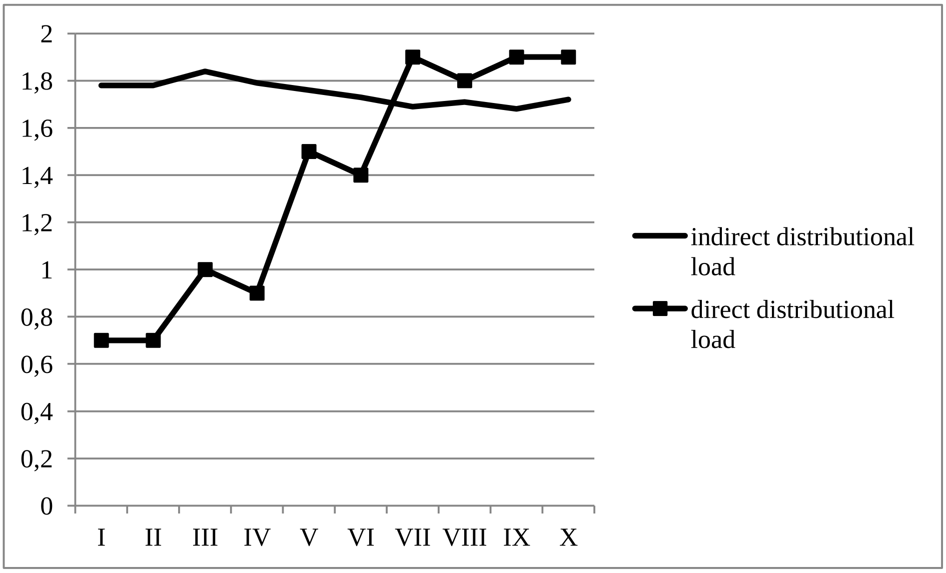

Figure 4.3.6. Direct and indirect environmental tax loads in Estonian households, % of expenditure, 2011 (author's calculations)

The pattern for the environmental tax load remained progressive in 2011, even when accounting for the regressivity of the indirect effect, as the total tax load ranged from 2.5% of expenditure for the lowest income deciles up to 3.6% for the highest income deciles (Figure 4.3.6 and annex 12). The same is affirmed by the redistributional measures, the Kakwani and Reynolds-Smolensky indices (Table 4.3.1): the general pattern is progressive, although its magnitude is decreased by the indirect distributional effect compared to the effect of the direct environmental tax load.

Table 4.3.1. Redistributional measures of direct and indirect environmental tax loads in Estonia in 2011

	Direct environmental tax load	**Direct and indirect environmental tax load**
Kakwani index	0.11	0.04
Reynolds-Smolensky index	0.0019	0.0011

Source: author's calculations

To sum up, the general distributional effect is favourable for the poorer population as the environmental tax share of expenditure paid by higher income groups is higher. Still, it must be noted that the fuel excise changes of recent years (new and higher taxes on heating fuels and electricity) have had adverse effect on poorer households and the indirect effect of environmental taxes, even when the direct effect is progressive, has a negative effect on poorer households, as they spend a higher share of their budgets on energy-intensive goods and services. Furthermore, it is also necessary to estimate how households have reacted to the price changes induced by environmental taxes to see whether the response has been different by income group: if the poorer population has reacted to price changes and the richer ones have not, then the inequality in society might actually be increasing.

4.4. Behavioural effects of Estonian environmental taxes

As seen from the overview of empirical studies provided in section 2.3, different methods can be used to analyse how people respond to changes in environmental taxes, which in turn change the prices of taxed goods. This section focuses only on petrol consumption, because the considerable tax changes for fuels used for domestic heating took place quite recently: electricity and gas excises were imposed in 2008, the light fuel oil excise rate was raised significantly in 2010. Hence the time period for analysing households' responses is too short: switching to alternative fuels or investing in insulation is a longer-term process, as the necessary investment is quite large.

The descriptive statistics of the researched data is presented in annex 14. The table contains the information for the whole researched period, 2000–2007 and 2011. We can see that the variability of monthly income and expenditure data is very high. According to calculations based on expenditure data, the average petrol quantity consumed per month by Estonian households over 2000–2011 was 21.2 litres. Excluding the zeros from calculation, the average consumed quantity is 60.9 litres. There are no good sources to validate these figures. One report, based on which the Statistical Office of Estonia estimates the energy quantities consumed by households, is the energy consumption in households, according to which in 2010 the average consumed quantity of petrol was 844 litres per household, calculated only for positive values (70.3 litres per month) (Statistics Estonia 2012b). Hence, the amount estimated based on the HBS is quite close.

As for the total consumption by household, the data can be validated by the aggregated petrol consumption of households, also reported by Statistics Estonia under sectoral fuel consumption data. Figure 4.4.1 shows that the petrol consumption data assessed in the HBS was significantly lower in 2000–2004 compared to the consumption data expressed by sectoral data on fuel consumption. However, it must be noted that the sectoral data is assessed by different sources: firstly, a specific survey that was last conducted in 2010, but the previous version was from 1996. In the intermediate period, the data was imputed. The petrol consumption data assessed based on the HBS follows quite closely the pattern of final expenditure of households in that period, except for

2011 when petrol consumption decreased much more abruptly than final consumption expenditure. Comparing the data of passenger-kilometres, which expresses transportation of one passenger over one kilometre, this has been constantly increasing over the observed period and hence the petrol consumption must have increased too.[9]

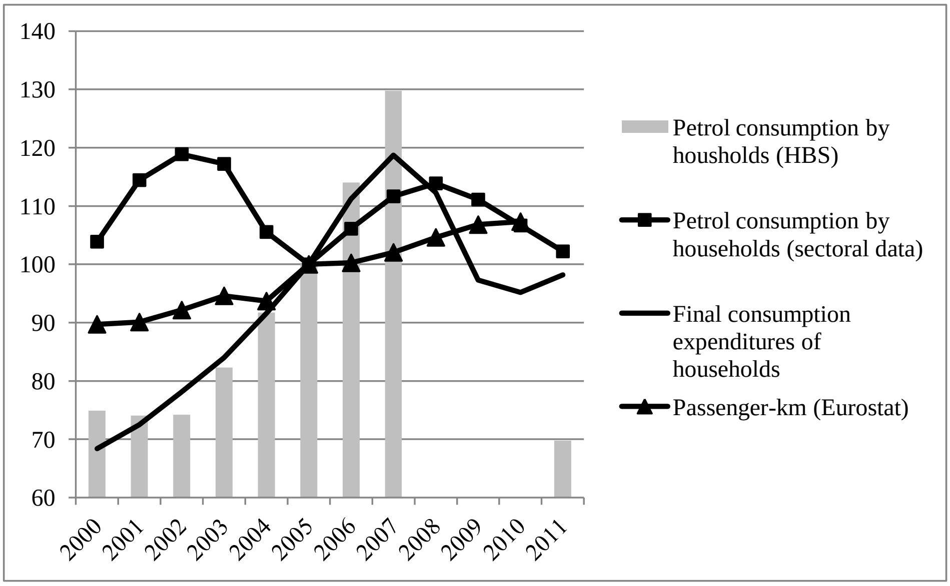

Figure 4.4.1. Validation of data of household petrol consumption, index, 2005=100 (compiled by the author based on Statistics Estonia HBS and Eurostat 2013b)

There are other trends taking place which could also affect decisions about whether to own a car and how much to use it (Figure 4.4.2): the price index of vehicle purchases significantly dropped in Estonia over 2000–2010, while the general price index has risen. The decision to use a car might be considerably affected by such a trend.

[9] It must be noted, however, that passenger transport includes transportation based on diesel.

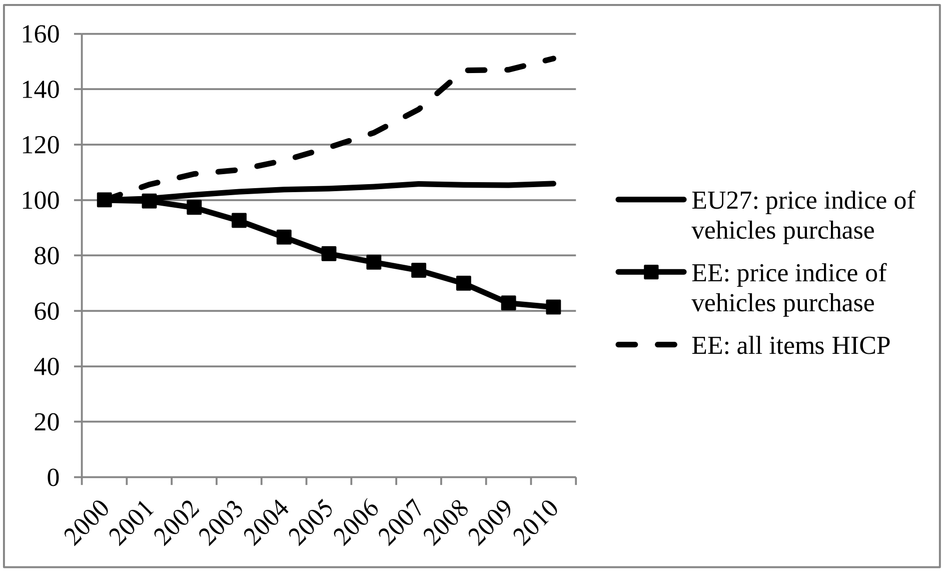

Figure 4.4.2. Price indices for purchasing vehicles and all items HICP, EU27 and Estonia, 1996–2010, 2000=100 (Eurostat 2013b)

The next section estimates petrol consumption income and price elasticities and discusses different socio-demographic characteristics of households that have an effect on that.

As discussed before, the problem with the HBS over 2000–2007 was the decreasing rate of response. As less than half of respondents are asked to respond in two sequential years and some fill in only one part of the survey (household picture, diary book, income data), and as less than half of those who fill in all parts claim to expend some money on petrol (see the following table for an overview of the proportion of zero expenditures), panel data methods are not used in this thesis, as the panel is very small. In addition, there were no households surveyed for both 2007 and 2010/2011, when the change of methodology of the HBS was implemented. Thus the thesis makes use of micro-econometrics, modelling first the decision whether a household spends on petrol or not and for the second step, the factors that influence the amount of expenditure among those who spend on petrol.

Table 4.4.1. Proportion of households reporting no expenditure on petrol in the Estonian Household Budget Survey

	2000	2004	2007	2011
Number of households reporting zero expenditures on petrol	3962	1940	1801	2638
Total sample	6006	3066	3174	3503
Percentage of households reporting zero expenditures on petrol	66	63.3	56.7	75.3

Source: author's calculations

The options for analysing such data are the Tobit model or the two-part model, of which the first part is the Probit model and the second part is truncated regression for positive values. The Tobit model assumes the same sign for two effects: what the household spends on petrol and the same sign for the amount of expenditure. This might not be the case in reality as some variables might have two-way effects on the dependent variable; to test that, a two-part model is assessed, containing the Probit model and the OLS/truncated regression model (see Table 4.4.2).

Table 4.4.2. Results of the two-part model, data for 2000–2007 and 2011[10]

	Probit (marginal effect)	**OLS for positive quantity**
Ln (expenditures)	0.284***	0.523***
Ln (price of petrol)		–0.564***
Number of adults	0.009***	–0.040***
Number of children	–0.021***	–0.041***
Gender of household head (0–male; 1–female)	–0.123***	–0.105***
Age of household head	–0.004***	–0.011***
Age of household head^2	–0.0001***	0.0001**
Education level (comparison level: elementary or lower)		
Secondary education	*0.043****	
Higher education	*0.037***	
Not known	*–0.056*	
Urban/rural	–0.206***	–0.103***
Household type (comparison level: one working member)		
Two or more working members	0.020***	0.033**
Unemployed	–0.027**	–0.080*
Retired	0.003	–0.112***
Other	–0.041***	–0.082
Ownership of car (number)		0.438***

Dependent variable in Probit model: spends on petrol or not (0/1), in OLS: ln(consumed petrol)
*** significant at 0.01 level; ** significant at 0.05 level; * significant at 0.1 level

[10] The yearly specifics have been controlled for, see more detailed data in annex 16

The first model assesses the probability that a household buys petrol at all and the second part the changes in consumed petrol quantity (only for positive values). The probability that a household spends on petrol increases with household size, but the relation is negative for petrol quantity, signalling that although the probability of using a car increases with every additional adult member, there is probably a synergy in its use and the level of expenditure is actually decreasing.

However, for a household with children, the relationship is negative: having more children decreases the probability that a household spends on petrol, and the amount of petrol consumed also decreases with the number of children. The conclusion regarding the relationship between the number of children and petrol expenditure is mixed in the empirical research presented in section 2.3: Kayser (2000) and Asensio (2002) find no significant relationship between the number of children and petrol expenditure, while Sardianou (2008) finds that petrol expenditure is higher for households without children. Hence the negative relationship found in this thesis is not in accordance with these studies. An explanation could be that larger families belong to lower income groups and hence cannot afford a car. It can also be seen from Figure 4.2.8 that car ownership and use are much lower in the first income deciles.

The remaining relationships are in line with previous studies: petrol expenditures are smaller in households headed by women, households living in urban areas and households whose heads are older. Similar results are confirmed for example in Kayser 2000, Asensio et al. 2002 and Sardianou 2008.

