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Assessment of Environmental Impacts of Landfilling and Alternatives for Management of Municipal Solid Waste

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

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Prügi ladestamise keskkonnamõjud ja alternatiivid olmeprügi käitlemisel

VIKTORIA VORONOVA

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LIST OF PUBLICATIONS INCLUDED IN THE THESIS

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Paper II: Voronova, V., Moora, H., Loigu, E., 2011. Environmental assessment and sustainable management options of leachate and landfill gas treatment in Estonian municipal waste landfills. Management of Environmental Quality: An International Journal, 22(6), 787 - 802.

Paper III: Moora, H.; **Voronova, V**.; Uselyte, R., 2012. Incineration of Municipal Solid Waste in the Baltic States: Influencing Factors and Perspectives. Avraam Karagiannidis (Toim.). Waste to Energy: Opportunities and Challenges for Developing and Tsansition Economies (237 - 260). London: Springer-Verlag London Ltd

Paper IV: Voronova, V., Piirimäe K., Virve M., 2013. Assessment of applicability of Pay As You Throw system in Estonia, Management of Environmental Quality: An International Journal (in print).

Author's Contribution to the Publications

Paper	Original idea	Study design and methods	Data collection and handling	Contribution to result interpretation and manuscript preparation	Responsible for result interpretation and manuscript preparation
I	x		x	x	
II	x	x	x	x	x
III			x	x	
IV	x	x	x	x	x

INTRODUCTION

Municipal solid wastes (MSW) often form a small part of the total annually produced wastes varying between 8– 9% in European Union (EU). In Estonia, the share of MSW in total waste production is about 2% (Eurostat, 2010). Environmental impact of MSW is largely defined by the methods used for handling it as well as people's lifestyle, which determines the amount of produced wastes.

Landfilling is still a dominant method of MSW handling in Estonia as well as in other Baltic countries. However, it can cause serious environmental damage by polluting the soil, ground water, and air. Air pollution is mainly caused by formation of greenhouse gas (GHG) emissions from landfills and other waste management practices such as waste incineration, recycling, and collection of MSW. Moreover, landfilling of MSW can contribute to soil and groundwater pollution by forming leachate. Such kind of environmental impacts can be significant if the share of biodegradable fraction in MSW is high and climate is relatively humid, as in Estonia.

The EU Landfill Directive (EC, 1999) aims to control environmental impacts by landfilled MSW in Europe. In accordance with this Directive, all Member States should have reduced the amount of landfilled biodegradable waste to 75% by 2006 compared to 1995 level, and to 50% and 35% by 2009 and 2015, respectively, with derogation period of four years for countries who landfill more than 80%.

Biodegradable part of landfilled MSW should not exceed 28% by 2020 following the requirements of landfill Directive in EU27 (ETC/SCP, 2011). Moreover, the ban to landfill the biodegradable fraction of MSW is also under consideration. Thus, landfilling remains as an option for handling MSW, but only the inert wastes can be landfilled.

The other alternative to MSW landfilling is the mass-burn incineration technique with energy recovery. This alternative has to be weighed, including in Estonia, in terms of its cost-effectiveness and sufficiency of MSW to run daily mass-burn incineration plants. On the other hand, the Baltic energy markets are undergoing a transition towards new sources of energy and significant changes will take place in the coming years (Moora and Lahtvee, 2009). The biodegradable fraction is defined as a part of MSW biomass, and counted as a renewable energy source. The use of MSW for energy production can contribute to achieving the 20% renewable energy goal and the 20% reduction in CO_2 emissions agreed upon at the EU level. Heat and electricity from waste contributes to the energy generated by conventional power plants, which still predominantly use fossil fuels. According to the Waste Framework Directive (EC, 2008), recycling of materials from household wastes should be increased to at least 50% by weight. Therefore, it would be reasonable to achieve this target by applying large-scale incineration of MSW in Estonia.

Possible environmental impact from landfilling can also be reduced indirectly by minimising the generation of landfilled wastes. Such minimisation can be achieved by implementing proper economic instruments, e.g., the variable pricing - pay as you throw (PAYT) model into the existing waste management system. Such direct form of unit pricing for wastes aims at stimulating households to divert an increased portion of their discards away from the traditional means of disposal mainly to recycling. This waste management approach ranks highly in the waste management hierarchy and is believed to be less hazardous to the environment and cost-efficient in the long run.

The waste management sector in Baltic States is strictly governed by relevant European policies and directives. Since 2004, when Estonia became an EU member, considerable changes in waste management have occurred. All old and unsanitary landfills were closed in Estonia by 16 July 2009 and only five new technically equipped landfills remained operational. Nevertheless, the experience with new landfills indicates that there are still problems in assuring compliance with some technical requirements, e.g., leachate treatment, gas collection and utilization systems. Leachate treatment particularly required additional efforts in terms of technology and related financial resources, since the initially designed treatment capacity was insufficient for the load of leachate (Loigu, 2010). Operators of new landfills in Estonia still lack knowledge about the key design parameters of emission treatment technologies, such as leachate production rate, composition, and landfill gas (LFG) potential.

Mass-burn incineration of MSW as an alternative option of waste utilisation was well received in Estonia. With regard to mass-burn incineration, economic, social and environmental problems have to be considered as a whole. As an economic instrument, favourable conditions have to be created to implement the waste-to-energy (WtE) concept. For example, a pollution charge for municipal waste disposal (landfill tax) was introduced in Estonia in 1990. Until 2005, the rate was very low at €0.10–0.20 per tonne. In 2006, it rose to €7.8 and in 2012 to €17.25 per tonne. Further increase will elevate the tax to €29.84 in 2015. Such an increase in disposal tax would favour the implementation of alternative options, e.g., MSW incineration. Social factors in terms of public opinion have to be accounted as well. Positive acceptance by people living in the neighbourhood is essential in the early developmental stages of MSW incineration unit facility.

Material recycling as a second alternative option raises the question of how to minimise MSW amounts and enhance material recycling rate. Its solution can be found through the assessment of possibilities of implementation of variable price models into the current waste management system (Billitewski, 2008; Karagiannidis et al., 2008; Sauer et al., 2008; Skumatz, 2008; Ventosa, 2008; Zotos et al., 2009; Dahlen and Lagerkvist, 2010). This will help significantly to reduce the generated waste amounts by enhancing individual sorting ability.

Huge investments were made to construct new landfills that are adjusted to recent European Regulations, which means that landfilling of MSW will continue at least next fifteen years. The question is in which way and what could be alternative options to reduce amounts of wastes to be landfilled? Therefore, to the study applied the Environmental assessment of MSW landfills from its life cycle perspective with the aim to investigate all environmental impacts from the generation and collection as well as the use of methane gas and leachate treatment options. Additionally the perspectives of mass-burn incineration were identified by this thesis. The results of this evaluation can be used in other countries. The separate collection of household wastes started to develop in Estonia quite recently. Therefore, the capacity and willingness to implement pay-as-you-throw (PAYT) system in Estonian municipalities was assessed. More specific objectives of the thesis involve:

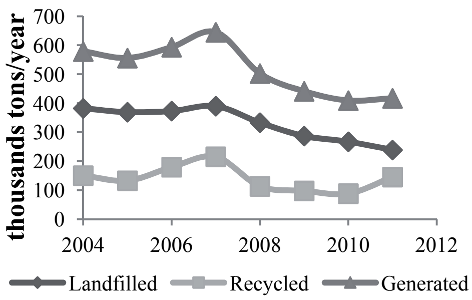
1. assessment of the impact of GHG emissions from MSW management to the climate change in Estonia in 2000 - 2020.
2. comparison of various landfill gas and leachate treatment technologies from a life-cycle perspective.
3. assessment of the possible environmental impact of inert landfill.
4. identification and discussion on the main influencing factors and perspectives for MSW incineration in Estonia and the other Baltic States.

To reach the proposed objectives the field data was collected in the frame of projects “Research and analyses of various treatment technologies of landfill leachate. Development of suitable leachate cleaning technology in Estonian conditions” and “Pay as you throw system development in Greece, Cyprus and Estonia”. Collected data were analysed and laboratory tests carried out to assess the effectiveness of leachate treatment in Estonian landfills. The waste management planning model (WAMPS) was applied for environmental impact assessment of MSW landfilling.

The work was carried out by implementing the studies on environmental assessment of management options of leachate and landfill gas treatment in Estonian landfills (**paper II**); assessment of the factors and perspectives of MSW incineration and the impact of GHG formation from MSW (**papers I and III**) and assessment of applicability of PAYT system in Estonian municipalities (**paper IV**).

2. MSW MANAGEMENT IN ESTONIA

MSW management is a rapidly changing field of activity nowadays. Waste handling technologies are improved and implemented through EU member states. Therefore, the share of landfilled MSW decreased from 66% in 2004 to 57% in Estonia in 2011 (Figure 1).



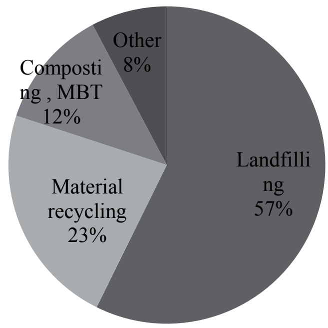


Figure 1: MSW amounts landfilled, recycled, generated (2004-2011) in Estonia (Source: Environment Information Centre, 2012)

Figure 2: The share of MSW treatment in Estonia in 2011 (Source: Environmental Information Centre, 2012)

Recycling rates of MSW are steadily increasing and it formed 35% of the total waste generated in 2011 (Environmental Information Centre, 2012). It can be explained by the enhancement of the mechanical-biological treatment (MBT) of MSW and composting of biodegradable part of wastes that formed 12% of MSW in 2011(Figure 2). In the Baltic States, landfilling of wastes is still the prevailing treatment method (more than 50% of MSW) despite the fact that the EU waste hierarchy considers landfilling as the least favourable option compared to prevention and minimisation that are the most preferred options. Huge amounts of landfilled wastes require, therefore, proper monitoring and environmental assessment of landfills.

2.1. Waste amounts and composition

The MSW amount generated depends a lot on the socio-economic conditions of the region (Beigl et al., 2004; Moora, 2009). In Estonia, MSW generation was also in-line with increasing GDP and reached the maximum in 2007 (Figure 3). After that year, both indicators showed a decrease. Further growth of both indicators can be expected, but not rapidly.

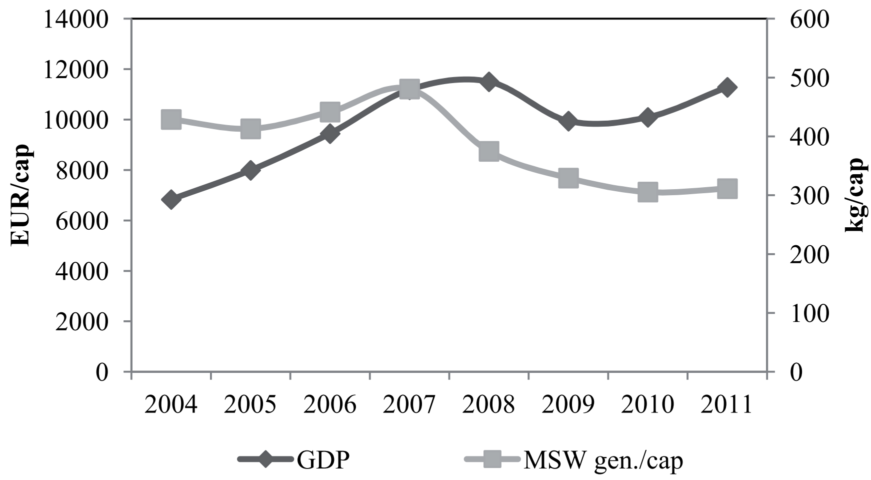


Figure 3: GDP and MSW generation trends in Estonia in 2004-2011 (Source: Estonian Statistics)

The second important factor for environmental assessment of MSW streams is the composition of wastes. Unfortunately, existing data about the composition of MSW, especially their mixed part, is quite poor. In Estonia, the main source of waste data is the database of the waste register at the Environment Information Centre of the Ministry of Environment, which is based on systematised waste reporting (Moora, 2009). However, the data sometimes is not complete and cannot be used as an input for waste management planning model. To obtain more precise data about mixed MSW, sorting analysis was carried out in 2005 and 2008 by Stockholm Environment Institute (SEI) and the results are presented in **paper I** and by Moora and Jürmann (2008). Another sorting study was carried out for Tallinn metropolitan area in 2005–2006 by Vilms (2006).