As for working status, the households of unemployed people have a lower probability of spending on petrol compared to households with employed members. Thus petrol consumption is related to working status and cars are increasingly being used for driving to work, as seen in Figure 4.1.6. Educational level is significant when determining the probability of whether a household spends on petrol, but not when determining the amount of petrol. The linkage is perhaps again via employment, as higher education levels increase the probability of working and hence also driving a car.

As for elasticities, the income elasticity[11] based on the assessed models is lower than expected: the meta-analyses done by Goodwin (2004) and Graham and Glaister (2004) estimate income elasticity to be slightly below or above unity in the long term. However, according to the models and data assessed here it is 0.52, which is similar to the short-term elasticity of the meta-analyses, but short-term is usually considered to be data for less than one year. One reason could be that the income elasticities of the meta-studies are overestimated, as argued by Basso and Oum (2007); the issue is more thoroughly discussed in section 2.3. We can also see from Table 2.3.1 that income elasticities based on

[11] Note that in this chapter, expenditure elasticity is used as synonym to income elasticity, based on the theoretical argument that expenditure level is a better proxy for lifetime income than income level.

studies using two-part models are lower than the results of meta-studies and the ones based on AIDS, and do not generally exceed 0.5.

The price elasticity based on the OLS model for positive amounts is –0.56, which means that a 1% increase in price results in a 0.56% decrease in consumption on average. Price elasticities based on the meta-analyses discussed in section 2.3 are –0.63 and –0.77 in the long term and around –0.3 in the short term. Hence the Estonian estimate is quite close to those of the meta-studies. However, it is higher than the one estimated by Dahl (2012) for Estonia, which was –0.32. It seems that that estimate was based on aggregate data and is quite similar in all the Eastern European countries according to this source.

A possible explanation is that only private expenditures are recorded under the HBS and employers in Estonia use the compensation of a car and petrol expenditure as part of a motivation scheme. This could explain why income elasticity is lower than expected. Unfortunately, the amount of petrol compensated for by employers has not been researched in Estonia and the issue has not been included in the HBS up to 2011. Based on 2011 data, we can see that more than two-thirds of respondents who claimed to have petrol compensation by employers belong to the four highest income decile groups and about one-third of these belong to the highest income decile, so there might be a relationship between income level and the probability of petrol compensation by an employer.

Comparing the estimated income and price elasticities, it can be seen that their magnitude is the same, although according to the literature income elasticity should be higher than price elasticity. To answer the research question about whether price changes in petrol excise have led to changes in consumption, to some extent yes, but at a slower pace than the price increase: when the price increases by 1%, consumption decreases by 0.56%. To relate the behavioural issues to the distributional ones, one way is to look at the income elasticity, which is below unity (0.52), according to which petrol can be considered a necessity. This is not in line with the distributional effect found in previous chapters, which was progressive and mainly resulted from motor fuel consumption. However, income elasticity might be underestimated, as the data does not contain petrol consumption which is compensated for by employers and as there are other factors than income related to car ownership and petrol consumption, for example, a relative reduction in the prices of vehicles and loans.

To analyse the effects of different variables on petrol consumption more in detail, quantile regression is additionally applied in this thesis. The results are presented in Table 4.4.3 and Figure 4.4.3. It must be remembered that in quantile regression, the percentiles are based on the dependent variable: in this case, petrol consumption. Only positive amounts are included in quantile regression.

Table 4.4.3. Results of quantile regression

	0.10	**0.25**	**0.50**	**0.75**	**0.90**
Ln(expenditures)	0.46***	0.51***	0.54***	0.59***	0.60***
Ln(price)	–0.19**	–0.15***	–0.22***	–0.26***	–0.23***
Adults	–0.05**	–0.05***	–0.03***	–0.05***	–0.05***
Children	–0.05**	–0.04***	–0.04***	–0.04***	–0.03**
Gender	–0.14***	–0.10***	–0.10***	–0.09***	–0.08***
Age	–0.01	–0.003***	–0.01**	–0.01***	–0.01***
Urban/rural (urban=1)	–0.04	–0.09***	–0.11***	–0.14***	–0.14***
Household type (comparison level: one working member)					
Two working members	0.04	0.07***	0.03	0.02	0.01
Unemployed	–0.01	–0.03	–0.07	–0.08	–0.18***
Retired	0.01	–0.04	–0.12***	–0.13***	–0.16***
Other	0.12	–0.09	–0.10	–0.16***	–0.19***
Number of cars	0.54***	0.51***	0.42***	0.33***	0.26***

Source: author's calculations

The results tell us that the relationship between petrol expenditure and total expenditure is more pronounced in the higher petrol expenditure groups, so we suspect that for those who do not spend as much on petrol, other factors are relatively more important. Also, the price effect is higher in upper expenditure groups as a general trend; however, the variance in the lowest 20th percentile is very large. The number of children decreases petrol consumption more in the groups with lower petrol expenditure, but the differences are small. There is also a clear relationship between settlement type (urban/rural) and number of cars. For lower petrol consumption groups, there is not much difference whether a household lives in an urban or rural area, but for high petrol expenditure groups, urban households spend 15% less than rural households. In the low petrol consumption groups, an increasing number of cars increases petrol consumption much more than in the groups of high petrol consumption (0.54 for the 10th percentile and 0.26 for the 90th percentile).

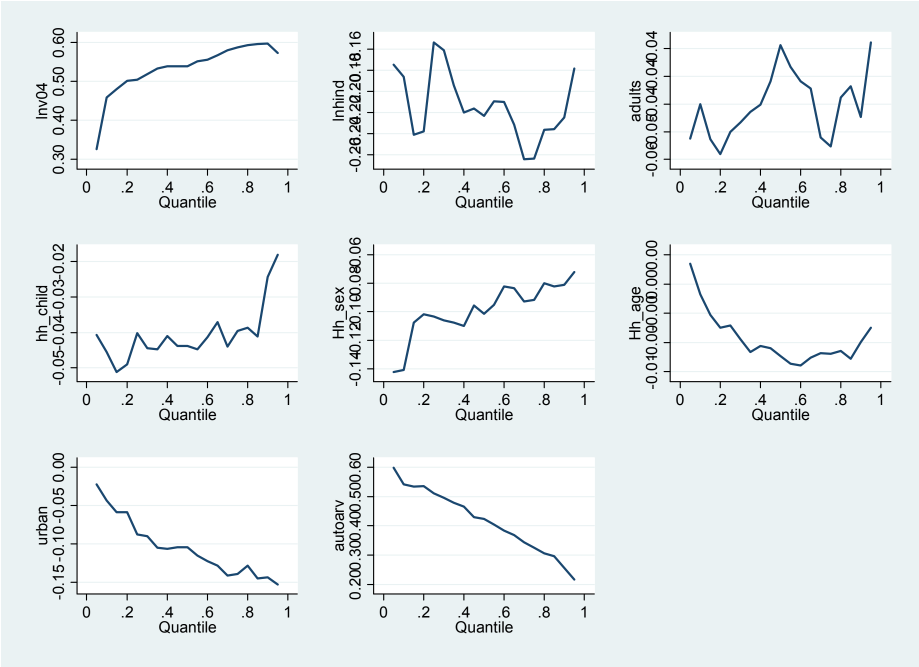

Figure 4.4.3. Graphical results of quantile regression (compiled by the author)

To sum up the results of the quantile regression, larger differences across different consumption groups appear in terms of the number of cars owned, type of residence, income level and gender of the household head. In households which consume the largest amounts of petrol, the effect of income is the largest, while the effect of additional cars in a household is the lowest compared to households with lower consumption quantities. As for households which consume the smallest amounts of petrol, type of residence is not significant, but the more petrol a household consumes, the larger the difference between urban and rural households.

The findings of the quantile regression imply that for those households that consume relatively small amounts of petrol, the effect of income is lower than in higher consumption quantiles. At the same time, the importance of additional cars in a household has a strong effect on these households. Hence, policy instruments to tackle increasing motor fuel consumption should be aimed at the purchase of more than one car in a household, and perhaps this is also related to the fact that income does not have such a large effect on consumption quantities.

CONCLUSIONS

Environmental taxes are an interesting research object: on one hand, they are intended to address environmental problems and are targeted at changing the consumption behaviour of households and enterprises. On the other hand, they are taxes, belonging to the group of instruments that have distributional effects in a society by taking money from people and delivering it back in the form of direct transfers or public goods. This thesis aimed to finding out the distributional effects of environmental taxes and the possible interlinkages to their effectiveness in terms of behaviour change in the example of Estonia. However, the implications of this research for policy are wider and could also be used by other countries which are experiencing rapid economic and consumption growth and also tightening of environmental requirements. As can be seen from the literature overview, so far research on the effects of environmental taxes has been mainly done for developed countries and these effects might not be the same in Central and Eastern European countries.

From a theoretical point of view, environmental taxes are an instrument for addressing externalities. A negative externality leads to a situation where private marginal costs and social marginal costs differ, which in turn leads to inefficient allocation, as the price of a good is too low and the quantity too large than when considering all related costs. According to Pigou (1920), the tax level should be set according to the marginal cost of the environmental damage. However, such a first-best instrument cannot always be used, if the pollution cannot be directly monitored or it is too expensive to do so. A branch of literature discusses optimal tax levels; Bovenberg and Mooij (1994) demonstrate that it depends on existing employment taxes, while Baumol and Oates (1995) argue that the tax level should be set so that the environmental objective is reached. Contrary to the last position, Common and Stagl (2005) argue that environmental objectives are arbitrary and, although environmental taxes lead in the right direction, there is no guarantee that the standard will be achieved.

Apart from specific work on environmental taxes, larger taxation principles can also be applied. A known example is the Ramsey taxation principle (1927), according to which those goods should be taxed for which demand is inelastic. The Ramsey taxation principle accords with the properties of a good tax system, as specified for example by (Stiglitz 1988), in which a tax should be easy to administer and should not hinder the efficient allocation of resources. Efficient resource allocation can be assessed by the Pareto criterion, according to which a decision should be implemented if it makes some people better off and no people worse off. However, in reality such decisions are unlikely, where nobody loses, and the potential Pareto improvement criterion is used instead: if the winners can theoretically compensate the losers, the decision is desirable. This idea is related to the utilitarian rule, which maximises the sum of the individual utilities. A different approach is proposed by Rawls (1971), that welfare maximisation means maximising the income of the poorest person.

The utility function is a well-known concept in economics. However, in empirics the term 'welfare' is hard to measure and hence income level or consumption level is used, as different authors have shown that consumption level is a better proxy for lifetime income than current income. Considering the externalities, the consumption of goods that have externalities affects utility negatively. Hence, environmental taxes have positive effects on those who suffer through externalities and negative effects on those who produce the externalities. But as a tax instrument, there are implications for distributional issues as well.

Looking back into theories of income distribution (a good overview is provided by Sandmo 2013), it can be seen that the classical school of economics dealt mostly with functional income distribution, i.e. with income from land, capital and labour. It was only the later generations of the neoclassical school that acknowledged that wage differences make the greatest contribution to income inequality and then the idea of human capital started to spread. On the normative side, there have also been theories that perceive income distribution as unequal because of the wastefulness of such inequality, as a higher welfare level is achieved by distributing the same income equally than by distributing it unequally.

This thesis takes a normative approach, as it analyses the distributional effects of environmental taxes, perceiving increasing inequality in a society as an undesirable tendency. The empirical studies done before on the subject have a similar approach, but mostly these were done in developed countries where income levels and tax systems are more stable than in case of, for example, new member states of the European Union. In short, the results produced so far regarding the direct distributional effects of environmental taxes indicate that if taxes are imposed on fuels used for domestic heating and electricity, there is an adverse effect on poorer households (for example, Barker & Köhler 1998, Bork 2006, Callan et al. 2009, Dresner & Ekins 2006, Jacobsen et al. 2003). Taxes imposed on vehicles are progressive (Jacobsen et al. 2003). As for motor fuels, results are mixed, ranging from regressive to progressive. Sterner et al. (2012) find that it is more likely to be slightly regressive or neutral in developed countries and progressive in developing countries.

As for the indirect distributional effects, it has been shown that these are regressive, as larger shares fall on the energy-intensive sectors (for example, Cornwell & Creedy 1996, Grainger & Colstad 2010, Kerkhof et al. 2008, Wier et al. 2005). On the contrary, Labanderia and Labeaga (1999) estimated them to be regressive. However, all these studies have been done only for carbon taxes and their results do not necessarily hold for other environmental taxes.

One way of estimating the behavioural effects (how consumption has responded to the price changes) is based on elasticities. The methods for estimating the demand elasticities of motor fuel are very varied: from two-part models to several equation demand systems. The results are varied as well; according to some studies, motor fuel is a necessity (Asensio et al. 2002, Kayser 2000, Sardianou 2008), while according to others it is a luxury good (Barros & Pietro-Rodriguez 2008, Labandeira et al. 2005). Some of these differences might be explained by the time perspective: according to meta-studies, long-run elasticities are higher than those for the short run. In addition, there is a tendency for price elasticities to be higher in Europe than in the USA. Taking this into account, there still seems to be a relationship to methdology as well: the elasticities calculated based on demand systems are higher than the ones calculated based on two-part models. This is in accordance with the conclusion of Basso and Oum (2007) that excluding demographic effects leads to over-estimation of income elasticities.