In 2010–2011, a sorting study was carried out in Tartu. Results of this study showed significant changes in the amounts of packaging wastes and their share in MSW decreased from 34.5% in 2008 to 24.2% in 2010–2011. Lower share of packaging material in mixed MSW can be explained by considerably improved sorting of packaging material in comparison with 2008. Landfilling of unsorted MSW still remains a problem, especially with regard to organic wastes. The amount of bio-waste in Tartu formed 38.4% of total MSW composition in 2010, which is 2% more than the Estonian average of 2008, which was 36.65% according to Moora and Jürmann (2008).

2.2 Requirements for waste collection in Estonia

Since 2005, all municipalities, where the number of residents exceeds 1500, are obliged to implement waste collection system (RT, 2004). The main aim of organised waste collection is to link all waste holders into a common waste management system, minimising illegal dumping and misuse of waste bins.

Since January 2008, requirement for separate waste collection came into force in Estonia. According to that, municipalities are responsible for providing containers to the residents for separate collection of paper and cardboard, mixed packaging, glass, biodegradable wastes and mixed municipal wastes.

According to the organised waste management system, municipalities should organise a bidding competition for waste handling companies. In every region, local authority should set up a marginal rate for waste handling: a company that suggests the lowest price will win a public tender and serve a region during the next five years. All residents in this region are obliged to make an agreement only with this company.

3. MATERIAL AND METHODS

3.1 Life - cycle assessment of landfilling

Life-cycle assessment (LCA) methodology was applied to assess environmental impact from waste management in terms of landfilling (**paper II**) of MSW and in comparison with the option of incineration (**papers I and III**).

The definition by SETAC (Society of Environmental Toxicology and Chemistry) states that, "Life Cycle Assessment is a process to evaluate the environmental burdens associated with a product, process, or activity by identifying and quantifying energy and materials used and wastes released to the environment; to assess the impact of those energy and materials used and released into the environment; and to identify and evaluate opportunities to affect environmental improvements. The assessment includes the entire life cycle of the product, process or activity, encompassing, extracting and processing raw materials; manufacturing, transportation and distribution; use, re-use, maintenance; recycling, and final disposal."

A complete LCA consistent with international standards (ISO, 2006) and handbooks (e.g., Guinee, 2002) consists of four interrelated phases:

1. **Goal and scope definition** explains the reason for carrying out the study, selects borders, function, functional unit and allocation technique.
2. **Inventory analysis** collects all the data of the unit processes within a product system and relates them to the functional unit of the study.
3. **Impact assessment** with four sub-phases: classification, characterisation, normalisation, and weighting, assesses the impact of the inputs and outputs identified in the inventory analysis.
4. **Improvement assessment** evaluates the results from the inventory analysis or impact assessment and compares them with the goal of the study defined in the first phase.

Correlation among the LCA phases makes LCA an iterative process (Hillary, 1995) (Figure 4). The calculation and evaluation procedure is repeated until the analysis reaches the required level of detail and reliability.

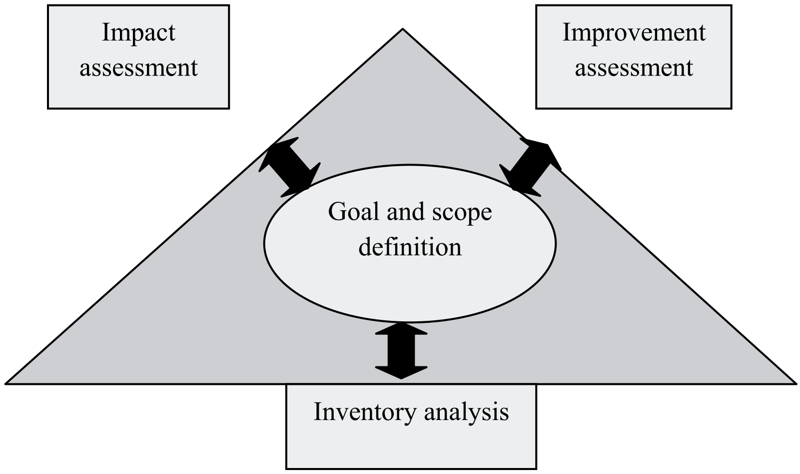


Figure 4:Interrelation of LCA phases (redrawn from Hillary, 1995)

3.1.1 Functional unit and system boundaries

The definition of the functional unit is very important in the first stage of the life-cycle process. The functional unit is a measure of the function of the studied system and it provides a reference to which the inputs and outputs can be related. The functional unit of the current study was the amount of MSW landfilled in Estonia annually.

System boundary determines which unit processes should be included in the process. Defining system boundaries is partly based on a subjective choice, made during the scope phase when the boundaries are initially set. The following boundaries can be considered:

- Boundaries between the **technological system** and **nature**. A life-cycle usually begins at the extraction point of raw materials and energy carriers from nature. Final stages normally include waste generation and/or heat production.
- **Geographical area:** Geography plays a crucial role in most LCA studies. For example, infrastructures, such as electricity production, waste management and transport systems, vary from one region to another. Moreover, ecosystems' sensitivity to environmental impacts differs regionally too (Ekvall, 2007).
- **Time horizon:** Boundaries must be set not only in space, but also in time. Basically, LCAs are carried out to evaluate present impacts and predict future scenarios. Limitations to time boundaries are given by technologies involved, pollutants lifespan, etc.

3.1.2 Data quality requirements

The data quality requirements have to be met in all LCA studies. Different or non-reliable data sources would have a great impact on the result of the study. The following parameters should be considered:

- Time-related coverage: age of data and the minimum length of time over which data should be collected (i.e., 1 year); the desired age of data (5 years).
- Geographical coverage: area from which data for unit processes should be collected to satisfy the goal of the study (i.e., place, region, country, etc.).
- Usage of average or marginal data (data resolution, i.e., level of aggregation) (Moora, 2009).

3.2 The waste management planning model (WAMPS)

A variety of LCA models concerning waste management planning have been developed since 1990. A detailed review of the models was provided by Gentil et al. (2010).

The WAMPS model is the waste management planning model that was developed based on the more in-depth LCA model ORWARE by the Swedish Environmental Research Institute (Björklund, 2000; Eriksson et al., 2000; Sundqvist et al., 2002). The WAMPS model was adjusted to the Estonian conditions by calibration, testing and generation of regional database by Estonian Institute for Sustainable Development, Stockholm Environment Institute (SEI). The WAMPS model consists of different sub-models, which allow comparing different waste management options such as landfilling, incineration, recycling, composting, and anaerobic digestion of MSW in terms of environmental assessment of their impacts. A detailed description of each sub-model for waste management planning was provided by Moora (2009) in his PhD thesis.

The author of this thesis contributed to the generation of regional database and testing of the landfilling sub-model (**paper II**).

WAMPS compares a waste management system with a background system. The waste management system can produce different products depending on the choice of treatment and recycling: heat, steam, electricity, vehicle fuel (biogas), compost, paper, plastic, metals, etc. In background system, similar products are produced from virgin origin. When a product is produced from waste, it substitutes a product in the background system. Each waste product has an alternative in the background system with a virgin raw material source and a production process that is included in the model. In WAMPS, different recovery options are compared with the background system and potentially ‘saved emissions’ are assessed. The background system consists of heat production (alternative to waste incineration and combustion of biogas and landfill gas), electricity production (alternative to waste incineration and combustion of biogas and landfill gas), vehicle fuel production (alternative to biogas), fertiliser production (alternative to compost and digestate), and production of materials (plastic, newsprint, paper packages, glass, steel, aluminium, etc). All these products from the background system can also be produced by different waste recovery methods.

The net emissions from the studied system are calculated according to:

$E_{net} = E_{waste} - E_{Background}$

E_{net}: Net emission (tonnes/year or kg/year)

E_{waste}: Emission from a waste process that produces a certain amount of product (tonnes/year or kg/year).

$E_{Background}$: Emission from the same amount of alternative virgin production in the background system (tonnes/year or kg/year).

Environmental impact is considered in LCA as a consequence of a physical interaction between a studied system and the environment. In LCA, all environmental impacts are represented by different categories. WAMPS model focuses on four main environmental impact categories (Sundqvist et al., 2002):

- **Global warming:** All emissions are expressed as CO_2-equivalents: 1 kg of methane (CH_4) is equal to 25 kg of fossil carbon dioxide (CO_2) and 1 kg of nitrous oxide (N_2O) is equal to 310 kg of fossil CO_2.
- **Eutrophication of water:** All emissions are expressed as phosphate (PO_4^{3-}) equivalents: 1 kg of phosphorus (P) is equal 3.06 kg of PO_4^{3-}, 1 kg of nitrogen (N) is equal to 0.42 kg of PO_4^{3-}, 1 kg of NH_3/NH_4 is equal to 0.34 of PO_4^{3-}, and 1 kg of COD is equal to 0.022 kg of PO_4^{3-}.
- **Acidification:** All emissions are expressed as SO_2-equivalents: 1 kg NO_x is equal to 0.7 kg SO_2, 1 kg of NH_3 is equal to 20.88 kg SO_2, and 1 kg of HCl is equal to 0.88 kg of SO_2.
- **Photooxidant formation:** Photooxidant formers have been divided into volatile organic compounds (VOC) and NO_x. CH_4 is included in the VOC, but with a relatively low factor: 1 kg of CH_4 is equal to 0.006 kg ethane (C_2H_4), CO is equal to 0.03 kg C_2H_4, and non-methane volatile organic compounds (NMVOC) is equal to 0.416 kg of C_2H_4.

Toxicity-related impacts were neglected in this study because there are still many uncertainties in modeling toxicological impacts (Finnveden and Lindfors, 1998; Reap et al, 2008).

3.2.1 WAMPS landfill module

The landfill used in this study is a municipal solid waste landfill (Figure 5) where mixed household wastes are deposited (excluding hazardous wastes). The landfill has a bottom sealing, e.g., plastic, rubber, betonite or clay, and a top cover of soil. Another input is water from precipitation (rain and snow). The precipitated water will evaporate, run-off the cover or percolate through the landfilled waste. The model has gas and leachate collection possibilities with further treatment options.

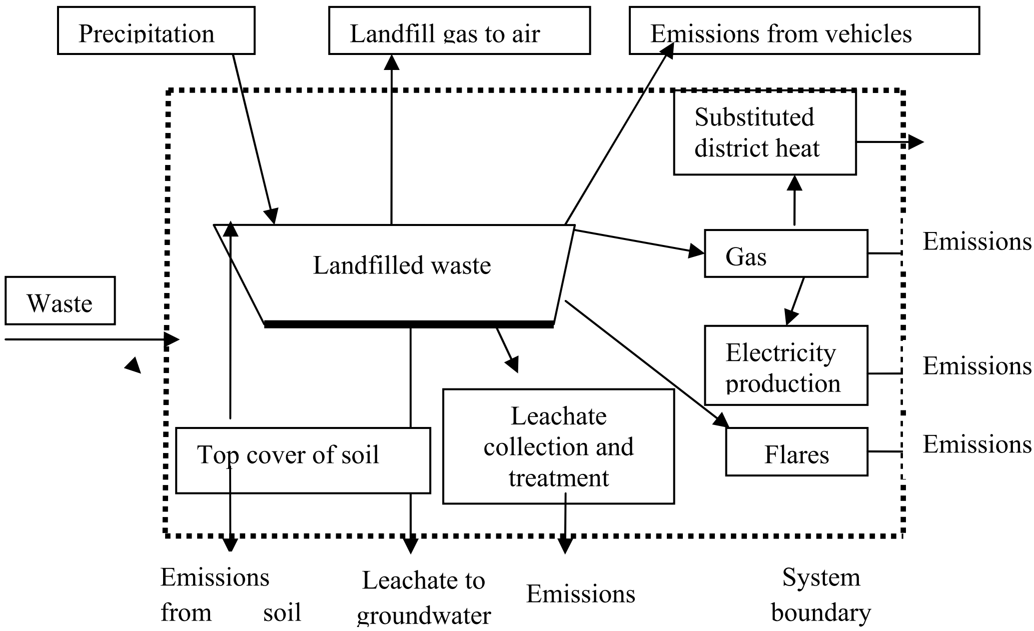


Figure 5: Schematic illustration of modelled average operating MSW landfill.

The landfill gas consists mainly of methane and carbon dioxide. Emissions of methane and carbon dioxide were calculated according to the carbon balance of a conventional MSW landfill (Sundqvist, 1999):

$$CH_4 = (1 - \gamma) \cdot x\ (1 - \varepsilon)\ x\ \alpha\ x\ \beta\ x\ 0.99\ x\ 16/12\ kgCH_4\ kgC_{in} \quad (1)$$

$$CO_2 = (1 - \varepsilon)\ x\ (1 - \beta)\ x\ \alpha\ x\ 0.99\ x\ 44/12\ + $$
$$+\ \gamma\ x\ (1 - \varepsilon)\ x\ \alpha\ x\ \beta\ x\ 0.99\ x\ 44/12\ kgCO_2\ kgC_{in} \quad (2)$$

α – degradation yield *kg degr.C kg* C_{in}

β – molar (or volume) ration *kmol* CH_4 *kmol* $(CO_2 - CH_4)$

γ – oxidation yield of CH_4 in soil cover *kg oxidised* CH_4 *kg* CH_4 *transported through soil*

ε – part formed methane that is recovered *kg recovered CH_4 kg formed CH_4*

Landfilling of wastes will cause emissions during a long period of time (Sabbas et al., 1998; Sundqvist et al., 1997; Hellweg, 2000; Doka and Hischier, 2005, Obersteiner et al., 2007). WAMPS landfill can evaluate long-term future emissions by separation over different time periods. The model integrates emissions between two time periods according to Sundqvist (1999):

- <u>A surveyable period,</u> which is the period until a pseudo steady-state in the landfill processes is obtained. This period should usually be of the magnitude of one century.
- <u>A hypothetical, infinite time period</u>, which is the period from the start until the landfilled material is completely released in the environment.

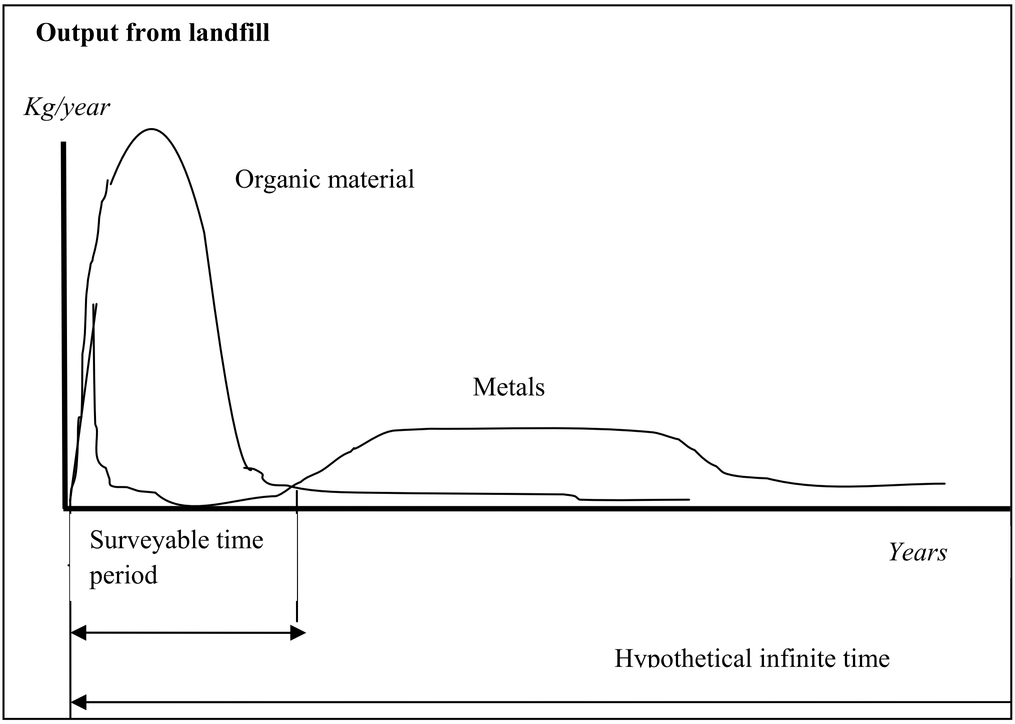


Figure 6: Schematic illustration of surveyable time and hypothetical infinite time periods [redrawn from Sundqvist, 1999]

The end of a surveyable time period is characterised mainly by full degradation of organic material when the methane generation ends (Figure 6). The hypothetical time period is the time until the landfilled material is completely released in the environment.

For characterisation of results, CML-IA database, version 4.1 was used. The database was developed by Hejungs et al. (1992) in the Institute of Environmental Sciences, Leiden University. Updated version of database presents characterisation factors for Problem Oriented Approach and for some other methods, such as Eco-indicator 99, EPS and USEtox. Normalisation factors were calculated based on the annual impacts of relevant pollutants in Estonia provided by the Estonian Environmental Information Centre for the year 2010. In this research, the Problem Oriented Approach in WAMPS model has been used.

3.3 Limitations

The reliability of results of the LCA modelling of waste management can be limited by the following factors: data quality, impact categories, emissions, and allocation (Ekvall et al., 2007; Ericksson and Bekay, 2010; Clavreul et al., 2012).

3.3.1 Data quality

Data quality largely depends on the source and time frame in any LCA study. The primary data (measured) is always preferable than secondary data (literature sources). Sometimes, when primary data is not available, the knowledge gaps can be filled in by importing data from secondary sources, which creates uncertainty in data quality.

As for waste management, the data concerning waste composition and amounts are essential. According to several authors (Ekvall et al., 2007; Moora, 2009; Ericksson and Bekay, 2010; Clavreul et al., 2012), such parameters as waste amount and composition are very uncertain in terms of analyses of changes in the quantities of waste generated and future prognoses. Therefore, the functional unit has to be specified to the annual quantity of waste generated in a geographical area. Different forecasting models can be used for defining future waste generation scenarios, e.g., dynamic modelling suggested by Börjeson et al. (2006).

3.3.2 Emissions and impact categories

During the LCA, emissions occurring in the process are characterised according to the impact categories, using the characterisation factors suggested in the database. Usually, different pollutants are summarised into one specific category with the loss of the spatial information. This will increase the uncertainty of the environmental impacts caused by emissions. To solve this problem several impact assessment site-dependent models have been developed (e.g., Finnveden and Nilsson, 2005; Mutel and Hellweg, 2009).

Another opportunity to reduce uncertainty in impact results is to use the normalisation of the results, which link the study to the specific region or country. According to Zbicinski et al. (2006), normalisation effect is the percentage of a given emission's annual contribution to that effect in a certain area. Normalised results can be compared among the impact categories and help to understand the significance of impacts in a specific region.

The processes in LCA model of waste management are usually considered as linear (Ekvall et al., 2007; Clavreul et al., 2012). If one parameter, e.g., the recycling rate, significantly increases, the system will require additional consumption of fuel and the linear model will not be able to calculate the optimum of recycling rate. Therefore, the non-linear model or linear programming of models that account for boundary conditions (Ekvall et al., 2007) have to be used. This makes LCA of waste management more complicated due to the need for high quality data.

3.3.3 Allocation

Allocation is one way of handling multifunctional processes in LCA (Finnveden et al., 2009). Waste management processes are usually multifunctional, e.g., landfilling of waste has the function of landfill wastes and production of energy by recovering the gas. According to ISO (2006), allocation can be avoided by dividing the unit processes into several sub processes or through system expansion with the inclusion of additional functions related to the co-products. Otherwise, the allocation should be performed. The principles of allocation are provided in several studies (e.g., Rebitzer and Hunkeler, 2004; Hejungs and Guinee, 2007; Finnveden et al., 2009).

The multifunctionality in waste management can be solved through the system expansion (Finnveden et al., 2009; Moora, 2009; Clavreul et al., 2012) and by accounting for the substitution of primary energy and virgin material productions. However, system expansion makes the analyses more complicated and sometimes allocation is needed.

Generally, the LCA methodology of waste management is limited by environmental impact. According to Ekvall et al. (2007), the methodology can be expanded to include the integration of economic model and social aspects.

3.4 Scenarios

In the frame of the environmental impact assessment of landfilling, the author concentrated on the impact of landfill gas to the air and leachate to water bodies. A detailed description of possible emissions of landfill gas and leachate is presented in **paper II**. Two main groups of scenarios were developed: (i) MSW management scenarios with the baseline year 2000 and (ii) two forecasted scenarios for the year 2020 (**papers I and III**) and scenarios for landfilling of MSW in terms of impact of landfill gas and leachate treatment options (**paper II**).

Table 1:Summarised scenarios of MSW management and landfilling

Scenario	Material recycling	Biological recycling (composting)	Incineration	Landfilled waste	Remarks
2000 Base scenario	4%	4%	0	92%	**Papers I, III**
2020 Scenario I Recycling + Incineration	27%	15%	45%	13%	**Papers I, III**
2020 Scenario II Recycling + Composting	27%	37%*	0	36%	**Papers I, III**
Landfill gas management	Option 1 Flaring	Option 2 Flaring	Option 3 Heat/electricity generation	Option 4 Electricity generation	**Paper II**
Collection (% of generated)	8,5%	75%	75%	75%	**Paper II**
Flared (% of collected)	100%	100%	0%	0%	**Paper II**
Utilized (% of collected)	0%	0%	Electricity 35% Heat 60%	Only electricity production 39%	**Paper II**
Leachate treatment	**BOD_7**	**COD**	**P_{total}**	**N_{total}**	**Paper II**
	Treatment efficiency - removal rate of main pollutants (%)				
Option 1 – no treatment of leachate	0	0	0	0	**Paper II**
Option 2 – aerobic treatment with activated sludge	63%	37%	17%	30%	**Paper II**
Option 3 – reverse osmosis	96%	90%	90%	95%	**Paper II**

Two sensitivity scenarios were developed based on the assumptions that mass-burn incineration of MSW will replace the major part of landfilling of wastes in near future: (i) scenario 2015, where the amount of MSW landfilled was reduced to 10% with the baseline of 2010 and major part of wastes were incinerated and (ii) scenario 2020, where only 5% of non-combustible material is assumed to be landfilled (Table 2).

Table 2:Scenarios of MSW mass-burn incineration

Scenario	**Material recycling**	**Biological recycling (composting), MBT**	**Incineration**	**Landfilled waste**
Scenario 2010	23%	12%	0%	57%
Scenario 2015	25%	10%	55%	10%
Scenario 2020	25%	10%	60%	5%
Landfill gas management	**Collection (% of generated)**	**Flared (% of collected)**	**Utilized (% of collected) in gas-drived motors**	**Utilized (% of collected) in boilers**
Option 1	50%	20%	30%	50%
Leachate treatment option	**BOD_7**	**COD**	**P_{total}**	**N_{total}**
	Treatment efficiency - removal rate of main pollutants (%)			
Option 2 – treatment with municipal sewage waters	98%	96%	67%	85%

Scenario 2010 describes the current situation in Estonian waste management sector. Forecast scenarios for 2015 and 2020 were developed following the Regional Waste Management Plan and the requirements of EU Directives. As it can be seen from Table 2, upto 2015, significant reduction of landfilled MSW is awaited, partly due to the mass-burn incineration unit that at Iru Power Station that utilises 50–55% of all MSW. The part of material recycling, composting, and MBT will remain approximately at the same level and will not exceed 35% for MSW.

Option 1 for landfill gas management was assumed as follows: 50% of the generated gas will be collected, which is the maximum collection rate for the MSW conventional landfills and much higher rates can be achieved only in biocell (Sundqvist, 1999; Finnveden et al., 1992). About 20% of the collected gas will be flared, 30% can be used in gas-driven motors for electricity and heat production, and 50% will be used in boilers for heat production.

The leachate Option 2 was assumed to be purified with the municipal sewerage at wastewater treatment plant. The efficiency of the treatment is presented in Table 2.

The composition and amounts of landfilled MSW are assumed to be minimised to 10% in 2015 (scenario 2015). According to that scenario, the landfilled MSW was assumed to compose 5% of organic part and 5% of non-combustible wastes. The scenario for 2020 predicted that landfilling would consist only 5% of non-combustible wastes.

4. RESULTS AND DISCUSSIONS

The environmental assessment study results showed that intensive incineration scenarios with energy recovery should be favoured compared to other waste management options where energy is not recovered when aiming to reduce GHG emissions (**papers I and III**).

The environmental impact of the emissions from landfills was assessed based on the LFG potential and leachate treatment technologies (**paper II**). The best result is achieved when LFG is used for heat and electricity production. Then, the avoided impact from energy recovery is greater than direct impacts of GHG emissions from landfill. The results of LCA support the fact that leachate treatment with reverse osmosis has the best environmental performance compared to other leachate treatment technologies. The long-term environmental impact when using RO is very small in case of nitrogen, ammonium, and other pollutants.