In the following, the empirical findings of the thesis are presented according to the propositions set up and the research questions.

Table I. Research questions, propositions and summary results of the research

Research questions and propositions	**Confirmed or not**
RQ1. What are the direct distributional effects of environmental taxes in Estonia?	
Proposition 1. Environmental taxes in general are regressive and hence increase the income inequality in a country.	Not confirmed: environmental taxes in Estonia are progressive and do not increase income inequality.
Proposition 2. The adverse effect of environmental taxes appears specifically for certain taxes (on heating fuels) and certain household types (retired people, households with children, households living in rural areas).	Confirmed: taxation of heating fuels and electricity is regressive in Estonia. The household types that are affected most are households of employed people and households with children. Comparing households by settlement type, households living in rural areas are affected more than households living in urban areas.
RQ2. What are the indirect distributional effects resulting from price changes induced by environmental taxes in Estonia?	
Proposition 3. Households in low-income groups consume more goods for which the environmental tax load is higher.	Confirmed: as households in low-income groups consume proportionally more food, alcohol and housing and these sectors' environmental tax load is higher, then the indirect distributional effect is regressive.
Proposition 4. The direct and indirect distributional effects of environmental taxes are increasing income inequality.	Not confirmed: as the magnitude of the progressivity of the direct distributional effect is greater than that of the regressivity of the indirect distributional effect, the general pattern is progressive, thus decreasing income inequality.
RQ3. Which has been households' consumption response to environmental taxes and how is it related to their socio-demographic characteristics?	
Proposition 5. On average, households' response to environmental taxes (specifically motor fuel excise) has been low and consumption of a taxed good has not decreased significantly.	Confirmed: the price elasticity of petrol consumption is rather low and rather than price changes, the economic crisis and increasing unemployment have changed the consumption pattern of motor fuel.
Proposition 6. The reaction to price changes differs by specific socio-demographic characteristics.	No clear conclusions. The reaction to price change is very varied and differs by socio-demographic characteristics, but clear patterns cannot be specified, as the data includes only private expenditure on petrol and other factors have an effect as well: for example, car prices, loan availability, public transport availability and quality, etc.

Research question 1: What is the direct distributional effect of environmental taxes in Estonia?

It was found that the direct distributional effect of environmental taxes in Estonia is progressive. This results from the tax burden of the motor fuels excise, which is progressive. However, the fuel excise imposed on electricity is clearly regressive, but its magnitude is not the same as for the motor fuels excise and hence the overall pattern of fuel and electricity excises is progressive, confirmed by the different measures used.

The household groups that are affected the most are households of employed people and households with children. The suspected adverse effect on retired people was not confirmed. An important variable affecting the tax burden is settlement type, as households living in rural areas have to drive longer distances and do not have good alternatives in a form of public transportation compared to households living in urban areas.

Research question 2. What is the indirect distributional effect resulting from price changes induced by environmental taxes in Estonia?

The indirect distributional effect of Estonian environmental taxes is regressive, resulting from the different consumption patterns by different income deciles. The lower income deciles spend more on food, alcohol and housing. The environmental tax shares of these products are higher than for commodities that are related to services or durables (for example, communication, recreational services, furnishings, etc).

Summing up the direct and indirect distributional effects, the overall pattern is still progressive, as the magnitude of the direct distributional effect is greater than for the indirect effect. Hence, the distributional effects of environmental taxes are decreasing income inequality in Estonia. However, if the same environmental taxation policy is continued in Estonia, this might soon turn into adverse distributional effects, because the share of new, regressive taxes has increased in recent years and the indirect effect of the analysed progressive taxes is regressive.

Research question 3. Which has been households' consumption response to environmental taxes and how is this related to their socio-demographic characteristics?

Households' consumption response to environmental taxes was estimated based on elasticities and was limited to petrol. A distinction was made between two consumption decisions: firstly, whether a household buys petrol at all (this is a proxy for the decision about whether to own a car) and secondly, how much petrol to use. It appears that income level affects the decision to own a car positively, but its magnitude is rather low. It seems that the decision is more related to employment status: if somebody in a household is working, then it is probable that the household spends on petrol.

As for demographic variables, petrol consumption is lower in households headed by women and it increases with the age of the household head and decreases with the number of children. The number of adults in a household is positively related to the decision to own a car, but negatively related to the amount of petrol consumed.

Both, the price and income elasticity of households' petrol consumption are lower than expected: income elasticity is 0.52 and price elasticity is –0.56. The low income elasticity can be at least partly explained in that petrol consumption is often compensated for by employers. Another reason might be that car prices and loan conditions were favourable during the years of economic growth and hence car ownership is increasing even in low-income groups. According to income elasticity, petrol consumption is not a luxury good but a necessity.

The environmental taxes in Estonia have not worsened the equity in society, at least in the form of monetary payments. In respect of the effectiveness of the taxes, whether these have led to a decrease in consumption of motor fuel cannot be confirmed. It seems that there are more important factors that have driven the decrease in consumption in the economic crisis years: the crisis itself and people's negative perceptions of future consumption opportunities, but also employment status: car ownership and use seem to be closely related to whether any household members are working or not. This is perhaps one of the relationships with the progressive pattern of tax distribution: these who work earn more and consume more, including the consumption of petrol.

As for the price elasticity, this is estimated to be –0.56 in the thesis, which implies that consumption reacts to price, but not very sharply. The fuel excise on petrol is in line with good tax policy as it has been a good source for state budgets and can be classified as a fiscal environmental tax. It is also in line with the Ramsey taxation principle, which says that goods with low demand elasticity should be taxed. However, what this principle does not account for is the existence of externalities (the taxation principle proposed by Pigou). For example, according to an estimate (Anspal & Poltimäe 2009), the external costs of land transport in 2007 were significantly higher than the revenues from the fuel excise, even without considering the fact that 75% of fuel excise revenues is used on road maintenance. Hence the level of fuel excise should be much higher to be environmentally effective and so decrease the related environmental problem. To sum up, this could be a general problem when addressing environmental policy with tax instruments, as different criteria are applied: environmental effectiveness versus tax efficiency.

The policy implications for emerging countries are the following: taxation of fuels used for domestic heating is regressive and increases income inequality in a country. This regressive pattern is in line with those of studies done for developed countries. For motor fuel taxation, there is no clear pattern from different empirical studies. Based on this thesis, we can claim that even when the direct distributional effect of tax on motor fuel is progressive, there are secondary effects that are regressive, although smaller in magnitude. However, it cannot be claimed that motor fuel and car driving are luxury goods in Estonia, which is the reason that motor fuel taxation is progressive in developing countries as well. Hence the implications for Estonia seem to be somewhere between those for developed and developing countries: its taxation is progressive, but is not a luxury good according to income elasticity. In terms of consumers' responsiveness to price changes, it seems that the possibility of price increases driving the demand is limited, as income has risen quickly, but other factors also seem important as drivers of demand. Hence, if a tax is targeted at effectiveness, a decrease in a taxed good, petrol tax does not achieve this.

Further analysis on the subject is dependent on data availability, as the limitations of this thesis are related to a lack of data. The data used in this thesis is about the general consumption budgets of households and does not include specific environmentally related questions. A more thorough analysis could be carried out if data was collected about environmentally related behaviour on an individual or household level: for example, what determines decisions about whether to drive a car and to what extent and the type of car (engine size, fuel type, etc). Also, there is no data available to assess the role of car cost compensation schemes in enterprises: again, there is data on a sectoral level, but nothing can be said on a more specific level.

Analysis that would complement the findings of this thesis could be about the effects on enterprises and specifically the changes induced by environmental taxes: has the economic structure changed, what kind of investments have been made and to what degree has the substitution of energy sources taken place? In addition, it is important to consider not only environmental taxes, but also other instruments: CO_2 quota trading, environmental regulations, etc. Another important issue is the scheme for petrol compensation by enterprises: it is unknown how much employers compensate employees for these costs and to what degree this also affects personal transportation and how responsive such compensation is to price changes. These aspects need to be clarified in order to make further decisions about how to direct a country towards energy and resource efficiency as proposed by different policy targets.

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ANNEX 1. FUEL AND ELECTRICITY EXCISE RATES IN ESTONIA 2000–2010

(in EEK)

	01.01 2000	**01.09 2000**	**01.04 2003**	**01.05 2004**	**01.01 2005**	**01.05 2005**	**01.01 2008**	**01.07 2009**	**01.01 2010**	**01.03 2010**
Petrol (per 1000 l)	3000	3500	3500	4500	4500	4500	5620	6228	6615	6615
Diesel (per 1000 l)	2320	3040	2550	3840	3840	3840	5165	5787	6148	6148
Light fuel oil (per 1 ton)	240	500	420	420	690	690	960	1056	1736	1736
Heavy fuel oil (per 1 ton)	–	–	–	200	235	235	235	235	235	235
Coal (per GJ of upper calorific value)	–	–	–	–	–	4.7	4.7	4.7	4.7	4.7
Natural gas (per 1000 m^3)	–	–	–	–	–	–	157	367	367	367
Electricity (per 1 MWh)	–	–	–	–	–	–	50	50	50	70

Source: Ministry of Finance 2013

ANNEX 2. ENVIRONMENTAL TAXES REVENUES IN ESTONIA

(thousand euros and % of total tax revenues)

	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
Fuel excise	71958	109119	116435	132456	168422	214098	238318	278228	300202	311625	357061	361418
Packaging excise	111	28	49	28	63	112	185	-89	73	30	10	197
Electricity excise									20411	21968	29311	33670
Heavy motor vehicle tax				3	3667	3553	3909	4197	4034	3544	3496	3680
Motor vehicle excise	8793	10163	11166	577	13	9	24	8	3	1	2	2
Pollution charge		10148	10999	18260	18534	23559	32115	41437	40480	39089	29273	32442
Resource charge		4130	4262	4805	5266	5120	9291	12316	13124	13317	17846	22607
Environmental taxes in total		133589	142913	156130	195965	246450	283842	336186	378327	389574	436999	454044
Environmental taxes % of total tax revenues		6.3	5.9	5.8	6.6	7.2	7.0	6.7	7.4	7.9	9.0	8.7

Source: Fuel, packaging and electricity excise, heavy motor vehicle tax and motor vehicle excise: Statistics Estonia (2013); pollution and resource charges: Ministry of the Environment (2011)

ANNEX 3. FUEL AND ENERGY PRODUCTS' EU MINIMUM AND AVERAGES RATES, ESTONIAN RATES AND RATES IN SELECTED EU MEMBER STATES

	Petrol (€ per 1000 L)	Diesel (€ per 1000 L)	Heavy fuel oil (€ per 1000 kg)	Coal and coke (€ per gigajoule)	Natural gas (€ per gigajoule)	Electricity (€ per MWh)
EU minimum rate	359	330	15	0.3	0.3	1
EU average rate	527.2	420.9	79.0	1.5	1.5	10.6
Highest rate in the EU	730.5 (Netherlands)	674.2 (UK)	485.8 (Sweden)	13.8 (Sweden)	9.5 (Sweden)	110 (Denmark)
Rate in Estonia	422.8	392.9	15	0.3	0.7	4.5
Rate in Germany	654.5	478.1	25	0.3	1.5	20.5
Rate in Denmark	587.3	443.6	399.4	10.2	9.4	110
Rate in Sweden	616.9	599.7	485.8	13.8	9.5	34.6
Rate in Finland	650.4	469.5	192.1	5.2	2.9	17
Rate in Poland	379.8	354.6	15.6	0.3	0	4.9
Rate in Latvia	407.5	336.1	15.8	0.3	0.5	1
Rate in Lithuania	434.4	330.2	15.1	0.3	0	1

Source: author's compilation based on European Commission (2013)

ANNEX 4. DIRECT DISTRIBUTIONAL EFFECT: PERCENTAGE OF EXPENDITURE AND INCOME

	2000	2001	2002	2003	2004	2005	2006	2007	2011
Per household member, % of expenditure									
I	0.5	0.6	0.7	0.6	1.1	1.3	0.9	0.7	0.7
II	0.5	0.5	0.4	0.4	0.6	0.9	0.6	0.5	0.8
III	0.6	0.5	0.5	0.7	0.6	0.8	0.6	0.8	1
IV	0.7	0.6	0.5	0.7	1	0.9	0.9	1.2	1
V	0.8	0.7	0.9	1.2	1	1.5	1.2	1	1.6
VI	0.9	0.9	1.1	1.1	1.2	1.3	1.3	1.7	1.5
VII	1	1.1	1.1	0.9	1.3	1.6	1.2	1.5	2
VIII	1.1	1.3	1.1	1.1	1.5	1.6	1.6	1.5	2
IX	1.4	1.5	1	1.3	1.7	1.7	1.8	1.7	2.1
X	0.9	1	1	1	1.4	1.1	1	1.4	2.3
Per household member, % of income									
I	0.9	0.9	1.2	1	1.9	2	1	0.8	0.7
II	0.6	0.6	0.5	0.5	0.6	1	0.6	0.5	0.7
III	0.7	0.5	0.5	0.7	0.6	0.8	0.6	0.7	1
IV	0.8	0.7	0.6	0.7	1	0.9	0.8	1.2	0.9
V	0.9	0.8	1	1.2	1	1.4	1.1	0.9	1.5
VI	0.9	0.9	1.1	1	1.1	1.2	1.2	1.4	1.4
VII	1	1.1	1.1	0.9	1.2	1.5	1.2	1.3	1.9
VIII	1.1	1.2	1.1	1.1	1.5	1.5	1.5	1.3	1.8
IX	1.4	1.4	0.9	1.2	1.5	1.5	1.6	1.5	1.9
X	0.8	0.9	0.8	0.9	1.2	0.9	0.7	0.9	1.9
Per household, % of expenditure									
I	0.3	0.5	0.5	0.4	0.8	0.9	0.6	0.5	0.5
II	0.5	0.4	0.4	0.4	0.4	0.8	0.4	0.5	0.8
III	0.6	0.5	0.6	0.6	0.9	0.8	0.8	0.7	0.9
IV	0.7	0.7	0.8	0.8	1.1	1.2	0.9	1.4	1
V	0.9	0.7	0.8	1.1	1.3	1.5	1.2	1.1	1.2
VI	1	1	0.9	1	1.4	1.4	1.4	1.4	1.5
VII	1.1	0.9	1.1	1.2	1.3	1.4	1.6	1.6	1.9
VIII	1.1	1.6	1.2	1.2	1.6	2	1.7	1.7	2.1
IX	1.2	1.3	1.2	1.4	1.6	2	1.7	1.7	2.5
X	0.9	1	1	1.1	1.4	1.1	1.4	1.4	2.6
Per household, % of income									
I	0.8	0.6	0.8	0.8	1	1.5	1	0.9	2
II	0.8	0.6	0.8	1.3	0.9	1.3	1	1	2
III	0.9	0.8	0.8	0.9	1	1.3	1.2	1	1.8
IV	0.6	0.7	0.7	0.9	0.9	1	1.2	0.9	1.3
V	0.8	0.9	0.9	1.1	1.1	1.4	1	1.2	1.6
VI	0.8	0.9	0.8	1	1.1	1.4	1.1	1.3	1.8
VII	0.9	1.1	0.9	1.1	1.3	1.6	1.5	1.2	2.2
VIII	1	1.1	0.9	1.1	1.5	1.5	1.2	1.4	1.9
IX	1	1.2	1.1	0.9	1.2	0.8	1.2	1.2	2
X	0.9	1	0.8	1	1.2	1.1	0.8	1.1	1.5

Source: author's calculations

ANNEX 5. DIRECT DISTRIBUTIONAL EFFECT, PERCENTAGE OF EXPENDITURE ACCORDING TO FUEL, PER HOUSEHOLD MEMBER, 2000, 2007 AND 2011

	Petrol	Diesel	District heating	Gas	Electricity
2000					
I	0.5	0.02	0.01		
II	0.5	0.02	0.02		
III	0.6	0.01	0.02		
IV	0.7	0.03	0.02		
V	0.8	0.06	0.02		
VI	0.9	0.07	0.02		
VII	1	0.04	0.01		
VIII	1.2	0.04	0.02		
IX	1.4	0.06	0.01		
X	1.2	0.06	0.01		
2007					
I	0.64	0.02	0.04		
II	0.43	0.03	0.04		
III	0.70	0.06	0.03		
IV	1.12	0.09	0.02		
V	0.93	0.07	0.03		
VI	1.51	0.12	0.02		
VII	1.34	0.13	0.02		
VIII	1.31	0.1	0.02		
IX	1.57	0.13	0.01		
X	1.22	0.1	0.01		
2011					
I	0.21	0.01	0.08	0.01	0.37
II	0.18	0.02	0.2	0.03	0.32
III	0.45	0.06	0.23	0.03	0.27
IV	0.44	0.04	0.21	0.03	0.25
V	0.9	0.21	0.2	0.04	0.25
VI	0.85	0.2	0.18	0.02	0.22
VII	1.32	0.31	0.14	0.01	0.19
VIII	1.34	0.33	0.1	0.02	0.2
IX	1.57	0.28	0.08	0.03	0.16
X	1.57	0.5	0.05	0.02	0.13

Source: author's calculations

ANNEX 6. ENVIRONMENTAL TAX SHARE ACCORDING TO SOCIO-DEMOGRAPHIC CHARACTERISTICS

	2000	2001	2002	2003	2004	2005	2006	2007	2011
Household type									
Household with one employed person	1.1	1.2	0.9	1.1	1.5	1.4	1.4	1.6	1.8
Household with two or more employed people	1.2	1.3	1.3	1.5	1.7	1.9	1.7	1.9	2.4
Household of unemployed people	0.7	0.7	0.6	0.7	0.9	1.3	0.8	0.8	1
Household of retired people	0.5	0.5	0.5	0.6	0.8	0.9	0.6	0.8	1.4
Household of other inactive people	0.5	0.5	0.6	0.6	1.5	0.8	0.3	0.6	0.8
Number of children									
No children in a household	1	1	0.9	1	1.4	1.4	1.3	1.5	1.7
1–2 children in a household	1.1	1.2	1.1	1.4	1.6	1.8	1.6	1.7	2.2
More than 3 children in a household	1.3	1.4	1.1	1.6	2.3	2	1.4	1.4	2.1
Settlement type									
Urban	0.9	0.9	0.8	1	1.2	1.3	1.2	1.3	1.5
Rural	1.6	1.6	1.6	1.7	2.1	2.1	1.9	2.1	2.7

Source: author’s calculations

ANNEX 7. SECTORAL AGGREGATION LEVEL USED IN INDIRECT DISTRIBUTIONAL ANALYSIS AND THE CORRESPONDENCE BETWEEN NACE REV 2 AND NACE REV 1.1

Sector	**Divisions of NACE rev. 2**	**Divisions of NACE rev. 1.1**[12]
Agriculture and fishery	01, 02, 03	01, 02, 05
Mining and quarrying	07, 08, v.a. 0892, 099	14
Manufacture of food	10, 11,12	15
Manufacture of textiles	13, 14, 15	17, 18
Manufacture of wood	16	20
Manufacture of paper	17, 18	21, 22
Manufacture of chemicals	20, 21	24
Manufacture of other non-metallic mineral products	23	26
Manufacture of basic metals	244, 2453, 2454	27
Manufacture of machinery	25, 26, 27, 28	28, 29, 30, 31, 32, 33
Manufacture of transport equipment	29, 30, 33	34, 35
Other manufacture	22, 31, 32	25, 36
Energy sector	05, 06, 0892, 091, 19, 35, 0721	
Construction	41, 42, 43	45
Land transport	49	
Water transport	50	
Air transport	51	
Services	36–39, 45–47, 52–56, 58–98	37

Source: Eurostat 2013c

[12] The sectoral codes are only for those used in the data about the electricity excise

ANNEX 8. ENVIRONMENTALLY RELATED TAXES PAID BY SECTORS, 2009

(thousand euros)

	Fuel excise	Resource charge	Pollution charge	Electricity excise	Total
Agriculture and forestry	8467.1	216.6	219.9	0	8903.6
Mining and quarrying	2207.2	1998.4	40.1	68.7	4314.3
Manufacture of food products	4626.5	285.1	502.2	923.9	6337.7
Manufacture of textiles	993.8	14.3	41.4	430.7	1480.3
Manufacture of wood	4935.4	16.9	73.6	1105.0	6130.9
Manufacture of paper	1569.7	192.5	622.5	686.7	3071.4
Manufacture of chemicals	832.3	0	31.1	730.4	1593.8
Manufacture of other non-metallic mineral products	2633.9	829.4	168.2	636.8	4268.3
Manufacture of basic metals	41.0	0	1.4	49.9	92.4
Manufacture of machinery	3473.6	4.4	60.8	886.5	4425.3
Manufacture of transport equipment	1430.8	0	22.5	193.5	1646.8
Other manufacturing	1481.1	2.1	86.1	674.2	2243.5
Energy sector	17795.0	20106.2	27757.2	0	65658.5
Construction	22493.6	1664.2	124.4	399.5	24681.7
Land transport	75198.4	3.9	36.3	0	75238.6
Water transport	2625.7	0	0	0	2625.7
Air transport	3335.4	0	0	0	3335.4
Services	55122.1	3549.6	7541.9	18.7	66232.3

Source: Ministry of the Environment (2011), Statistics Estonia (2013)

ANNEX 9. ENVIRONMENTALLY RELATED TAXES AS PERCENTAGE OF OUTPUT BY SECTORS, 2009

	Environmental taxes (million euros)	Output (million euros)	Environmental taxes share (%)	Environmental tax share considering intersectoral flows (%)
Agriculture and forestry	8.9	828.0	1.1	1.9
Mining and quarrying	4.3	185.0	2.3	3.2
Manufacture of food products	6.3	1030.1	0.6	1.7
Manufacture of textiles	1.5	561.5	0.3	0.6
Manufacture of wood	6.1	1028.1	0.6	1.8
Manufacture of paper	3.1	362.4	0.9	1.6
Manufacture of chemicals	1.6	308.9	0.5	1.0
Manufacture of other non-metallic mineral products	4.3	303.0	1.4	2.5
Manufacture of basic metals	0.1	745.0	0.01	0.4
Manufacture of machinery	4.4	1789.7	0.3	0.4
Manufacture of transport equipment	1.7	264.3	0.6	1.1
Other manufacturing	2.2	650.8	0.3	0.9
Energy sector	65.7	2072.2	3.2	4.4
Construction	24.7	676.5	3.7	4.6
Land transport	75.2	1066.8	7.1	7.6
Water transport	2.6	244.6	1.1	1.5
Air transport	3.3	137.5	2.4	2.8
Services	66.2	11967.0	0.6	1.3

Source: author's calculations based on Statistics Estonia 2012c and Ministry of the Environment (2011)

ANNEX 10. PRODUCT GROUPS' TAX LOAD ORIGINATING FROM ENVIRONMENTALLY RELATED TAXES AND CHARGES

COICOP code	Product group	Environmental tax load
0111	Bread and cereals	1.67
0112	Meat	1.67
0113	Fish and seafood	1.67
0114	Milk, cheese and eggs	1.70
0115	Oils and fats	1.67
0116	Fruit	1.88
0117	Vegetables	1.81
0118	Sugar, jam, honey, chocolate and confectionery	1.69
0119	Food products n.e.c.	1.67
0121	Coffee, tea and cocoa	1.67
0122	Mineral waters, soft drinks, fruit and vegetable juices	1.67
0211	Spirits	1.67
0212	Wine	1.67
0213	Beer	1.67
0221	Tobacco products	1.67
0311	Clothing materials	0.60
0312	Garments	0.60
0313	Other articles of clothing and clothing accessories	0.61
0314	Cleaning, repair and hire of clothing	1.19
0321	Shoes and other footwear	0.60
0322	Repair and hire of footwear	1.21
0411	Actual rentals paid by tenants	1.25
0431	Materials for the maintenance and repair of the dwelling	1.96
0432	Services for the maintenance and repair of the dwelling	4.45
0441	Water supply	0.66
0442	Refuse collection	1.25
0443	Sewage collection	1.25
0444	Other services relating to the dwelling n.e.c.	1.25
0451	Electricity	4.37
0452	Gas	4.37
0453	Liquid fuels	4.37
0454	Solid fuels	2.44
0455	Heat energy	4.37
0511	Furniture and furnishings	1.03
0512	Carpets and other floor coverings	0.66
0513	Repair of furniture, furnishings and floor coverings	1.25
0521	Household textiles	0.63
0531	Major household appliances whether electric or not	0.43
0532	Small electric household appliances	0.43
0533	Repair of household appliances	1.25
0541	Glassware, tableware and household utensils	0.97
0551	Major tools and equipment	0.44

COICOP code	Product group	Environmental tax load
0552	Small tools and miscellaneous accessories	0.43
0561	Non-durable household goods	1.07
0562	Domestic services and household services	1.25
0611	Medical products, appliances and equipment	1.04
0612	Other medical products	1.03
0613	Therapeutic appliances and equipment	0.88
0621	Medical services	1.25
0622	Dental services	1.25
0623	Paramedical services	1.25
0631	Hospital services	1.25
0711	Motor cars	1.12
0712	Motor cycles	1.12
0713	Bicycles	1.12
0721	Spare parts and accessories for personal transport equipment	1.01
0722	Fuels and lubricants for personal transport equipment	4.34
0723	Maintenance and repair of personal transport equipment	1.25
0724	Other services in respect of personal transport equipment	1.25
0731	Passenger transport by railway	7.55
0732	Passenger transport by road	7.55
0733	Passenger transport by air	2.82
0734	Passenger transport by sea and inland waterway	1.49
0736	Other purchased transport services	7.55
0811	Postal services	1.25
0821	Telephone and telefax equipment	0.43
0831	Telephone and telefax services	1.25
0911	Equipment for the reception, recording and reproduction of sound and picture	0.43
0912	Photographic and cinematographic equipment and optical instruments	0.43
0913	Information processing equipment	0.43
0914	Recording media	1.15
0915	Repair of audio-visual, photographic and information processing equipment	1.25
0921	Major durables for outdoor recreation	0.88
0922	Musical instruments and major durables for indoor recreation	0.88
0923	Maintenance and repair of other major durables for recreation and culture	1.25
0931	Games, toys and hobbies	0.49
0932	Equipment for sport, camping and open-air recreation	0.82
0933	Gardens, plants and flowers	1.81
0934	Pets and related products	1.61
0935	Veterinary and other services for pets	1.25
0941	Recreational and sporting services	1.25
0942	Cultural services	1.25
0943	Games of chance	1.25
0951	Books	1.25