Developed sensitivity scenarios on the environmental impacts to global warming, photo-oxidant formation, acidification and eutrophication for 2015 and 2020 were based on the assumption that most of the now landfilled MSW will go for mass-burn incineration (Figure 7).

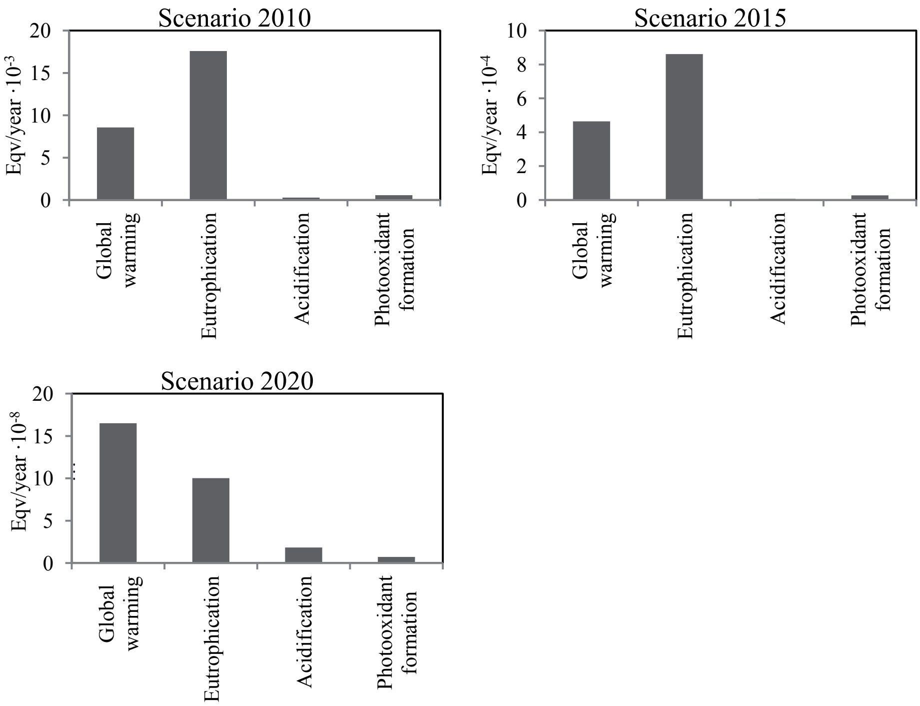


Figure 7[1]: Normalised results of impacts from landfilling of MSW in 2010,2015 and 2020

The study results indicate that minimisation of landfilling of MSW significantly reduces emissions to the air and water in all of four impact categories as presented in Figure 7. The main impact to global warming and eutrophication was found in 2010. According to the prognoses made, the environmental impact for 2015 will be reduced by ten times equivalents of emitted pollutants per year, and by ten thousand times equivalents/year for 2020. It can be explained by the changes in waste composition and significant reduction in quantity of landfilled waste for those years.

Recovery options of landfill gas are described in the methodology section in Tables 1 and 2. All nitrogen from the organic material is mainly transformed to the ammonia

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[1] *The impact of life cycle contains four impacts expressed as carbon dioxide equivalents (global warming),SO_2 equivalents (acidification), PO_4 equivalents (eutrophication) and ethylene equivalents (photooxidant formation). Normalisation consists of the division of these emissions with the total emission in each category over an entire year in Estonia. The resulting relative sizes of the impacts are in the order of 10^{-3} to 10^{-8} of the total annual impacts in different scenarios.*

or ammonium (NH_3/NH_4) in the leachate water and contributes to eutrophication. The photoxidant formation category is presented mainly by NMVOC. The environmental impact of operating landfill to the acidification processes is relatively low and the main polluting substances are sulphur dioxide, dinitrogen oxide and hydrogen chloride, which contribute less than 1% of all other impact categories (Assamoi and Lawryshyn, 2012).

Intensive incineration of MSW that is one of the basic assumptions for scenarios 2015 and 2020 will lead to the generation of rather large amounts of solid and liquid residual materials. According to Sabbas et al. (2003), typical residues from MSWs are bottom ash with grate sifting (20–30% by mass of the original waste on wet basis), particulate matter (15%), and fly ash (1–3%). Bottom ash can be reused, mainly in road construction or as aggregate for concrete products (Sabbas et al., 2003; Karagiannidis et al., 2013). Fly ash is classified as a hazardous material and has to be pre-treated before any use. Another option is landfilling at special hazardous landfill sites. The bottom ash can also be used as a landfill cover (Travar et al., 2009).

4.1 Landfill gas

Produced landfill gas consists mainly of methane (55%) and carbon dioxide (45%) (Assamoi and Lawryshyn, 2012) of which 50–75% can be utilised for energy recovery during the operating period of landfill (Sundqvist, 1999; USEPA, 2008). The collection rate of landfill gas starts to slow down after the closure of the landfill site (Niskanen et al., 2012). In the current study, the landfill gas collection efficiency was assumed to be 50%. It was also assumed that 30% of the collected gas will be used in gas-driven motors for electricity and heat production, 50% in boilers for heat production, and 20% will be flared. According to the Estonian Statistics for 2011, 95% of electricity in Estonia was produced from oil-shale and 5% from wood. At the same time, 50% of heat was produced from wood, 33% from natural gas and 17% from oil-shale.

The heat and electricity produced by landfill gas utilisation substitutes fossil fuel (oil-shale, wood and natural gas) in the background system (**paper II**). It could save up to 4040 tonnes of GHG emissions expressed in CO_2/eq when following the scenario for 2010. Saved emissions from mass-burn incineration of MSW were 4770 tonnes CO_2/eq a year. Therefore, it can be concluded that intensive incineration of MSW with energy recovery will have a better energetic potential compared to the landfill gas utilisation.

Due to the diversion of wastes from the landfill in scenarios 2015 and 2020, the GHG emissions in a hundred year perspective will be negligible (Figure 7). The composition of wastes to be landfilled plays a crucial role and the after-care period of gas generation will be longer if the organic content of landfilled wastes remains high (Niskanen et al., 2012). In the studied case, the composition of landfilled waste in 2015 was assumed to be 50% of organic fraction and 50% of other non-combustible wastes;

for 2020, only non-combustible wastes were assumed to be landfilled. Waste amounts for the future scenarios were calculated based on the amount of wastes landfilled in 2010 (267365 tonnes) (Estonian Statistics, 2012). It was assumed that 10% (32084 tonnes) and 5% (16042 tonnes) of MSW will be landfilled in 2015 and 2020, respectively, compared to the baseline in 2010. Such rapid decrease of landfilled wastes will probably lead to significantly lower collection efficiencies of the LFG in the studied landfills after the year 2020. The LANDGEM software (**paper II**) was used to investigate the total generation potential of LFG, based on the WAMPS model scenario of considerably reduced amounts of landfilled wastes by 2020. The results of the simulation are presented in Figure 8.

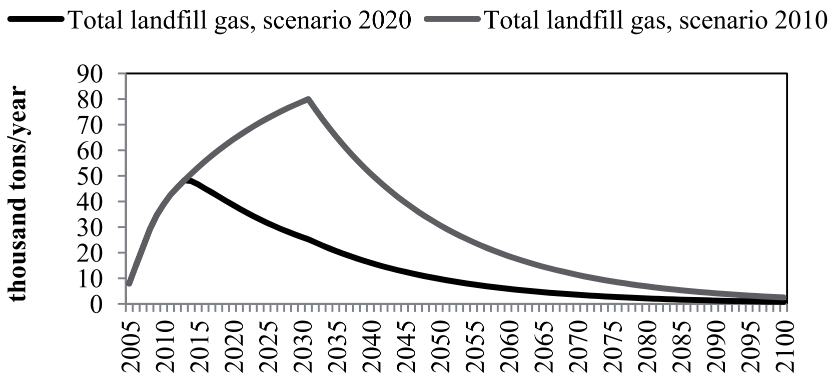

Figure 8: Potential landfill gas production, using LANDGEM software.

The graph presents the potential production of landfill gas, taking into account two scenarios:

(i) Scenario 2010 that represents the average Estonian sanitary landfill, which started operating in 2004 and will be closed in 2030. Since 2012, the amount of landfilled MSW was assumed to be constant and on par with 2011.

(ii) Scenario 2020 represents the amounts of landfilled MSW from 2004–2011 as in previous scenario and prognoses were made for the years 2015 and 2020, when the amounts of landfilled wastes assumed to be significantly lower, forming only 10% and 5% of the 2010 level, respectively.

The results showed that, if landfilling still remains the dominant method for waste handling, the amounts of LFG will increase until 2032 and slow down after the closure of the landfill in 2030. In the scenario of 2020, the peak of the LFG production will be achieved in 2015 and then the production rate will drop down. It can be concluded that diversion of waste away from landfill will lead to significant (50%) reduction of total

LFG generation. Therefore, LFG still has to be managed for at least 30 years after the closure of the landfill following the scenario of 2010. Similarly, landfill gas should be managed at least till 2040 if significant reduction of landfilled waste occurs (scenario 2020).

4.2 Landfill leachate

The landfill leachate is generated by percolating rainwater through the waste layers in the landfill. Main pollutants of leachate include (Kjeldsen et al., 2002): dissolved organic matter quantified as Chemical Oxygen Demand (COD) or Total Organic Carbon; inorganic macrocomponents, e.g., calcium (Ca^{2+}), magnesium (Mg^{2+}), sodium (Na^{+}), potassium (K^{+}), ammonium (NH_4^{+}), iron (Fe^{2+}), manganese (Mn^{2+}), chloride (Cl^{-}), sulfate (SO_4^{2-}) and hydrogen carbonate (HCO_3^{-}); heavy metals, e.g., cadmium (Cd^{2+}), chromium (Cr^{3+}), copper (Cu^{2+}), lead (Pb^{2+}), nickel (Ni^{2+}), and zinc (Zn^{2+}); xenobiotic organic compounds (XOCs) such as aromatic hydrocarbons, phenols, chlorinated aliphatics, pesticides, and plasticisers.

Leachate generation was analysed in the three studied landfills (Jõeläthme, Uikala and Väätsa) based on field measurements in 2007 (Loigu, 2010). To characterise the leachate according to the landfill development phases, BOD/COD ratio should be considered (Barlaz and Ham, 1993; Reinhart and Grosh, 1998; Kjeldsen et al., 2002). On the studied landfills, BOD/COD ratio was found to be around 0.3, which indicated that in the initial methanogenic phase in 2007, Estonian landfills were characterised mainly by increasing methane production rate and conversion of acids to methane and carbon dioxide. It can be assumed that the next phase will be achieved when the BOD/COD ratio drops below 0.1, stable methanogenic phase is attained and the methane generation rate reaches its maximum (Kjeldsen et al., 2002).

The detailed composition of leachate in the studied landfills is presented in **paper II**. The toxicity tests revealed quite interesting results. Concentration of heavy metals was relatively low, which can be explained by a very high sulphur content in landfill leachate and also high conductivity that leads to precipitation of metals from leachate water. At the same time, a high concentration of phenols was found in two landfills. The high content of sulphur and phenols can be explained by landfilling of oil-shale incineration ashes, which is a by-product of electricity production.

Once created, landfill leachate has to be pretreated. There are various treatment methods for landfill leachate, and some of them are applied in Estonia also. The assessment of the leachate treatment methods used in Estonia is provided in **paper II**. The results showed that the highest level of the treatment efficiency was reached by using reverse osmosis technology, which is the most expensive option (Loigu, 2010). After the leachate is filtrated through the membrane, permeate and concentrate are generated. The concentrate was landfilled and the effectiveness of the treatment was assessed only from permeate.

4.3 PAYT model as a possibility for waste prevention and minimisation

Application of Pay-as-you-throw (PAYT) model into current waste management system in Estonia was assessed in **paper IV**. Results showed that the implementation of PAYT into existing waste management system is not feasible at the moment. The main constraints can be considered as follows:

- The lack of financial resources to invest in the initial stage of PAYT scheme development including equipment for the trucks and lockers for the bins. Preliminary assessment of the costs for one rural municipality showed that the cost for emptying the container would increase by approximately 20–45% based on estimated investments over the coming three years and assuming that the production of waste would decrease by 20% in the first year and 15% thereafter.
- The lack of interest of municipalities to change the existing scheme. The current waste management scheme shows that the rights of local authorities for choosing the disposal stage of MSW are limited. After the tendering procedure, all administrative work including waste handling is the responsibility of the private company that wins the tender.
- Mass-burn incineration of MSW at Iru power station shows that the scheme of incineration of waste does not support a decrease in waste production and separate collection. However, the requirements of the EU Packaging Directive (EC, 2004) with minimum packaging recovery target 60% for packaging waste and the Landfill Directive (EC, 1999) targeting reduction of the amount of landfilled biodegradable municipal waste by 35% by 2020 (baseline 1995) have to be achieved.
- The lack of legal support. The legal right of waste transportation companies to bill according to the PAYT model must accompany a legally accepted waste amount measurement system.
- Environmental awareness among citizens is still very low. To sufficiently inform citizens, local governments should allocate more resources. Achieving transparency among the public obviously helps to combat apprehensions about PAYT, such as concerns regarding increased illegal dumping, perception that the introduction of PAYT will result in an additional financial burden for residents, or their natural resistance to change.
- Economically, people still remain less motivated to change the existing waste charge model. In Tallinn, for example, waste management services cost annually only 0.02–0.10 €/m^2 of apartment, which forms less than 1% of family income and only 3–5% of all communal costs.
- Generally, significant changes are required to create favourable conditions for more efficient and sustainable waste management. More in-depth investigation

of the options for PAYT system implementation with its practical adoption in pilot municipalities is recommended.

5. CONCLUSIONS AND RECOMMENDATIONS

The study aimed to assess:

- Environmental impacts of landfilling of MSW in terms of GHG emissions, landfill gas options and leachate treatment technologies;
- Perspectives for MSW incineration; and
- Possibilities for PAYT implementation in Estonia.

As a result of the case study in Estonia, it can be concluded that better management of municipal wastes, especially diversion of municipal wastes away from landfills, could significantly reduce the emissions of GHGs. Material recycling and incineration with high rates of energy recovery should be favoured compared to other waste management options where energy is not recovered (e.g., composting). A MSW mass-burn incineration plant at Iru power station in Estonia started to operate in 2013 with the capacity to incinerate 220,000 tonnes of wastes annually. The planned capacity of heat production is about 50 MW and electricity production is 17 MW. Thus, about 70 million m^3 of natural gas could be substituted in the future.

As a result of environmental assessment of landfilling, it is important to stress that the collection rate and proper utilisation of collected gas plays an essential role in minimisation of the environmental impact of GHG emissions. The importance of gas collection and control measures has been highlighted in several studies (Gheewala and Wanichpongpan, 2007; Manfredi et al., 2009; Daamgard et al., 2011; Nikannen et al., 2012). The leachate treatment technology should be chosen based on leachate composition and local conditions. The study results showed that the most efficient method for treatment of leachate is reverse osmosis adopted at Uikala landfill. However, while designing the treatment technology, proper collection and treatment of leachate for a relatively long time period (up to 40 years) should be taken into account. Therefore, the economic aspects of different treatment technologies are important and should be considered.

The perspective of incineration of MSW in Estonia was assessed by conducting several LCA studies. It can be concluded that thermal treatment of MSW in large-scale WtE facilities has a relatively good outlook in the Baltic States. Although, the initial investments are relatively high, the favourable conditions in the energy sector allow WtE facilities to treat municipal waste at a relatively low cost. Therefore, it can be expected that WtE provides an environmentally and economically efficient way to meet the stringent EU waste management targets. However, large scale municipal waste incineration has to be discussed within the context of an overall waste management strategy, rather than as a single option.

Results showed that the waste management sector does not directly facilitate PAYT implementation at the moment. Significant changes are required in the legislation to create necessary support from governmental level and to change the collection scheme and charging mechanisms of waste handling. Additional efforts of local authorities are needed to enhance the environmental awareness of people. Future work can be directed to create the pilot-scale project of PAYT implementation in a selected urban as well as rural municipality.

It should be noted that the results of this study focused only the eutrophicating impacts, global warming, acidification and photo-oxidant formation related to the studied leachate and LFG management options and thus do not express the total environmental impacts of the entire landfill system. Therefore, it is recommended that further LCAs investigate other relevant impact categories also. It is especially relevant for toxicity-related impacts, since it was found that the content of toxic substances in leachate was relatively high. However, this can change significantly in the future.

It can be expected that due to increasing recovery and waste incineration, the amount and composition of waste will change dramatically in the near future. Results of the study showed that after such changes, further collection of the landfill gas will not be relevant anymore and the environmental impact in terms of methane formation will be significantly lower. Incineration of waste generates bottom ash and fly ash. Therefore, it is necessary to study the bottom ash utilisation methods and find appropriate management options for Estonian conditions.

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And at last, I would like to thank my family: husband, parents, and my sister for their patience and support, giving me opportunity to finalise my work.

ABSTRACT

Nowadays, MSW management is very rapidly changing. Stringent waste management laws have forced to improve existing technologies and test new practices. In Baltic States, including Estonia, the landfilling has remained the dominant method of MSW utilization till now. Therefore, the objectives of this PhD thesis were the environmental assessment of impacts from existing landfills with the aim to suggest the best available option for landfill gas utilisation and leachate treatment, to assess the alternative options for MSW utilisation, e.g., mass-burn incineration of wastes, and evaluate the possibilities for reduction of MSW generation at households.

For environmental assessment, a life-cycle tool was chosen. Life-cycle assessment is one of the widely used tools for assessing the environmental impact of a product or service during its entire life-cycle, from raw material excavation till disposal stage. Recently, the life-cycle assessment has gained recognition in waste management planning. For this purpose, different models were developed, such as WAMPS. In this research, the landfilling sub-model of WAMPS was adjusted to be used in Estonia.

Research was conducted in a series of case studies in the period of 2009–2013. The first study assessed the environmental impacts from Jõelähtme landfill (serves Tallinn city and its surroundings), Väätsa landfill (serves Western region of Estonia) and Uikala landfill (serves East region of Estonia). Results of the first study indicate that LFG utilisation for energy recovery is an essential part of the treatment system since it leads to saved emissions and avoids global warming impact potential. Saved emissions were found as 4040 tonnes CO_2/eq annually in comparison with the reference system without gas collection. Therefore, the measures, which combine LFG collection with energy generation, should be preferred to treatment in flare. Among the compared treatment options of leachate used by Estonian landfills, results of both direct measurements in the studied landfills (Väätsa, Uikala, Jõelähtme) and LCA modeling support the fact that leachate treatment with reverse osmosis has the best environmental performance compared to other leachate treatment technologies. The treatment efficiency of leachate was 96% BOD_7, 90% COD, 90% P_{total} and 95% N_{total}, respectively.

Another case study about the perspective of MSW incineration at Iru Power Station in Estonia was performed. Results show that waste incineration with energy recovery can partly offset the emissions to about 4770 tonnes $CO_{2/}$eq annually that occurred when energy was produced from fossil fuels. This is an important aspect in the context of climate change, since the oil shale combustion technology currently used in Estonia to generate electricity has a very high climate change impact. This means that in case a high share of recyclable waste fractions are sent to material recycling and the maximum amount of remaining waste is incinerated with energy recovery, municipal

waste management scenario should be preferred from the environmental impact point of view.

The possibility of minimisation of MSW generation by introducing the PAYT system (to pay a variable amount depending on the quantity of waste generated) was discussed with local authorities in Estonia. The results of the assessment study showed that half of the respondents among local municipalities deemed the effective implementation of the PAYT system as a possibility through significant explanatory work within the community. The lack of financial resources and demand for change in the policies concerning the waste management sector could hinder this process.

The environmental impact from MSW landfilling is significant. The study results revealed that optimisation of LFG collection with further usage for energy recovery and proper leachate treatment technique can lead to saved emissions when compared to the background system where fossil fuels such as oil-shale are used for energy production. Another option for minimisation of environmental impact is reduction of landfilled wastes amount and changes in their composition through implementation of mass-burn incineration of MSW. Further reduction of MSW generation at households can be achieved by implementing the PAYT system into current waste management process.

KOKKUVÕTE

Tänapäeva jäätmemajandus on väga kiiresti muutuv valdkond. Range seadusandlus sunnib täiustama kasutatavaid jäätmekäitlustehnoloogiad ja testima uusi meetodeid. Balti riikides, sh Eestis oli kuni 2013. aastani olmejäätmete ladestamine prügilasse kõige valdavam käitlusmeetod. Järgneb jäätmete masspõletamine. Seetõttu oli käesoleva doktoritöö eesmärgiks olmejäätme prügilate keskkonnamõju hindamine ning parima võimaliku alternatiivi pakkumine prügilagaasi utiliseerimiseks ja nõrgvee puhastamiseks. Ka võrreldi olmejäätmete ladestamise ja Iru elektrijaamas toimuva jäätmete masspõletuse keskkonnamõju ning hinnati võimalusi olemjäätmete tekke vähendamiseks majapidamistes.

Keskkonnamõju hindamisel kasutati olelusringi meetodit, mis on valdav toote või teenuse keskkonnamõju hindamisel alates toorme kaevandamisest kuni toote utiliseerimiseni. Viimastel aegadel on olelusringi hindamine leidnud kasutamist ka jäätmemajanduse planeerimisel. Selleks on välja töötatud erinevad mudelid, sh jäätmekäitluse planeerimise mudel – WAMPS. Käesolevas doktoritöös rakendati WAMPSi prügila moodulit Eesti oludes.

Töö viidi läbi, tuginedes erinevatele 2009.–2013. aasta juhtumiuuringutele. Esimese juhtumiuuringu käigus hinnati Jõelähtme (teenindab Tallinna linna ja selle ümbrust), Väätsa (teenindab Lääne-Eestit) ja Uikala prügila (teenindab Ida-Eestit) keskkonnamõju. Uuringu tulemused näitavad, et prügilagaasi utiliseerimine energiakasutuses võimaldab vähendada gaasi emissiooni ning vältida seeläbi mõju globaalsele soojenemisele. Välditud heitgaaside kogus on kuni 4040 tonni/CO_2ekv aastas võrreldes süsteemiga, kus prügilagaasi ei koguta. Seetõttu tuleks kasvuhoonegaaside vähendamise eesmärgil prügilagaasi põletamisele eelistada meetmeid, mis on suunatud prügilagaasi kogumisele ja sellest energia tootmisele. Nõrgvete puhastamise tehnoloogiate võrdlus ja olelusringi hindamise modelleerimise tulemused Eesti prügilates (Väätsa, Uikala, Jõelähtme) näitasid, et nõrgvee puhastamine pöördosmoosiga on osutunud teiste nõrgvee puhastamise tehnoloogiatega võrreldes keskkonnasäästlikumaks. Nõrgvee puhastuse efektiivsus ulatub vastavalt 96% BHT, 90% KHT, 90% $P_{üld}$, 95% $N_{üld}$ järgi.

Teise juhtumiuuringu eesmärgiks oli hinnata olmejäätmete põletamise perspektiivi Iru elektrijaamas. Tulemused näitavad, et jäätmepõletus koos toodetud energia kasutamisega võib vähendada kasvuhoonegaaside emissioone kuni 4770 tonni$CO_{2/}$ekv aastas võrrelduna olukorraga, kus kogu energia toodetakse fossiilsetest kütustest. Seega on tegemist olulise aspektiga kliima muutuste kontekstis, arvestades käeoleval ajal kasutatavat põlevkivi põletamise tehnoloogiat, mille käigus emiteeritakse suures koguses kasvuhoonegaase. Seega tuleb olmejäätmete ladustamisele eelistada stsenaariumi, kus suur osa korduvkasutatavast jäätmematerjalist suunatakse taaskasutusse ning maksimaalne kogus ülejäänud jäätmetest põletatakse ning toodetud energia leiab kasutamist.

Olmejäätmete tekke vähendamise võimalusi hinnati Eesti omavalitsustes PAYTi skeemi (äraantavate jäätmete maksustamine koguse järgi) võimaliku rakendamise kaudu. Analüüsitulemused näitavad, et pooled küsitlusele vastanud omavalitsustest pooldavad PAYTi juurutamist, pidades seda üheks võimaluseks jäätmekäitluse edasiarendamisel koostöö ja kommunikatsiooni parandamiseks ühiskonnaga. Skeemi juurutamise oluliseks takistuseks on rahapuudus ning vajadus jäätmemajandusealase seadusandluse muutmiseks.

Olmejäätmete prügilasse ladestamise keskkonnamõju on suur, kuid doktoritöö tulemused näitavad, et prügilagaasi kogumine edasise kasutamisega energia tootmiseks ja nõrgvee puhastuse optimeerimine aitavad säästa heitkoguseid võrreldes taustsüsteemiga, kus elektri tootmiseks kasutatakse põlevkivi. Teiseks keskkonnamõju vähendamise võimaluseks pakuti prügilasse ladestatavate jäätmete koguse minimeerimist ja nende koostise muutmist, rakendades alternatiivina jäätmete masspõletust. Täiendava jäätmekoguse vähenemise majapidamistes võib tagada PAYTi süsteemi juurutamine.