COICOP code	Product group	Environmental tax load
0952	Newspapers and periodicals	1.25
0953	Miscellaneous printed matter	1.25
0954	Stationery and drawing materials	1.36
0961	Package holidays	1.25
1011	Pre-primary and primary education	1.25
1021	Secondary education	1.25
1041	Tertiary education	1.25
0151	Education not definable by level	1.25
1111	Restaurants, cafes and the like	1.25
1112	Canteens	1.25
1121	Accommodation services	1.25
1211	Hairdressing salons and personal grooming establishments	1.25
1212	Electric appliances for personal care	0.43
1213	Other appliances, articles and products for personal care	1.16
1231	Jewellery, clocks and watches	0.79
1232	Other personal effects	0.77
1241	Social protection	1.25
1252	Insurance connected with the dwelling	1.25
1253	Insurance connected with health	1.25
1254	Insurance connected with transport	1.25
1255	Other insurance	1.25
1262	Other financial services n.e.c.	1.25
1271	Other services n.e.c.	1.25

Source: author's calculations based on CPA/COICOP transition tables and environmental tax loads of sectors

ANNEX 11. RATIOS OF INCOME AND EXPENDITURE BY INCOME DECILES, 2000–2007

	2000	2001	2002	2003	2004	2005	2006	2007	Average
I	0.89	0.9	0.89	0.92	1.01	0.99	1.06	1.06	0.97
II	0.95	0.94	0.98	1.01	1.06	1.08	1.08	1.1	1.03
III	0.95	0.98	1.01	1.01	1.08	1.08	1.14	1.15	1.05
IV	1	1.02	1.04	1.09	1.13	1.14	1.15	1.18	1.09
V	1	1.02	1.03	1.02	1.06	1.13	1.17	1.21	1.08
VI	1.04	1.02	1.09	1.08	1.12	1.17	1.18	1.24	1.12
VII	1.04	1.06	1.11	1.09	1.13	1.14	1.16	1.19	1.12
VIII	1.04	1.1	1.06	1.09	1.08	1.09	1.13	1.25	1.10
IX	1.1	1.11	1.1	1.12	1.12	1.18	1.26	1.22	1.51
X	1.23	1.25	1.41	1.23	1.29	1.31	1.51	1.56	1.35

Source: author's calculations based on HBS 2000–2007 (Statistics Estonia)

ANNEX 12. INDIRECT DISTRIBUTIONAL EFFECT: PERCENTAGE OF EXPENDITURES AND INCOME PER HOUSEHOLD, 2011

	Indirect tax load, % of income	Indirect tax load, % of expenditure	Direct tax load, % of expenditure	Total tax load, % of expenditure
I	1.81	1.79	0.7	2.5
II	1.69	1.77	0.7	2.5
III	1.63	1.77	1	2.8
IV	1.58	1.72	0.9	2.7
V	1.59	1.72	1.5	3.3
VI	1.56	1.73	1.4	3.1
VII	1.57	1.75	1.9	3.6
VIII	1.56	1.76	1.8	3.5
IX	1.4	1.69	1.9	3.6
X	1.27	1.68	1.9	3.6

Source: author’s calculations

ANNEX 13. SOCIO-DEMOGRAPHIC CHARACTERISTICS OF HOUSEHOLDS ANALYSED

Relevant socio-demographic characteristics according to other studies	**Corresponding socio-demographic data in HBS**	**Transformations made by the author**
Income	Income up to 2007 –> EEK, from 2011 –> EUR	Transformation to same currency (1 EUR = 15.6466 EEK)
Expenditure	Expenditures up to 2007 –> EEK, from 2011 –> EUR	Transformation to same currency (1 EUR = 15.6466 EEK)
Settlement type	Settlement type (up to 2005): 1. a large town (more than 50 000 inhabitants) 2. a county town (other than (1)) 3. a town or township other than (1) or (2) 4. a large village 5. a village Settlement type (2006–...): 3. small town 4. town 5. township 6. town district 7. borough 8. village	A dummy variable urban has been composed (0: rural; 1: urban) (up to 2005): 1–3 –> urban 4–5 –> rural (2006–...) 3–6 –> urban 7–8 –> rural
Age	Age of household head	
Gender	Gender of household head 1 – male; 2 – female	0 – male; 1 – female
Race	Not used in the thesis	
Education	Education of household head 1. primary education 2. secondary education 3. higher education 9. not known	
Employment	Household social group: 11 one working member 12 two working members 20 unemployed 30 retired 40 other inactive	
Number of children	Number of members of a household less than 16 years old	
Size of a household	Number of members of a household	
Number of adults		Number of adults = (size of a household) – (number of kids)

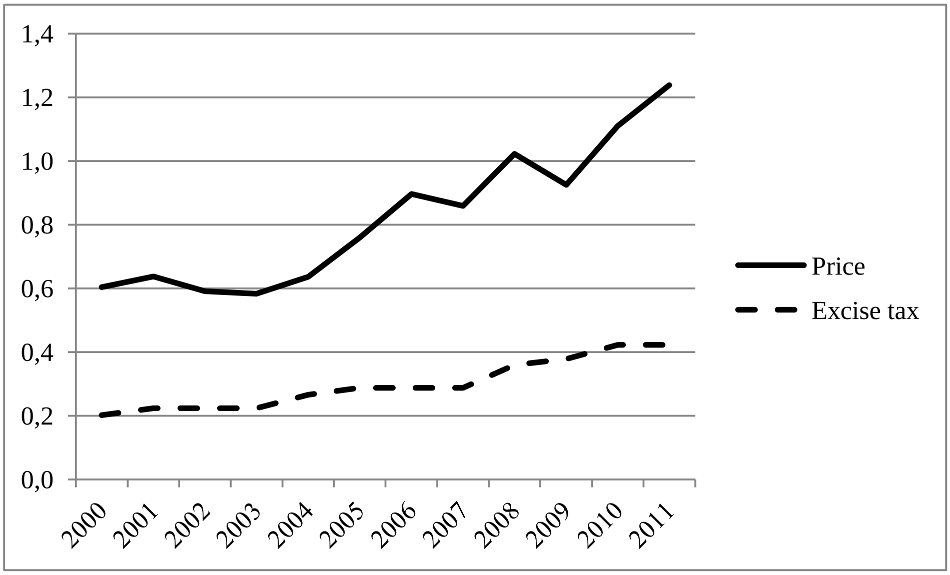
ANNEX 14. DESCRIPTIVE STATISTICS OF USED DATA

<table>
<tr><th></th><th>Median</th><th>Mean</th><th>Standard deviation</th></tr>
<tr><td colspan="4">Continuous and discrete variables</td></tr>
<tr><td>Expenditures (monthly, euros)</td><td>330.1</td><td>435.7</td><td>375.57</td></tr>
<tr><td>Log(expenditure)</td><td>5.8</td><td>5.8</td><td>0.78</td></tr>
<tr><td>Income (monthly, euros)</td><td>371.1</td><td>490.0</td><td>399.98</td></tr>
<tr><td>Log(income)</td><td>5.9</td><td>5.9</td><td>0.79</td></tr>
<tr><td>Petrol quantity (monthly, litres consumed)</td><td>0</td><td>21.2</td><td>41.96</td></tr>
<tr><td>Petrol quantity (for positive amounts)</td><td>48.0</td><td>60.9</td><td>51.30</td></tr>
<tr><td>Log (petrol quantity)</td><td>3.9</td><td>3.8</td><td>0.89</td></tr>
<tr><td>Age of household head</td><td>51</td><td>51.6</td><td>15.81</td></tr>
<tr><td>Number of adults in a household</td><td>2</td><td>2.2</td><td>0.98</td></tr>
<tr><td>Number of children in a household</td><td>0</td><td>0.5</td><td>0.89</td></tr>
<tr><td>Number of cars owned or used by a household</td><td>0</td><td>0.6</td><td>0.62</td></tr>
<tr><td colspan="4">Binary and categorical variables</td></tr>
<tr><td>Gender of household head (male)</td><td colspan="3">Male: 51%
Female: 49%</td></tr>
<tr><td>Settlement type of a household</td><td colspan="3">Urban: 61%
Rural: 39%</td></tr>
<tr><td>Education level of household head</td><td colspan="3">Primary education: 7%
Secondary education: 34%
Higher education: 59%
Not known: <1%</td></tr>
<tr><td>Household type</td><td colspan="3">One working member: 33%
Two or more working members: 35%
Household of unemployed people: 5%
Household of retired people: 23%
Household of other inactive people: 4%</td></tr>
</table>

Source: author’s calculations based on HBS 2000–2007 and 2011 (Statistics Estonia)

ANNEX 15. PETROL PRICE AND EXCISE TAX SHARE IN PETROL PRICE IN ESTONIA 2000–2011

(per litre, annual averages)



Source: compiled by the author based on European Commission (2013b) and Ministry of Finance (2013)

ANNEX 16. RESULTS OF THE TWO-PART MODEL

	Probit (marginal effect)	OLS for positive quantity
Ln (expenditures)	0.284***	0.523***
Ln (price of petrol)		–0.564***
Number of adults	0.009***	–0.040***
Number of children	–0.021***	–0.041***
Gender of household head (0–male; 1–female)	–0.123***	–0.105***
Age of household head	–0.004***	–0.011***
Age of household head^2	–0.0001***	0.0001**
Education level (comparison level: elementary or lower)		
Secondary education	*0.043****	
Higher education	*0.037***	
Not known	*–0.056*	
Urban/rural	–0.206***	–0.103***
Household type (comparison level: one working member)		
Two or more working members	0.020***	0.033**
Unemployed	–0.027**	–0.080*
Retired	0.003	–0.112***
Other	–0.041***	–0.082
Ownership of car (number)		0.438***
Yearly dummies (comparison level: 2000)		
2001	–0.020**	–0.007
2002	–0.040***	–0.036
2003	–0.030***	0.029
2004	–0.035***	0.069**
2005	–0.016	0.080**
2006	–0.037***	0.109**
2007	–0.056***	0.123***
2011	–0.230***	0.293***

*** significant at 0.01 level; ** significant at 0.05 level; * significant at 0.1 level
Probit model: dependent variable: consumes petrol or not (0/1); N=37148; Pseudo R^2=0.207; correctly classified: 73.5%
OLS: dependent variable: ln(consumed petrol); N=12957; Pseudo R^2=0.294

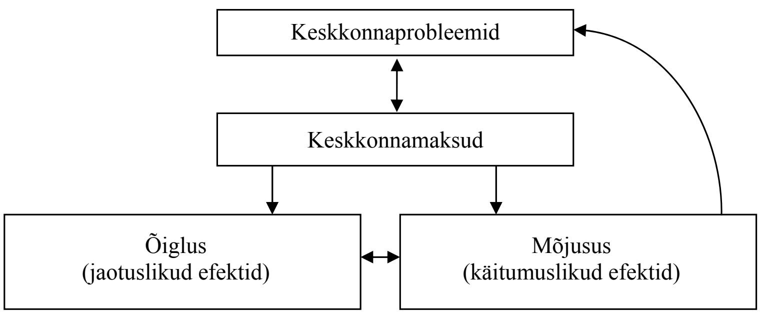
SUMMARY IN ESTONIAN

Keskkonnamaksude jaotuslikud ja käitumuslikud efektid Eesti näitel

Töö aktuaalsus ja motivatsioon

Keskkonnamakse kasutatakse keskkonnapoliitikas juba aastakümneid, kuid nende majanduslikud ja sotsiaalsed mõjud on endiselt ebaselged, kuna erinevad autorid jõuavad erinevatele tulemustele. Kuna keskkonnamaksude teoreetiline põhjendus tuleneb välismõjudest, siis on nende kehtestamise eesmärk muuta inimeste või ettevõtete käitumist hinnasignaali kaudu. Seega peaks keskkonnamaksude tagajärjel vähenema maksustatud hüvise tarbimine või tootmine ning keskkonnaseisund paranema. Samas on tegu aga ka maksumeetmega, mis muudab tulujaotust ning seega ka ühiskonnas valitsevat ebavõrdust.

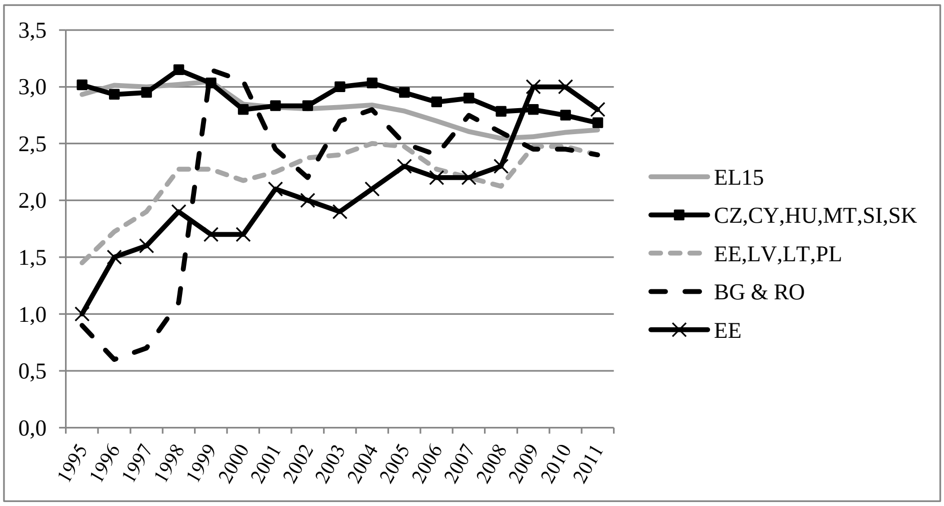
Tihtipeale väidetakse, et keskkonnamaksu koormust kannab enam vaesem elanikkond ning see saab takistuseks keskkonnamaksude kehtestamisele. Keskkonnamaksude jaotuslikku efekti võib võimendada vaesemate inimeste piiratud võimalused maksustatud hüvise tarbimist vähendada, kuna selleks tuleb teha suuri investeeringuid: näiteks kodu soojustamine või energiaefektiivsema auto ost. Seega on jaotuslik ja käitumuslik efekt omavahel seotud. Käesolevas töös keskendutaksegi nimetatud kahele keskkonnamaksudega kaasnevale efektile: jaotuslik efekt (mis seondub õiglusega) ning käitumuslik efekt (mille kaudu hinnatakse keskkonnamaksude mõjusust). Vastavad seosed on välja toodud ka joonisel I. Kuigi jaotuslikku efekti võib käsitleda sotsiaalvaldkonna teemana, tunnustatakse üha rohkem ka selle seost majandusega. Näiteks on leitud, et üha kasvava sissetulekute ebavõrdsuse tõttu vähenevad ka säästmismäärad (Levine et al 2010).



Joonis I. Seosed keskkonnaprobleemide, keskkonnamaksude ning nende efektide vahel (autori joonis)

Antud töö fookuseks on Eesti kui näide riigist, mis on viimastel aastakümnetel kogenud kiireid majanduslikke ja sotsiaalpoliitilisi muutusi. Kuigi keskendutakse vaid ühele riigile, võib sellest kasulikke järeldusi teha ka teiste riikide jaoks. Seoses Euroopa Liiduga liitumisega on tulnud Eestil kehtestada oluliselt karmimad keskkonnanõuded ning seetõttu on ka keskkonnamaksude määrad üsna kiiresti tõusnud. Kiire majanduskasvu tõttu viimasel kahekümnel aastal on oluliselt muutunud ka tarbimismustrid. Käesolevas töös läbiviidud analüüs võimaldab välja tuua, millised majapidamiste rühmi keskkonnamaksude tõus enim puudutab ning kuidas ebasoovitavaid efekte vältida.

Võrreldes peamise tarbimismaksu, käibemaksuga, ei ole Eesti keskkonnamaksude tase väga kõrge, kuid viimasel aastakümnel on see oluliselt tõusnud: kui 2001. a moodustasid keskkonnamaksud 6% kogumaksulaekumisest, siis 2011. a oli see 9%. Ei saa öelda, et Euroopas valitseks ühtne trend keskkonnamaksude laekumiste osakaalus, kuid jooniselt II võib näha, et vanades liikmesriikides ning ka Kesk- ja Lõuna-Euroopas asuvates uutes liikmesriikides keskkonnamaksude suhteline osatähtsus pigem väheneb. Balti riikides ja Poolas on keskkonnamaksude osakaal SKP-s suurenenud. Seega on Eestis, Lätis, Leedus ja Poolas keskkonnamaksude tulu trend olnud sarnane, kuigi kasutatavad maksud ja nende tasemed on erinevad.



Joonis II. Keskkonnamaksude osakaal SKP-s Euroopa Liidu riikides, 2000–2011 (autori joonis Eurostat 2013a andmete põhjal)

Nii Euroopa Liidu kui ka Eesti tasandil on mitmeid poliitilisi dokumente, milles rõhutatakse säästva arengu olulisust ning ressursi- ja energiaefektiivse majanduse poole liikumise vajadust, näiteks Euroopa 2020, Eesti säästva arengu strateegia Säästev Eesti 21, jne. Samal ajal propageeritakse nii teaduskirjanduses kui ka reaalses poliitikas ökoloogilist maksureformi, mille põhimõtteks on suurendada keskkonnakahjuliku tegevuse maksustamist ning vähendada tööjõu maksustamist. Seetõttu suureneb tulevikus keskkonnamaksude roll maksupoliitikas ning oluline on analüüsida kaasnevaid efekte.

Seni on keskkonnamaksude erinevaid mõjusid uuritud eelkõige arenenud riikides, kus nii sissetulekute tase kui ka maksusüsteem on suhteliselt stabiilne. Samuti on jaotuslikke efekte enamasti hinnatud hüpoteetiliste maksude puhul. Viimastel aastatel on tähelepanu pälvinud ka bensiini maksustamise jaotuslikud efektid ning 2012. a ilmus sellele teemale pühendatud raamat (Sterner 2012). Siiski on ka selles raamatus keskendutud pigem arenenud riikidele ning arengumaadele ning mitte riikidele, kes jäävad nende kahe kategooria vahele. Erandiks on vaid peatükk Tšehhi mootoriküttuse aktsiisi jaotuslike efektide kohta (Ščasny 2012). Senised empiirilised tööd on enamasti keskendunud ühele kahest teemast: kas jaotuslikele või käitumuslikele efektidele. Kui keskendutakse jaotuslikele küsimustele, siis on võimalused käitumuslike efektide uurimiseks piiratud (näiteks valides välja mõned majapidamiste tüübid ja kajastades üksnes neid). Teisalt aga on käitumuslikud efektid väga mahukas uurimisteema, kus kasutatakse väga erinevaid meetodeid, sõltuvalt andmetest, uurija huvist ja taustast ning uurimuse fookusest. Käitumuslike efektide hindamiseks kasutavad majandusteadlased tihtipeale elastsusi, kuid tulemused on väga varieeruvad: mõnede tööde järgi on bensiin väga madala hinnaelastsusega, kuid teiste järgi väga kõrgega. Samuti on väga erinevad sissetulekuelastsuste hinnangud. Seega puudub ühtne arusaam, kas bensiini maksustamine toob kaasa ka tarbimise muutuse ning missuguseid sissetulekurühmi see enam mõjutab.

Käesoleva töö uudsus tulenebki nimetatud kahest aspektist: esiteks on ta tagasivaatav analüüs keskkonnamaksude rakendamisele kiiresti muutuvates majandus- ja tarbimistingimustes, mis seni teaduskirjanduses erilist tähelepanu pälvinud ei ole. Eesti oma väiksuses on ka unikaalne, kuna võimaldab uurimise alla võtta kõik riigis rakendatud keskkonnamaksud. Teiseks antud töö uudseks momendiks on, et käsitletakse nii jaotuslikke kui ka käitumuslikke efekte ning nendevahelisi seoseid.

Eesmärk ja uurimisküsimused

Antud töö eesmärk on välja selgitada keskkonnamaksude jaotuslikud efektid ja nende võimalik seos käitumuslike efektidega Eesti näitel. Kuigi keskkonnamaksudel on oluline mõju ka ettevõtetele, keskendub antud töö majapidamistele, kuna tarbija valikud mõjutavad omakorda ka ettevõtteid.

Töö kolm peamist uurimisküsimust on järgmised:

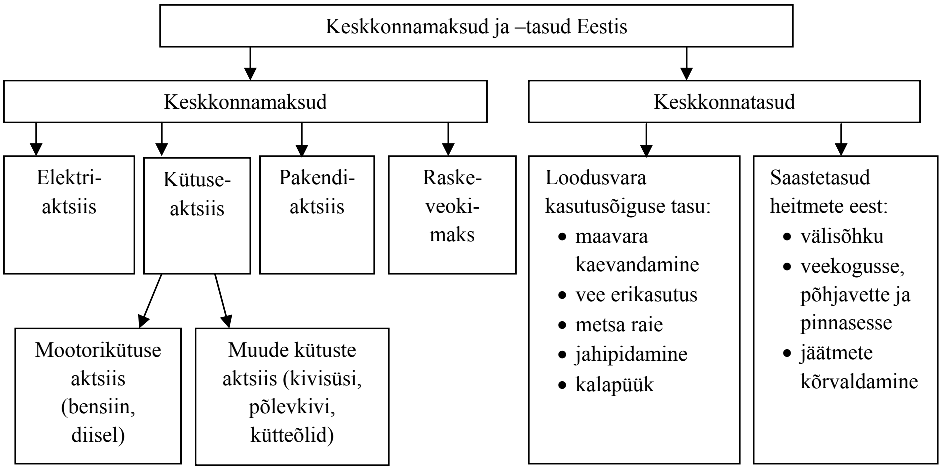
1. Millised on Eesti keskkonnamaksude otsesed jaotuslikud efektid?
2. Millised on Eesti keskkonnamaksude kaudsed jaotuslikud efektid?
3. Kuidas on majapidamised tarbimise muutuste kaudu reageerinud Eesti keskkonnamaksudele ning kuidas on see seotud erinevate sotsiaaldemograafiliste teguritega?

Seega keskendub töö kaht liiki efektidele: esmalt jaotuslikele, mille saab omakorda jagada otseseks ja kaudseks efektiks. Otsene jaotuslik efekt tuleneb maksustatava toote tarbimisest, näiteks bensiini ostust tulenev kütuseaktsiis. Kaudne efekt tuleneb maksustatud toote tekitatud hinnamuutustest, näiteks kütuseaktsiisi osa toidukaupade hinnas. Teisalt, kuna keskkonnamaksude eesmärgiks on leevendada teatud keskkonnaprobleemi, on oluline hinnata ka seda, kas need maksud on toonud kaasa ka tarbimise muutumist, ehk n-ö käitumuslikku efekti. Nagu eelpoolgi toodud, on need efektid omavahel seotud.

Kui keskkonnamaksude jaotuslike efektide kohta on teaduskirjanduses artikleid ilmunud (hea ülevaade on näiteks toodud Ekins et al 2011, bensiinimaksude jaotuslike efektide kohta Sterner 2012), ning selles valdkonnas on üsna selge ettekujutus, kuidas seda mõõta võiks, siis käitumuslikke efekte on tunduvalt keerulisem hinnata. Mõnedes riikides viiakse läbi spetsiaalseid keskkonnateemalisi küsitlusi, kuid Eestis pole seda kahjuks tehtud, kuna see on aja- ja ressursimahukas ning pole tegu olnud riigi jaoks prioriteetse valdkonnaga. Seega kasutatakse antud töös maksustatud toodete hinna- ja sissetulekuelastsust, mis arvutatakse leibkondade üldise tarbimise eelarve andmetest.

Uurimisobjekt, andmed ja metoodika

Keskkonnamaksude olemus on aastakümnete jooksul muutunud: traditsiooniliselt on olnud maksubaasiks spetsiifiline saasteaine või ressurss ning maksu eesmärgiks konkreetset heidet või ressursikasutust vähendada. Tänasel päeval kasutatakse keskkonnamakse üha enam ka fiskaalsetel eesmärkidel ning keskkonnamaksu objektiks on tihti mingi saasteaine lähend. Laialtlevinud keskkonnamaksu definitsiooni järgi ongi keskkonnamaks maks, mille maksubaasiks on tõestatud ja spetsiifilise negatiivse keskkonnamõju füüsiline ühik või selle lähend (Eurostat 2001). Seega kuuluvad ka näiteks energiamaksud (kütuseaktsiis) keskkonnamaksude alla, kuna kütusekasutus on seotud CO_2 heitmetega. Käesolev töö keskendub Eestis rakendatud keskkonnamaksudele ja –tasudele: elektri- ja kütuseaktsiis ning keskkonnatasud (joonis III). Antud töö keskendub mikrotasandile ehk majapidamistele, ning ei kajasta makromajanduslikke efekte, nagu näiteks mõjusid riigieelarvele, halduskoormusele, tulude kasutuse efektiivsusele, jms.



Joonis III. Keskkonnaga seotud maksud ja tasud Eestis (autori joonis)

Eesmärgist lähtuvalt võib keskkonnamaksud jagada kulusid katvateks, fiskaalseteks ja stimuleerivateks maksudeks. Kui kulukatva maksu eesmärgiks on katta teatud teenustega kaasnevaid kulusid (nt jäätmekäitlus, heitvee puhastus), siis fiskaalne maks ei pruugi olla seotud konkreetse teenusega ja selle eesmärgiks on koguda riigieelarvesse raha. Stimuleeriv maks on aga kavandatud sellisena, et muutuks maksustatavate käitumine ning maksustatud toote või teenuse tarbimine väheneks.