APPENDIX I ORIGINAL PUBLICATIONS

PAPER I

Moora, H., **Voronova, V**., Reihan, A., 2009. The impact of Municipal Solid Waste Management on Greenhouse Gas Emissions in Estonia. Walter Leal Fihlo (Toim.). Interdisciplinary Aspects of Climate Change (311 - 325). Frankfurt am Main: Peter Lang Publishers House.

Chapter 15

The Impact of Municipal Solid Waste Management on Greenhouse Gas Emissions in Estonia

Harri Moora, Viktoria Voronova, Alvina Reihan

Abstract

Waste management has an influence on the greenhouse gas (GHG) formation. The emissions of greenhouse gases vary between the EU countries depending on waste treatment practices and other regional factors such us composition of waste. The aim of this paper was to examine, from a life-cycle perspective, Municipal Solid Waste (MSW) management in the context of greenhouse gas formation and to evaluate the possible reduction of climate change potential of alternative waste management options in Estonia. Air temperature as the main climatic parameter is analysed in the context of climate change. The paper summarises the results of a case study in Estonia, assessing the climate change impact by 2020 in terms of net greenhouse gas emissions from two possible management scenarios. The paper also provides updated information on the composition and generation of MSW in Estonia. As a result it can be concluded that better management of municipal waste and diversion of municipal waste away from landfills could significantly reduce the emissions of GHG and, if high rates of recycling and incineration with energy recovery are attained, the net greenhouse gas emissions may even become negative.

1. Introduction

It is emphasised worldwide that the main climatic parameters such as air temperature, precipitation, etc. have changed and the changes are related to the increase of greenhouse gases (IPCC 2007). Estimation of climate change impacts on different ecosystems is based on different rates of GHG emissions. Therefore, it is necessary to minimise the GHG emissions resulting from waste management, in order to enhance the use of low carbon economy and adaptation measures to ensure ecosystem sustainability. GHG emissions represent a considerable environmental effect from the management of municipal solid waste. The GHG emissions from the waste sector depend on a number of factors including waste generation, waste composition and used waste management practices.

The generation of MSW in the European Union (EU) has increased steadily. The MSW generation and treatment options vary significantly in the old Member

States (EU-15) and the new Member States (EU-12). The EU-15 landfilled less than 60% of the municipal waste in 2005, while the majority of the EU-12 landfilled most of the MSW (more than 80%) (ETC/RWM 2008). The new EU Member States undergo currently a rapid economic development, resulting in a significant increase of waste quantities, while their waste management systems still require much effort to be adjusted to the European state-of-the-art. Therefore, in the context of climate change the diversion of municipal waste away from landfills and choosing optimal waste recovery practices is especially important in the new Member States including Estonia.

Through iterative examination of various waste management technologies and treatment alternatives, life cycle based methodologies can help to identify optimal environmental solutions for managing the waste. Life-cycle information can also help to identify the benefits and trade-offs of different waste management options in terms of direct GHG emissions (from landfills, incineration plants, recycling and collection of the waste) and indirect GHG emissions that are associated with the extraction and processing of primary resources or fossil fuels versus those associated with recycling or incineration operations. In the new Member States the life-cycle assessment (LCA) models are not widely used for waste management planning and calculation of GHG emissions. The major limitation of using LCA in waste management planning in these countries is the lack of relevant data and knowledge of the analysed systems. Also it is difficult to compare specific local waste management information and data with the data used by LCA models developed for other countries.

The main aim of this paper is to examine the climate change impact in Estonia in terms of GHG emissions from MSW management between years 2000 and 2020. Two most feasible waste management options for Estonia, material recycling with biological recycling in terms of composting and material recycling with intensive incineration, were compared in terms of their possible contribution to climate change. The paper also provides information on MSW quantities and composition in Estonia.

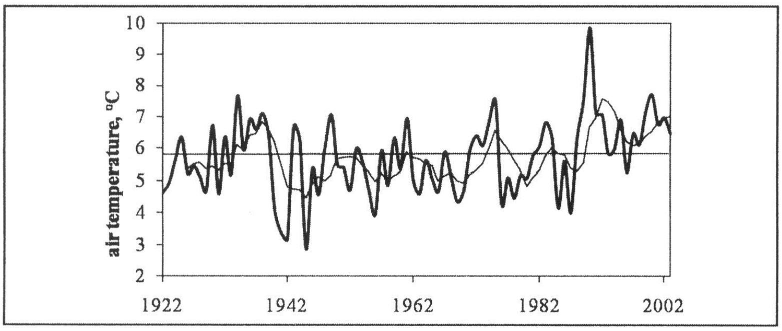
Most of the earlier studies have focused on the calculation of direct GHG releases associated with municipal waste management. In this paper calculations are based on life-cycle information and the calculated emission levels include not only direct emissions from MSW management, but also the 'avoided' emissions from material recycling and energy recovery. The study is based on the LCA model for waste management planning – WAMPS (Moora et al. 2006).

As such the results of the study also provide additional country specific background information and data for an ongoing discussion about the implementation of waste hierarchy and provisions of the EU Waste Framework Directive.

2. Climate change aspects in Estonia

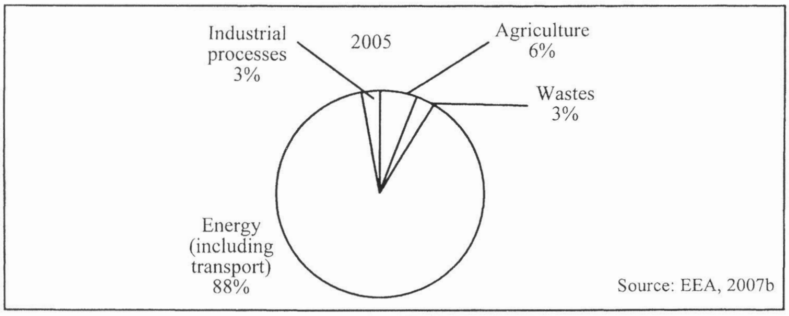
Investigations of climate features in Estonia have a long tradition (Kant 1927; Kirde 1940). While earlier investigations were focused mainly on climate impacts on agricultural production, the present climatological studies are focused on the environmental and human impact aspects. Recent studies show different variations in climate parameters, such as increase in air temperature and precipitation in autumn and winter, decrease of snow and ice duration (Jaagus 1998) and a decrease in reflected radiation in spring (Russak 1998), increase of frequency of extreme events such as storms, inundations, droughts, etc. The most intensive warming (figure 1) took place in the winter and spring seasons during the last forty years with significant growth in the continental part of Estonia (Reihan 2008). All these changes are in good agreement with those reported by the Intergovernmental Panel on Climate Change (IPCC 2007) and are of a great importance for the development of any ecological management plans including waste management.

Fig. 1: Variation of the long-term air temperature with 5 years moving average in Pärnu MS (South-West Estonia); the straight line indicates the long-term mean value



Estonia has signed and ratified the Kyoto Protocol, according to which emissions of GHG must be reduced by 8% in 2008-2012, compared to the baseline year 1990. All EU Member States report regularly their direct and projected GHG emissions in order to apply policies and measures to reach this target. The methodology used for the estimation of GHG emissions follows so called IPCC guidelines (IPPC 2006) produced by international expert groups for the IPCC and are followed also by Estonia to calculate the national yearly GHG emissions. These calculations focus mainly on direct GHG emissions of different sectors. According to the latest GHG emission calculations, which were made for the 4th National Communication under the United Nations Framework Convention on Climate Change (UNFCCC) in 2006, the total emission level in 2005 was 20.7 Mt of CO_2 equivalents. The main contributors to GHG emissions in Estonia are energy supply and use together with transport (88%) (figure 2). The electricity production in Estonia is mainly based on an Estonian specific fossil fuel – oil shale. The production of electricity from oil shale entails higher emissions of CO_2 than most of the other fossil fuels used in EU (OSELCA 2006). The share of waste management (mainly CH_4 emissions from landfills and waste water sludge treatment) is only 3% of the total GHG emissions. However, waste management is an important source of GHG reductions. Modern waste management is very closely linked to energy production, therefore, when planning waste management systems the context of climate change should be taken into account, since it contributes to the meeting of the Kyoto targets.

Fig. 2: Greenhouse gas emission by sectors in Estonia



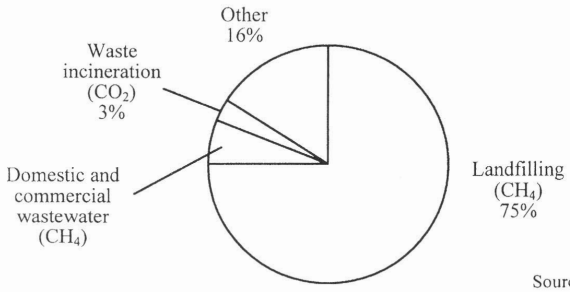
3. Greenhouse gas emissions from waste management

The GHG emissions from the MSW management are sum of the direct emissions (from landfills, incineration plants, recycling operations and collection of the waste) and indirect emissions. Indirect emissions arise from the energy and secondary products produced when incinerating and recycling waste replace energy production from fossil fuels and the use of raw materials for plastic, paper, metals, etc

Direct emissions from the waste management sector in the EU-15 contributed by 2,6% of the total GHG emissions in 2005 (EEA 2007a). The key sources of waste-related greenhouse gas emissions are illustrated in Figure 3.

As shown in Figure 3, the main sources of direct GHG emissions from waste management are *landfills*. It could be estimated that GHG emissions from landfills are larger in new Member States including Estonia due to a bigger share of landfilling in these countries. The landfilling of biodegradable waste results in the formation of landfill gas which contains mainly methane (CH_4). In addition to CH_4, landfills also produce carbon dioxide (CO_2), non-methane volatile organic compounds (NMVOC) and a small amount of nitrous oxide N_2O, nitrogen oxide NO_x and carbon monoxide CO (IPPC 2006). The origin of CH_4 in landfill gas is the degradation of organic waste under anaerobic conditions. Compared with CO_2, the global warming potential of CH_4 is 21 times higher over a time horizon of 100 years. In the modern landfills the landfill gas is partly (usually 10-50%) collected and either disposed by flaring or used as a fuel for energy production. Contrary to GHG emissions from other waste management practices (e.g incineration or recycling), landfill GHG emissions are characterised by the large time lag of emissions (Sundqvist 1999).

Fig. 3: Main sources of GHG emissions from the waste management sector in EU-15, 2005



Source: EEA, 2007a

The most widely practised alternative to landfilling is *mass-burn incineration* where MSW is burnt with little or no pre-treatment. The modern MSW incinerators are required to recover energy released by the combustion process. Incineration is a source of GHG emissions like other types of combustion process. GHG emissions are estimated by the carbon content of the incinerated waste material. The carbon content contributes mainly to CO_2 emissions and less to CO, CH_4 and NMVOC emissions. Exhaust gas cleaning or incineration technology does not influence CO_2 emissions. Emissions of CO_2 from incineration of biological waste material do not contribute to net GHG emissions and should therefore not be taken into account. Calculation of net CO_2 emissions from waste incineration is based on the fossil carbon content of the MSW. The net climate change impact of incineration depends on how much fossil carbon CO_2 is released – both at the incinerator itself (direct emissions) and in savings of fossil fuel from marginal energy sources displaced by incineration.

Recycling diverts components of the waste stream for reusing the materials. If the GHG emissions resulting from the separating and processing of the recycled material into new products are less than those generated while manufacturing the products from primary material, net savings of GHG emissions results.

Composting is an aerobic process and a large fraction of the degradable organic carbon (DOC) in the waste material is converted into CO_2. CH_4 is formed in anaerobic sections of the compost, but it is oxidised to a large extent in the aerobic sections of the compost. The estimated CH_4 released into the atmosphere ranges from less than 1 percent to a few per cent of the initial carbon content in the material (Beck-Friis 2001; Detzel et al. 2003; Arnold 2005). Composting can also produce emissions of N_2O. The range of the estimated emissions varies from less than 0.5 percent to 5 percent of the initial nitrogen content of the material (Petersen et al. 1998; Hellebrand 1998; Vesterinen 1996; Beck-Friis 2001; Detzel et.al. 2003). Poorly working composts are likely to produce more both of CH_4 and N_2O (e.g., Vesterinen 1996). This is the reason why the so-called home composting could have a relatively high climate change impact in terms of CH_4 emissions.

There are several other waste management options available. One of the most common practices is *mechanical-biological treatment* (MBT), which is a combination of mechanical and biological steps to reduce the amount and biological activity of the processed MSW. Pre-treatment of MBT prior to landfilling significantly reduces CH_4 emissions from the landfilled waste, compared with untreated MSW.

A common link between different waste management practices is the need for *collection and transport* from the source of the waste to the waste treatment/disposal facilities. It all has GHG impacts, mostly through the use of fossil fuels and associated emissions of CO_2. N_2O is also emitted from vehicle engines, but this has a minor impact (EC 2001).

4. Methodology

To calculate the GHG emissions from MSW management the important input data in terms of waste composition and generation in Estonia were studied. Also a review of possible future waste management technologies and practices was conducted. Based on collected information two possible waste management scenarios by 2020 were developed. As a basis for determination of GHG emissions from municipal waste management the LCA model for waste management planning WAMPS was applied. WAMPS model is intended to be applied during the waste management planning process to find optimal solutions and alternatives for waste management systems (Moora et al. 2006). WAMPS presents the environmental and economic consequences of different waste management scenarios in a life cycle perspective. WAMPS was developed by the Swedish Environmental Research Institute and is based on a more in-depth LCA model ORWARE (Sundqvist et al. 2002; Björklund 2000; Eriksson 2000).

WAMPS compares a waste management system with a background system. The waste management system can produce different products depending on the choice of treatment and recycling: heat, steam, electricity, vehicle fuel (biogas), compost, paper, plastic, metals, etc. In background system similar products are produced from virgin origin. When a product is produced from waste, it substitutes a product in the background system. Each waste product has an alternative in the background system with a virgin raw material source and a production process that is included in the model. In WAMPS different recovery options are compared with the background system and potentially 'saved emissions' are assessed. The background system consists of heat production (alternative to waste incineration and combustion of biogas and landfill gas), electricity production (alternative to waste incineration and combustion of biogas and landfill gas), vehicle fuel production (alternative to biogas), fertilizer production (alternative to compost and digestate) and production of materials (plastic, newsprint, paper packages, glass, steel, aluminium, etc). All these products from the background system can also be produced by different waste recovery methods.

The net emissions from the studied system are calculated according to:

$$E_{net} = E_{waste} - E_{Background}$$

E_{net}: Net emission (tonnes/year or kg/year)
E_{waste}: Emission from a waste process that produces a certain amount of product (tonnes/year or kg/year).
$E_{Background}$: Emission from the same amount of alternative virgin production in the background system (tonnes/year or kg/year).

This calculation can give negative net emissions. This means that the recycling method will give lower emissions or energy consumption than the corresponding virgin production. The global warming impact is calculated as CO_2-eqivivalent emissions. The basic functional unit in WAMPS is the waste generated within a specific region.

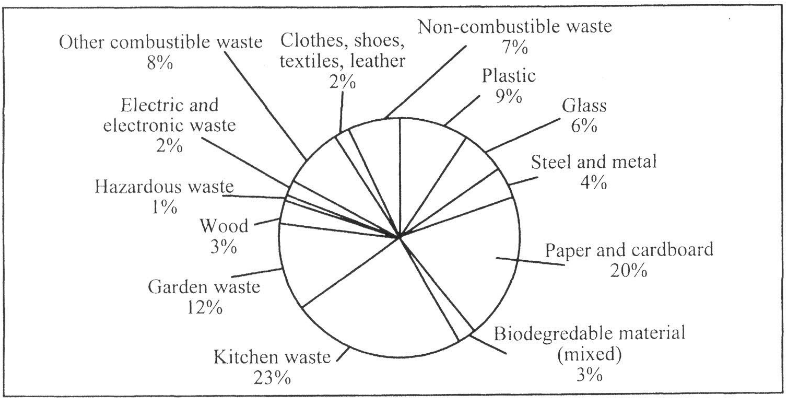
5. Case study

5.1 Waste composition and generation

The contribution made by the waste management sector to climate change is primarily determined by the volume and composition of municipal solid waste as well as the waste management options chosen.

The amount of MSW generated per person and its composition varies significantly in different Member States. Figures from different data sources cover different time periods and geographical locations and are rarely comparable. Most of the waste studies rely on official databases such us Eurostat. However, in most cases, especially in the EU-12, these data are notoriously unreliable. Since the lack of information on actual waste composition is one of the main barriers for waste management planning in Estonia, the composition of mixed municipal waste was explored by a countrywide sorting analysis of MSW in 2007/2008. The waste analysis was carried out as a part of the current study. Based on the results of the sorting analysis and the corrected statistical data, the most recent and accurate composition of MSW in Estonia was compiled (figure 4). The biodegradable fraction (organic waste, paper and cardboard, wood and textiles), which is the main source of GHG emissions, makes up a considerable share of municipal waste in Estonia (63%). The packaging waste amounts to 27% and the share of combustible waste to 80% of total MSW.

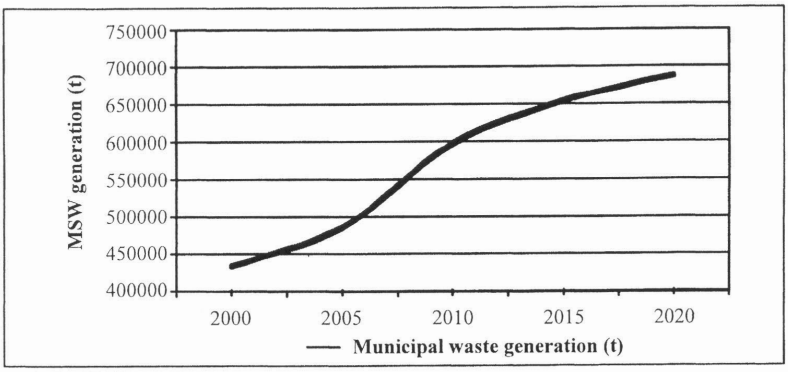
Fig. 4: The average composition of MSW for Estonia



The generation of MSW relates mainly to a number of economic activities and the size of the population. In general there is a strong link between GDP and waste generation. The quantity of municipal waste in Estonia has rapidly risen in line with the economic growth and growing consumption. According to specified statistical data, approximately 435000 tonnes of municipal waste (302 kg per person) were generated in 2000. In recent years the amount of municipal waste has been growing by 6% per year and reached 542000 tonnes in 2007 (402 kg per person). MSW volumes are expected to grow even more. Since the number of population is expected to remain roughly the same, economic growth or specifically private final consumption will be the key driving force behind the growing waste volumes in Estonia. The forecast of the municipal waste generation in this study is based on the future estimations of final consumption by households expressed in Purchasing Power Standard (PPS). The growth rate of municipal waste generation is expected to decrease in the coming years (average growth 3% per year) due to the slowdown of economic growth. It is expected that waste generation will stabilise after 2015 (average growth 1% per year). During the period 2000 to 2020 the generation of municipal waste is projected to increase by approximately 58% (figure 5). In 2020, the generation of municipal waste per person is estimated to be 509 kg (690000 tonnes). In general, this is in line with the projections made by the EEA

– in the new Member States, the generation of MSW is projected to increase by approximately 50% from 2005 to 2020 (ETC/RWM 2007).

Fig. 5: Projected growth in MSW generation between years 2000 and 2020



5.2 Scenarios

For the GHG emission calculation the waste management situation in 2000 was taken as a starting point or a base scenario. Two waste management scenarios were developed to analyse possible future alternative waste treatment options and their climate protection potential by 2020. Estonia, similarly to other new EU Member States, has to comply with the EU legal requirements and recovery targets for waste management. Since the pros and cons of waste incineration as a possible MSW management option were recently discussed in Estonia, the incineration-based scenario was compared with the scenario where legal targets are achieved with intensive biological recycling (composting) (see also table 1). Both alternative future scenarios are in compliance with the requirements and recycling targets of the following legal acts:

1. European Packaging Directive 2004/12/EC – minimum packaging recovery target 60% for packaging waste.
2. Landfill Directive 1999/31/EC – target amount of biodegradable municipal waste going to landfills must be reduced to 35% by 2020 (baseline 1995).

Base scenario (scenario 0)

In 2000, waste management in Estonia primarily involved landfilling of MSW (92% of the total MSW). There was no landfill gas collection in landfills at that time. Only a small amount of packaging waste (mainly PET-bottles and cardboard) was collected separately and sent to recycling. There was no centralised collection system for biodegradable waste. Approximately 17000 tonnes of biodegradable waste (mainly garden waste) were composted in the households (4% of the total MSW). It is assumed that the share of home composting will remain the same till 2020.

Material recycling with intensive incineration (scenario I)

Scenario I is a projection for 2020, where the dominant option of MSW management in Estonia is incineration. 45% of MSW is incinerated in the mass-burn incineration plant. This assumption is based on the plans to build an incinerator close to the capital of Estonia, Tallinn. The incineration plant is expected to start its operation in 2012. The incinerator complies with all the EU requirements and it is assumed that the gross efficiency of energy recovery from the incineration process will be relatively high (80%). A large amount of the heat could be utilised since Tallinn has large dwelling areas with district heating system. In this scenario increased amounts of recyclable materials (mainly packaging, paper, cardboard and metals) are separately collected and recycled to fulfil the recycling targets of the EU Packaging Directive. The recycling of material is expected to be 30%. As incineration is already contributing to the reduction of biodegradable waste, the share of biological recycling is not expected to exceed 15% of the total MSW. The centrally collected kitchen waste is composted using static composting method with forced aeration. Collected garden waste is composted in open windrows. Intensive material recycling and incineration leads to a relatively small amount of rest waste, which is landfilled (13% of the total MSW).

Material recycling with biological recycling in terms of composting (scenario II)

Scenario II is a projection for 2020, where the legal targets are archived by material and biological recycling. Also in this scenario material recycling is expected to amount to up to 30% of the total MSW. The Landfill Directive requirement to divert biodegradable waste away from landfilling, is fulfilled by increasing composting to 37% of the total MSW. An increased amount of wet biodegradable waste is composted using centralised reactor-composting method (without gas collection and energy recovery). It is assumed that the remaining waste will be deposited in a landfill.

The predictions about the future MSW generation presented in the earlier chapter were considered when developing the alternative future scenarios. It is assumed that waste composition remains the same during the studied period.

For both future scenarios it is also assumed that all landfills will be equipped with a landfill gas collection system at the latest by 2010 and the landfill gas recovery rate will increase up to 50% by 2020. Before 2010 the collected gas is flared and after 2010 it is used for electricity and heat production, which is substituting oil shale based electricity and natural gas based heat used for district heating. The energy produced in waste incineration is also substituting the electricity produced from oil shale and heat from natural gas.

Tab. 1: Municipal Solid Waste management scenarios

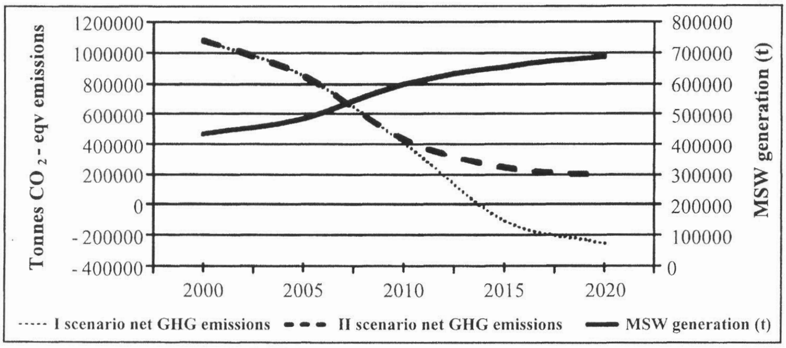
Scenario	Material recycling	Biological recycling (composting)	Incineration	Rest waste (landfilling)
2000 Base scenario	4%	4%	0	92%
2020 Scenario I Recycl + Incin	27%	15%	45%	13%
2020 Scenario II Recycl + Comp	27%	37%*	0	36%

*Landfill Directive: compliance with the target for 2020

6. Results

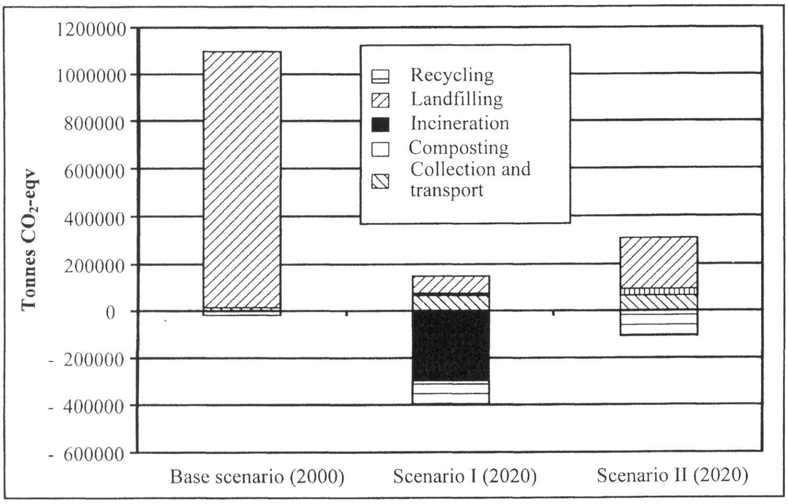
The results of the scenario analysis in terms of net GHG emissions are shown in Figures 6 and 7. The diagrams show the net GHG emissions from the waste management system minus saved emissions in the background system. When the emissions from the studied waste management scenario or waste management practice are less than the saved emissions in the background system then net result is negative.