Keskkonnamakse peetakse regressiivseteks maksudeks, mis tähendab, et proportsionaalselt langeb maksukoormus enam vaesematele leibkondadele. Jaotusliku efekti hindamine on oma olemuselt staatiline, kuna kajastatakse maksukoormuse jaotumist vaid ühel konkreetsel ajahetkel ning seda tehakse ristandmete põhjal. Jaotusliku efekti hindamiseks kasutatakse antud töös selliseid näitajaid nagu Kakwani indeks, Reynolds-Smolensky indeks ja Atkinsoni indeks. Jaotusliku efekti puhul kasutatakse mikrosimulatsiooni meetodit, mis võimaldab simuleerida erinevaid poliitikaefekte majandusagentide peal. Mikrosimulatsiooni eeliseks ongi, et saab arvesse võtta kõiki andmebaasis olevaid vaatlusi ning ei pea välja valima teatud tüüpilisi agente nagu tehakse makrotasandi analüüsides. Käesoleva töö põhiandmeallikaks on Eesti Leibkonna Eelarve Uuring (LEU), mille andmed pärinevad aastatest 2000–2007 ning 2010–2011 (kokku 50320 vaatlust). Paraku ei kogutud aastatel 2008–2009 LEU andmeid ning andmete kogumise metoodikat on mõnevõrra muudetud alates 2010. aastast. Käesolevas töös eristatakse jaotusliku efekti puhul otsest ja kaudset efekti. Otsene efekt tuleneb maksustatud toote tarbimisest, kaudne efekt tuleneb hinnamuutusest, mida näiteks energia maksustamine kaasa toob. Kaudse keskkonnamaksude maksukoormuse hindamisel on kasutatud lisaks LEU-le majanduse sisend-väljundtabelit, kütuse ja elektritarbimise kasutust sektorite kaupa ning ettevõtete poolt makstavaid keskkonnatasude summasid.

Probleemiks osutus erinevate klassifikaatorite kasutamine erinevates tabelites ning sisend-väljundtabeli kõrge agregeerituse tase. Kui kütuseaktsiisi puhul saab hinnata nii otsest kui kaudset jaotuslikku efekti, siis keskkonnatasude puhul saab rääkida vaid kaudsest jaotuslikust efektist, kuna tasud ei ole seotud konkreetse tarbitava ühikuga.

Käitumuslikke efekte hinnatakse antud töös hinna- ja sissetulekuelastsuse kaudu. Käitumuslikku efekti on töös analüüsitud vaid bensiini tarbimise puhul, kuna näiteks elektriaktsiisi ja mitmete kodude kütmiseks kasutatavate kütuste aktsiisi kehtestamine või oluline tõus toimus 2008. aastal, seega on aegrida inimeste käitumise hindamiseks liiga lühike. Kuigi teatud osa leibkondi küsitletakse järjestikusel kahel aastal, ei pruugi nad täita uuringu kõiki osi ning kuna paljud küsitletud leibkonnad pole küsitlusperioodil kulutusi bensiinile teinud, ei saa paneelandmete analüüsi kasutada. Elastsuse hindamiseks kasutatakse kaheosalist mudelit: probit-mudel (hindamaks tõenäosust, kas leibkond üldse bensiinile kulutab) ning lõigatud muutuja mudel positiivsete väärtuste jaoks. Kuigi töö lõppjärelduste tegemisel tugines autor kaheosalise mudeli tulemustele, hinnati tulemuste stabiilsuse kindlustamiseks töö käigus erinevaid mudeleid ning järeldused on sarnased. Modelleeritava protsessi heterogeensusest tulenevalt viidi läbi ka kvantiilregressioon, mis võimaldab muutujatevahelist suhet näidata erinevates punktides (kvantiilides). Sissetuleku- ja hinnaelastsuse puhul tuleb arvestada, et need on arvutatud muutujate keskväärtuste jaoks.

Kuigi kaudselt on antud töö üheks ajendiks säästva arengu põhimõtted, s.t. majandus-, sotsiaal- ja keskkonnavaldkonna sidusus ja kooskõlaline arendamine, ei kajastata antud töös seda, kas rakendatud keskkonnamaksude süsteem ja tase on optimaalne. Samuti käsitletakse antud töös vaid majapidamistele avalduvaid efekte, kuna majapidamised tarbivad suurema osa maksustatavatest kütustest. Kuigi keskkonnamaksud mõjutavad oluliselt ka ettevõtteid, piirdutakse käesolevas töös vaid keskkonnamaksude osakaalu hindamisega sektorite käibes, millest tuleneb majapidamiste kaudne maksukoormus. Ka siinkohal on tegu staatilise efektiga, käitumuslikku efekti (näiteks, kuivõrd on keskkonnamaksud toonud kaasa ettevõtete investeeringud puhtamasse tehnoloogiasse, kasutatava kütuse väljavahetamise või tootmise kolimise mujale riiki) ettevõtete puhul hinnatud ei ole.

Teoreetiline taust ja varasemad empiirilised uuringud

Keskkonnamaksude teoreetiliseks põhjenduseks on turutõrked, eelkõige välismõjud. Välismõjuks nimetatakse olukorda, kus üksikisik või firma mõjutab oma tegevusega teist üksikisikut või firmat, kellele seda ei kompenseerita (negatiivse välismõju puhul) ning välismõju tekitaja arvestab vaid erapiirkuludega. Üheks lahenduseks, kuidas välismõju tekitaja arvestaks ka sotsiaalseid piirkulusid, on Pigou maks, mis on nime saanud majandusteadlase A.C. Pigou järgi. Maksu suurus peaks võrduma välismõju piirkuluga, kuid tegu on n-ö idealiseeritud (*first-best*) meetmega, mille järgi peaks iga keskkonnakahju tekitajat maksustama vastavalt tekitatud kahjule, mis omakorda sõltub saasteainest, selle kogusest, asukohast, mõjutatud inimeste arvust, jne.

Bovenberg and Mooij (1994) näitavad, et keskkonnamaksu tase sõltub olemasolevatest maksudest, eelkõige tööjõu maksudest ja sellest, kuivõrd tööhõive reageerib maksumuudatustele. Levinud on ka näiteks Ramsey maksustamispõhimõte, mille järgi tuleb maksustada väheelastse nõudlusega kaupu. See põhimõte ühtib hästi ka hea maksu kriteeriumiga, mille järgi maksu peaks olema lihtne administreerida ning ta ei tohiks takistada ressursside efektiivset paigutust. Majandusteaduses rakendatakse ressursside optimaalse paigutuse hindamiseks Pareto kriteeriumi, mille kohaselt pole võimalik mõne agendi olukorda paremaks teha ilma et keegi teine kaotaks. Paraku on tegelikkuses selliseid poliitilisi otsuseid väga raske langetada, kus mitte keegi ei kaota. Seetõttu räägitakse pigem potentsiaalsest Pareto parendusest (Kaldor-Hicks kriteerium): parendus leiab aset ka siis, kui võitjad saavad kaotajatele kaotuse kompenseerida, isegi kui seda reaalsuses ei tehta. Seega, kui kasude summa on suurem kaotuste summast, on tegu soovitava parendusega. Mitmed autorid on selgitanud antud lähenemisega kaasnevaid probleeme, ning eelkõige seonduvad need tulujaotuse ja õigluse küsimustega.

Tulujaotuse teooriatel on pikk ajalugu ja erinevad lähenemised võib jagada positivistlikeks ja normatiivseteks. Klassikaline majandusteooria tegeles funktsionaalse tulujaotusega ehk kuidas jaotus tulu põhiliste tootmistegurite (maa, tööjõud ja kapital) vahel. Umbes sajand hiljem mõisteti, et suur osa ebavõrdsusest tuleneneb palgatulude erinevusest ning hakati uurima palga ja hariduse vahelisi seoseid. Kui positivistlik lähenemine kirjeldab seoseid, siis tulujaotuse küsimusele lähenetakse tihtipeale ka normatiivsest aspektist, kuna need baseeruvad teatud väärtushinnangutel. Normatiivsete teooriate ühine seisukoht on, et ebavõrdne tulujaotus on raiskav, kuna sellest tulenev sotsiaalne heaolu on madalam kui see oleks ühtlase jaotuse korral. Antud töös uuritakse keskkonnamaksude ja tulujaotuse vahelisi seoseid. Tööd võib pidada pigem normatiivseks lähenemiseks, kuna lähtutakse seisukohast, et keskkonnamaksudest tulenev koormus ei tohiks enam langeda vaesematele sissetulekurühmadele, mis suurendaks ühiskonnas valitsevat ebavõrdsust.

Tulujaotuse teema puhul tuleb käsitleda ka sotsiaalse heaolu funktsiooni, mis näitab, kuidas on ühiskonna heaolu seotud individuaalse heaoluga. Näiteks utilitaristliku käsitluse järgi tuleb maksimeerida individuaalsete kasulikkuste summat, mis ühtib ka Kaldor-Hicks'i kriteeriumi ehk potentsiaalse Pareto kriteeriumigaga. Selle järgi on parendus soovitav, kui potentsiaalsed võitjad saaksid potentsiaalsetele kaotajatele selle hüvitada, isegi kui seda tegelikkuses ei toimu. Kuigi seda printsiipi on palju kritiseeritud, rakendatakse poliitikaotsuste puhul tavaliselt siiski summaarse tulu ja kulu kriteeriumi. Teistsugune ja tuntud lähenemine pärineb Rawls'ilt (1971), kelle järgi heaolu maksimeerimine tähendab vaeseima inimese heaolu maksimeerimist.

Läbi ajaloo on valitsenud ka erinevad arusaamad kasulikkusfunktsioonist. Kui siinkohal rääkida hüvisest, mille tootmise või tarbimisega kaasnevad välismõjud, siis selle hüvise tarbija kasulikkusele tuleneb individuaalsest tarbimisest lähtuvalt positiivne mõju. Samas on kogutarbimisest tulenev efekt negatiivne, kuna sellega seonduvad välismõjud. Kui konkreetse hüvise individuaalset tarbimise kogust saab inimene valida, siis näiteks keskkonnakvaliteedi taset (ehk kogutarbimisega kaasnevat välismõju) individuaalselt valida ei saa, kuna see sõltub ka teiste tarbimisest.

Seega on keskkonnamaksudel kahetine eesmärk: ühelt poolt vähendada seonduvat keskkonnaprobleemi ehk adresseerida välismõjusid. Teisalt on tegu maksumeetmega, mis peaks vastama ka hea maksu nõuetele ehk väikeste kuludega tekitama võimalikult suurt riigieelarve tulu. Samuti mõjutab iga maks ühiskonna tulujaotust. Empiirikas on tehtud mitmeid uurimusi, kellele keskkonnamaksudest tulenev koormus enim langeb, et välja selgitada mõju tulujaotusele. Varasemad uuringud kinnitavad kütteainena kasutatava energia ja elektri maksustamise regressiivsust ehk vaesemaid majapidamisi enam mõjutavaks (Barker and Köhler 1998, Jacobsen et al 2003, Bork 2006, Dresner and Ekins 2006, Callan et al 2009). Sõidukite maksustamine on pigem progressiivne (Jacobsen et al 2003). Mootorikütuse maksustamise puhul on aga erinevaid tulemusi saadud: progressiivsena on seda näidanud Jacobsen et al (2003) ja Tiezzi (2005), enim keskmistele sissetulekurühmadele langevana Bork (2006) ja regressiivsena Sterner ja Carlsson (2012) Itaalia puhul[13]. Sterner et al (2012) on leidnud, et see on pigem kergelt regressiivne või neutraalne arenenud riikide puhul ning progressiivne arenguriikide puhul. Leibkondade tüübid, kes on enim haavatavad keskkonnamaksude poolt, on maapiirkondades elavad leibkonnad (Bork 2006, Jacobsen et al 2003, Callan et al 2009), lastega leibkonnad (Bork 2006, Dresner and Ekins 2006), üksikettevõtjad (Jacobsen et al 2003) ning pensionärid (Dresner and Ekins 2006).

Kaudset jaotuslikku efekti on uuritud eelkõige süsinikumaksu puhul ning leitud, et suurimat maksukoormust kannavad energiatootmise ja transpordi sektorid, millele järgnevad teised energiaintensiivsed sektorid, näiteks toidu ja jookide tootmine (Cornwell, Creedy 1996), pakkettreiside pakkumine (Wier et al 2005) ning lubja tootmine (Grainger, Colstad 2010). Kuna madala sissetulekuga leibkonnad kulutavad suurema osa sissetulekust energiaintensiivsetele kaupadele ja teenustele (kodude kütmine, elekter), on tulemuseks ebasoodne kaudne jaotuslik efekt vaesematele leibkondadele. Praktiliselt kõik empiirilised uurimused, mille ülevaade on toodud peatükis 2.2, on leidnud, et kaudne jaotuslik efekt on regressiivne, erandiks on vaid Labandeira ja Labeaga (1999), kus on leitud see proportsionaalne olevat.

Jaotuslikud efektid on aga tihedalt seotud käitumuslike efektidega: kõrgema sissetulekuga inimestel on enam võimalusi kohandumiseks, näiteks muutes elukohta, või investeerides kütusesäästu. Traditsiooniliselt on majanduses kasutatud käitumislike efektide hindamiseks elastsusi, kuid tuleb meeles pidada, et tavaliselt räägitakse keskmisest elastsusest. Elastsuste hindamisel kasutatavad metoodikad on väga erinevad, kaheosalistest mudelitest mitmevõrrandiliste

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[13] Teiste antud uurimuses kasutatud riikide puhul oli see progressiivne või neutraalne ehk Itaalia oli erandlik

nõudlussüsteemideni. Hinnangute varieeruvus on väga suur: näiteks on bensiini hinnaelastsused vahemikus -0,03 USAs (Nicol 2003) kuni -1,28 Itaalias (Tiezzi 2005). Samuti erinevad tihti ka ühe riigi kohta tehtud hinnangud: näiteks mootorikütuse sissetulekuelastsus Hispaanias on ühe töö põhjal 0,51 (Asensio et al 2002); teise põhjal 1,79 (Labandeira et al 2005). Seega võib mõnede tööde põhjal väita, et tegu on esmatarbekaubaga (Kayser 2000, Asensio et al 2002, Sardianou 2008), ning teiste põhjal, et luksuskaubaga (Labandeira et al 2005, Barros ja Pietro-Rodriguez 2008).

Osa hinnangute erinevusest võib tulla ajalisest perspektiivist: meta-analüüside põhjal võib väita, et pikaajalised elastsused on kõrgemad kui lühiajalised (Goodwin et al (2004). Samuti on USA mootorikütuse hinnaelastsused madalamad kui Euroopas. Kuid oma osa tundub olema ka metoodikal: näiteks nõudlussüsteemide abil leitud hinnaelastsused on kõrgemad kui kaheosaliste mudelite puhul leitud hinnaelastsused. Viimatimainitu on kooskõlas ka Basso ja Oum (2007) järeldusega, et demograafiliste efektide väljajätmisel on sissetulekuelastsused ülehinnatud. Elastsuse arvutamisel on olulised kontrollmuutujad näiteks asulatüüp, leibkonnapea vanus, rass, sugu, haridus, tööhõive staatus ning laste arv (Kayser 2000, Asensio et al 2002, Sardianou 2008).

Üha enam pööratakse tähelepanu ka efektide erinevusele erinevates sissetulekurühmades. Näiteks Wadud et al (2009) on näidanud, et mootorikütuse hinnaelastsus on suurim madalaimas ja kõrgeimas sissetulekukvintiilis. Sissetulekuelastsuse puhul on Asensio et al (2002) leidnud, et see on kõrgeim vaeseimates leibkondades ning madalaim rikkaimates leibkondades. Wadud et al (2009) on leidnud, et bensiini tarbimine ei sõltu sissetulekust madalaimas ja kõrgeimas sissetulekurühmas.

Varasemate empiiriliste tööde analüüsi tulemusel saab kokkuvõtvalt välja tuua, et seni on keskkonnamaksude jaotusliku mõju uurimisega tegeldud arenenud riikides, ning need tulemused ei pruugi olla üldistatavad Kesk- ja Ida-Euroopa riikidele. Senised tööd on näidanud, et kodude kütmiseks kasutatava energia maksustamine on regressiivne, kuid mootorikütuste maksustamise mõjude osas on tulemused erinevad. Veelgi enam erinevad aga hinnangud mootorikütuste hinna- ja sissetulekuelastsuse osas.

Analüüsi tulemused

Käesolevas töös leiti, et Eesti keskkonnamaksude otsene jaotuslik efekt on progressiivne ning pigem vähendab ühiskonnas valitsevat ebavõrdsust (väited ja uurimistulemused on kokkuvõtvalt esitatud tabelis I). Selle asjaolu põhjuseks on maksuobjekt, mis on peamiselt mootorikütus ning selle maksustamine on Eesti näitel progressiivne, kuna rikkamad inimesed tarbivad mootorikütust rohkem. Kuigi elektri ning kodude kütteks kasutatava energia maksustamine on selgelt regressiivne, on nendest tulenev maksukoormus väiksem kui mootorikütuse aktsiisist tulenev koormus. Teistest suuremat keskkonnamaksukoormust kannavad töötavate liikmetega leibkonnad ning lastega leibkonnad. Oluliseks teguriks on ka leibkonna elupaik: maapiirkondades elavate leibkondade keskkonnamaksudest tulenev koormus on oluliselt kõrgem kui linnapiirkondades elavatel leibkondadel.

Tabel I. Väited ning kokkuvõtlikult uurimistulemused

Väide	**Tulemus**
Uurimisküsimus 1. Millised on Eesti keskkonnamaksude otsesed jaotuslikud efektid?	
Väide 1. Keskkonnamaksud on regressiivsed ning suurendavad ebavõrdsust riigis	Ei leidnud kinnitust: keskkonnamaksud Eestis on progressiivsed.
Väide 2. Keskkonnamaksude negatiivne mõju ilmneb eelkõige teatud maksuliikide (kütteainena kasutava energia maks) ning teatud leibkonna tüüpide puhul (pensionärid, lastega leibkonnad, maapiirkondades elavad leibkonnad)	Kinnitatud: kütteainena kasutatava energia ja elektri maksustamine on regressiivne. Suuremat maksukoormust kannavad töötavad ja lastega leibkonnad ning maapiirkondades elavad leibkonnad.
Uurimisküsimus 2. Millised on Eesti keskkonnamaksude kaudsed jaotuslikud efektid?	
Väide 3. Madala sissetulekuga leibkonnad tarbivad enam kaupu, mille kaudne keskkonnamaksukoormus on suurem	Kinnitatud: kuna madalamates sissetulekurühmades kulutavad leibkonnad enam energiaintensiivsetele kaupadele ja teenustele, on ka sellest tulenev keskkonnamaksukoormus suurem.
Väide 4. Otsese ja kaudse jaotusliku efekti tõttu suureneb ebavõrdsus riigis	Ei leidnud kinnitust: kuna otsene jaotuslik efekt on suurem, on tegu siiski progressiivse mustriga.
Uurimisküsimus 3. Kuidas on majapidamised tarbimise muutuste kaudu reageerinud Eesti keskkonnamaksudele ning kuidas on see seotud erinevate sotsiaaldemograafiliste teguritega?	
Väide 5. Leibkondade käitumine ei ole keskkonnamaksude (mootorikütuse aktsiisi) tõttu oluliselt muutunud ning maksustatud toote tarbimine ei ole oluliselt vähenenud.	Kinnitatud: bensiini hinnaelastsus on suhteliselt madal ning bensiini tarbimist on vähendanud pigem majanduskriis ning töötuse suurenemine.
Väide 6. Leibkondade hinnatundlikkus sõltub erinevatest sotsiaal-demograafilistest tunnustest	Pole selget järeldust: tegureid on väga palju ja ühtset mustrit ei leitud.

Kuigi otsene keskkonnamaksudest tulenev koormus langeb enam rikkamatele leibkondadele, siis kaudne efekt on kergelt regressiivne, kuna madalama sissetulekuga leibkonnad kulutavad rohkem energiaintensiivsetele kaupadele ja teenustele nagu toit, alkohol ja eluase. Nimetatud kaupade ja teenuste keskkonnamaksude koormus on suurem kui näiteks kestuskaupade, puhke- ja kommunikatsiooniteenuste puhul, mida tarbivad enam rikkamad leibkonnad.

Otsese ja kaudse jaotusliku efekti summa on Eestis siiski progressiivne, kuna suurema osa moodustab otsene maksukoormus. Kui aga jätkatakse viimaste aastate trendi, kus suurendatakse oluliselt ka kütteainena kasutava energia maksustamist, siis võib praegune progressiivsus muutuda regressiivsuseks ning ebavõrdsus suureneda.

Majapidamiste käitumise muutust hinnati antud töös elastsuste abil, mis arvutati kaheosalisest mudelist. Selgus, et sissetulekute tase avaldab küll positiivset mõju tõenäosusele, et leibkond bensiinile üldse kulutab, kuid see mõju on väiksem kui varasemate empiiriliste tööde põhjal võinuks eeldada. Otsus bensiini tarbida on pigem seotud tööhõive staatusega: leibkondades, kus on töötavad liikmed, on tõenäosus bensiinile kulutada suurem kui teistes leibkondades. Antud töös leitud bensiini tarbimise sissetulekuelastsus on 0.52. Suhteliselt madal sissetulekuelastsus võib olla seotud asjaoluga, et mitmed tööandjad kompenseerivad töötajatele bensiinikulusid, ning ilmselt see tendents kasvab koos sissetulekuga. Täiendavalt võib seda mõjutada majanduskasvu aegse autohindade ja liisingute odavnemine, mistõttu auto omamine kasvab ka madalamates sissetulekurühmades. Sissetulekuelastsuse põhjal otsustades ei ole bensiin luksus-, vaid esmatarbekaup.

Hinnaelastsuseks leiti antud töös -0.56, mis tähendab, et bensiini tarbimine küll hinna tõustes väheneb, kuid väiksemal määral kui hinnatõus (siinjuures tuleb veel arvestada, et tegu on vaid erakulutustega bensiinile). Seega vastab kütuseaktsiis üsna hästi heale Ramsey maksustamispõhimõttele, mille järgi on soovitav maksustada madala nõudluselastsusega tooteid, kuid selle järgi võib antud maksu liigitada fiskaalseks maksuks. Mida Ramsey maksustamispõhimõtte puhul ei arvestata, on välismõjude olemasolu (Pigou maksustamispõhimõte): kütuseaktsiisi tase peaks olema tunduvalt kõrgem, et olla tõhus keskkonnaprobleemi lahendav meede ning tarbimise vähendamist stimuleeriv maks. Näiteks (Anspal, Poltimäe 2009) järgi olid Eesti maanteetranspordi väliskulud 2007. aastal oluliselt kõrgemad kui kütuseaktsiis (vastavalt 441 miljonit eurot ja 278 miljonit eurot), sealjuures tuleb veel arvestada, et kütuseaktsiisist 75% läheb teehoidu, mistõttu väliskulude katmiseks jääb veelgi väiksem summa. Üldistatult võib öelda, et keskkonnaprobleemi lahendamisel maksumeetme abil ongi probleemiks, et keskkonnapoliitika eesmärgid ei ühti maksupoliitika eesmärkidega: kui maks stimuleerib tarbimise vähendamist, väheneb maksubaas, kuid riigieelarve jaoks on eelistatud maksud, mille laekumine on stabiilne või isegi kasvav.

Kokkuvõttes ei ole keskkonnamaksud Eestis ebavõrdsust suurendanud, vähemalt mitte rahaliste maksete puhul. Samas, kui hinnata maksu tõhusust, siis bensiini puhul ei saa öelda, et kütuseaktsiis oleks kaasa toonud oluliselt väiksema tarbimise taseme. Tundub, et pigem on tarbimise vähendamine olnud seotud majanduskriisiga ning ilmselt ka sellest tuleneva tööhõive muutustega. Tööhõive staatus on seotud ka keskkonnamaksudest tuleneva progressiivse maksukoormusega: töötavate inimeste sissetulek on suurem ning seetõttu on ka nende tarbimise tase suurem, sealhulgas ka bensiini tarbimine.

Käesoleva töö põhjal võib teha poliitika järeldusi, mis kehtivad ka teiste riikide puhul, kus toimuvad kiired majandusmuudatused ning kus energiamaksude osakaalu suurendatakse. Kodude kütmiseks kasutatava energia ning elektri maksustamine on regressiivne ning viib kasvavale ebavõrdsusele ühiskonnas. Mootorikütuse maksustamise otsene jaotuslik efekt on progressiivne, kuid kaudne efekt regressiivne ehk lõppkokkuvõttes mõjutab see oluliselt ka madalamaid sissetulekurühmi. Tarbimise mõjutamine mootorikütuse hinnatõusu kaudu ei ole väga tõhusaks osutunud, kuna mootorikütuse hinnast suuremat rolli on mänginud näiteks suurenenud tarbimis- ja laenuvõimalused ning odavnenud sõidukihinnad. Tarbimist on piiranud pigem majanduskriis ning töötuse kasv. Seega, kui maksu eesmärk on keskkonnaprobleemi lahendamine, siis kütuse maksustamine ei ole parim meede selle saavutamiseks, kuna maksustamispoliitika osas kehtivad teised kriteeriumid kui väliskulude sisestamise puhul.

Antud töö edasiarendused sõltuvad oluliselt andmetest, mida Eestis praegu piisavalt ei koguta. Kuigi meil on hea ülevaade üldisest keskkonnaseisundist ja tarbimiskulutustest, siis puudus on sellistest andmetest, mille põhjal saaks hinnata inimeste keskkonnakäitumist (mikrotasandil), näiteks olulise keskkonnamõjuga toodete tarbimist, selle muutust ajas ning seotust erinevate teguritega. Samuti puuduvad andmed kütusekasutuse ja selle hinnatundlikkuse kohta ettevõtetes, taas võib leida andmeid üldiselt sektori tasemelt, kuid mitte mikrotasandilt.

Oluliste edasiarendustena antud tööle tuleb esmajärjekorras välja tuua keskkonnamaksude mõju ettevõtetele: kas nende tulemusena on muutunud majandusstruktuur, milliseid investeeringuid on tänu keskkonnamaksudele tehtud ning kuivõrd on muutunud ettevõtetes kasutatav energialiik. Lisaks peaks käsitlema ka teiste keskkonnapoliitika instrumentide nagu näiteks CO_2 kvootide, keskkonnaregulatsioonide, jne, võimalikku mõju ning koostoimet keskkonnamaksudega. Samuti oleks vaja uurida, millist mõju omab ettevõtete poolt kasutatav auto ja kütuse kompensatsiooniskeem: kui palju seda tegelikkuses kasutatakse, kuivõrd seda kompensatsiooni kasutatakse ka isiklikeks sõitudeks ning kui hinnatundlik selline kütusekasutus on. Need aspektid on olulised antud töö tulemuste täiendamiseks ning selgitamaks välja keskkonnamaksu kui majandusmeetme võimalused muuta riiki energia- ja ressursitõhusamaks.

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