Fig. 6: Emissions of net GHG from studied waste management scenarios, 2000-2020 (tonnes CO_2-equivalents)



The results of the case study indicate that net GHG emissions from the management of municipal waste in Estonia are projected to decline significantly by 2020 from a peak of around 1.1 million tonnes CO_2-equivivalents per year in 2000, largely because of increased recovery of MSW and the diversion of waste away from landfills.

Fig. 7: Emissions of GHG from studied waste management practices and scenarios (tonnes CO_2-equivalents)



When comparing the two studied scenarios we can see that the incineration scenario (scenario I) has a higher climate protection potential than the alternative composting scenario (scenario II). In scenario I where high rates of recycling and incineration with energy recovery are attained, the net emissions of CO_2-equivivalents are even negative. The reason for the negative net GHG emissions is a relatively low amount of waste sent to landfills as well as a high share of material recycling (avoided primary production of materials) and recovered energy in incineration plants (avoided emissions as a result of substituting heat and electricity produced from natural gas and oil shale in the background system). Incineration gives approx. 75% and recycling almost 25% of the total avoided emissions. In scenario II sources of GHG savings are mainly material recycling and the avoided emissions from landfilling. As in this scenario composting without energy recovery is applied, the net GHG emissions are higher than in scenario II.

Direct emissions from landfills continue to be a major source of GHG emissions till 2020 despite of the fact that the landfilling rate will degrease significantly and a relatively high share of landfill gas is recovered in both studied scenarios. The GHG emissions from waste collection and transport will increase by 2020 due to increased recycling. In scenario II a higher collection rate of biodegradable waste causes slightly more emissions of CO_2-equivivalents. In spite of that, the collection and transport of waste accounts for a relatively small amount of estimated net GHG emissions in both future scenarios.

7. Conclusions

As a result of the case study in Estonia it can be concluded that better management of municipal waste and especially diversion of municipal waste away from landfills could significantly reduce the emissions of GHG despite an almost 60% increase in waste generation by 2020. This is valid even if landfill gas is recovered at a high rate. Material recycling and incineration with high rates of energy recovery should be favoured compared to other waste management options where energy is not recovered (e.g. composting). It is important to stress that if high rates of recycling and incineration with energy recovery are attained, the net GHG emissions may even become negative, which means that these waste management options can partly offset the emissions that occurred when the products were manufactured from virgin materials and energy was produced from fossil fuel/oil shale. Intensive recycling and incineration of MSW usually also lead to a lower landfilling rate of MSW compared to other possible waste management scenarios. Overall, emissions of GHG associated with the collection and transport of the waste and recovered materials are small in comparison with other waste management practices. However, along the increased recycling rate (especially additional collection of biodegradables) the collection system and transport distances could have considerable impact on the overall GHG emissions of the waste management options.

In general, the conclusions of the study concur with other recent LCA studies (German Federal Environmental Agency 2005; ETC/RWM 2008; Sander 2008), but due to the fact that in Estonia energy produced from waste substitutes oil shale based electricity which has high climate change impact in terms of CO_2 emissions, incineration and other waste management options (e.g anaerobic digestion) where energy is produced, should be preferred to other waste management practices.

The aim of this study was to evaluate the climate change impact of the possible future waste management options in Estonia and not to predict the exact GHG emissions generated in the waste sector. The total emissions of GHG from MSW management depend on several factors. The waste composition was assumed to be the same throughout the studied period. In reality, it may be expected that certain waste fractions (e.g plastic packaging) will probably grow compared to other fractions. It would be good to study the possible change in the composition of waste and its possible impact on the results. Also the future energy source in Estonia is assumed to change including more renewable and nuclear energy while the share of oil shale is assumed to decline. Thus, it would be interesting to analyse how a possible change of the marginal energy source would affect the net GHG emissions, especially the relative effects of waste incineration. Waste management has a wide variety of impacts on the environment apart from those associated with climate change. Therefore, in the future also life cycle data and other parameters should be collected in order to evaluate such environmental impacts as acidification, eutrophication, toxicity, social impacts, etc.

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PAPER II

Voronova, V., Moora, H., Loigu, E., 2011. Environmental assessment and sustainable management options of leachate and landfill gas treatment in Estonian municipal waste landfills. Management of Environmental Quality: An International Journal, 22(6), 787 - 802.

Environmental assessment and sustainable management options of leachate and landfill gas treatment in Estonian municipal waste landfills

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Abstract

Purpose – The purpose of this paper is to compare various landfill gas (LFG) and leachate treatment technologies in a life-cycle perspective.

Design/methodology/approach – Since a landfill causes emissions for a very long-time period, life-cycle-based environmental assessment was carried out to compare different technological options for sustainable leachate treatment and LFG collection and utilization. WAMPS, the life-cycle assessment (LCA) model for waste management planning, was used for the environmental assessment of selected leachate and LFG treatment technologies.

Findings – Results of both direct measurements in the studied landfills and LCA support the fact that leachate treatment with reverse osmosis has the best environmental performance compared to aerobic-activated sludge treatment. Recently, the collection efficiency of LFG in the studied landfills is relatively low. In order to improve the overall environmental performance of LFG management the gas collection rate should be improved. LFG utilisation for energy recovery is an essential part of the system. The results of the study show that the avoided impacts of energy recovery can be even greater than direct impacts of greenhouse gas emissions from landfills. Therefore, measures which combine LFG collection with energy generation should be preferred to treatment in flare.

Research limitations/implications – It should be noted that the results of this study do not express the total environmental impacts of the entire landfill system, but only the eutrophicating impacts and global warming related to the studied leachate and LFG management options. Therefore, it is recommended that further LCAs investigate also other relevant impact categories.

Practical implications – The results of LCA modelling show that it is important to ensure the highest collection and treatment efficiency of leachate and LFG, since poor capture compromises the overall environmental performance of a landfill.

Originality/value – The paper provides a site-specific data on sustainable leachate and LFG management in selected Estonian conventional municipal solid waste landfills. As such, the paper contributes to the development of the regional reference input data for LCA in waste management.

Keywords Estonia, Waste management, Pollutant gases, Municipal solid waste, Sustainable landfilling, Leachate, Landfill gas, Life cycle assessment

Paper type Research paper

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1. Introduction

The new EU Member States including Estonia have recently experienced a rapid economic growth, resulting in a significant increase of waste quantities, while their waste management systems still require much effort to be adjusted to the European state of the art. Despite its low position in the waste hierarchy, landfilling has been the predominant method for municipal solid waste (MSW) management in Estonia. Approximately 76 percent of MSW was landfilled in 2007 (Moora, 2009).

Therefore, the implementation of EU waste legislation demands comprehensive research on solid waste landfill management issues to define a more sustainable landfilling approach (Philips *et al.*, 1999; Bovea and Powell, 2006; Seadon, 2010; Wagner, 2011) in Estonia.

The EU Landfill Directive 99/31/EC (EC, 1999), which was transposed into the Estonian legislation through the Waste Act (2004) and through the Regulation of the Minister of Environment No. 38, sets out provisions covering the location of landfills and establishes more stringent technical and engineering requirements for aspects such as water control and leachate management, protection of soil and water and landfill gas (LFG) emissions control. The landfill directive defines also progressive targets for the diversion of biodegradable fraction of MSW away from landfills.

In Estonia, all old landfills for depositing MSW were closed by 16 July 2009. After that date only five new landfills that comply with the technical requirements of the EU landfill directive remained operational.

Quantity, type and duration of the emissions from landfills depend largely on local factors such as the amount and composition of waste, the technical design of landfill and choice of effluent treatment technologies, and the location of landfill. Owing to the high share of biodegradable fraction of landfilled waste in Estonia (average 56 percent of total MSW) (Moora and Jürmann, 2008) and relatively humid weather, LFG and leachate are the main environmental impacts of landfilling.

The experience with new landfills in Estonia indicates that there are problems in assuring compliance with technical requirements (e.g. leachate treatment and gas collection systems). Especially, leachate treatment has necessitated additional efforts in terms of technology and related financial resources, since the initially designed treatment capacity has been insufficient for the load of leachate. Operators of new landfills in Estonia still lack knowledge about the key design parameters of emission treatment technologies such as leachate production rate and composition as well as LFG potential.

Wastes deposited to a landfill will cause emissions for a very long time. A landfill may give emissions for hundreds or even thousands of years. Therefore, it is important to take into account the life-cycle impact when assessing various technological options for leachate treatment as well as LFG collection and utilization.

The aim of this paper is to compare various LFG and leachate treatment technologies in a life-cycle perspective. WAMPS, the life-cycle assessment (LCA) model for waste management planning (Moora, 2009), was used for the environmental assessment of selected leachate and LFG treatment technologies.

The paper also presents site-specific data on sustainable leachate and LFG management in selected Estonian conventional MSW landfills. As such, the paper contributes to the development of the regional reference input data for LCA in waste management. The results of the research are based on a study carried out in three conventional landfills, undertaken in 2007-2009 by the Institute of Environmental Engineering at Tallinn University of Technology (Loigu, 2010).

2. MSW landfills in Estonia

Until the 1990s almost each larger settlement and company had a landfill which, in most cases, lacked any kind of environmental supervision. All old landfills for depositing MSW in Estonia were closed by 16 July 2009. Out of ca 350 landfills that did not meet the requirements of the EU landfill directive only five new landfills remained operational (Table I).

3. Description of the studied landfills and landfilled waste

As the aim of the paper was to analyse and compare different LFG and leachate treatment technologies that have been implemented in Estonian conventional landfills, three most representative landfills – Jõelähtme, Uikala and Väätsa – were chosen for the assessment.

Jõelähtme landfill is the biggest landfill located close to Tallinn City (capital area). More than half of the total MSW generated in Estonia is deposited in this landfill (Table II). Väätsa and Uikala landfills represent typical regional landfills. The studied landfills have implemented different measures for leachate and LFG management.

Since the waste quantity and composition play an important role in the leachate and LFG formation, the fractional composition of landfilled MSW was analysed

Name	Start of operation year	Description
Jõelähtme	2003	The biggest MSW landfill in Estonia. Located in the Northern part of Estonia (Harju County), 10 km from Tallinn. An area of appr. 9 ha has been covered by 800,000 m^3 of landfilled waste
Uikala	2002	Located in the Northeastern part of Estonia (Ida-Viru County). An area of appr. 2.2 ha has been covered by 750,000 m^3 of landfilled waste
Väätsa	2000	Located in the Central part of Estonia (Järva County), 13 km from Paide city. An area of appr. 1.2 ha area has been covered by landfilled waste
Paikre	2006	Located in the Southwestern part of Estonia (Pärnu County), 15 km from Pärnu city. An area of appr. 1.02 ha has been completed by covered waste
Torma	2001	Located in the Eastern part of Estonia (Jõgeva County). An area of appr. 1.76 ha has been covered by 122,000 m^3 of landfilled waste

Table I.
MSW landfills in Estonia

Landfill	Amount of MSW landfilled (t)									
	2000	2001	2002	2003	2004	2005	2006	2007	2008	Total
Väätsa	37	4,377	6,177	15,616	16,474	16,710	18,678	27,062	24,029	132,160
Uikala	–	–	23,375	37,362	41,267	52,901	42,740	48,106	36,272	282,023
Jõelähtme	–	–	–	80,466	177,132	179,581	185,675	206,146	182,314	1,011,314

Table II.
Amounts of MSW landfilled in the studied landfills (2000-2008)

in a separate sorting study (Moora and Jürmann, 2008). The focus was on the analysis of content of biodegradable waste fractions (organic waste, paper and cardboard, wood and textiles). Results of the sorting study show that the biodegradable fraction makes up a considerable share of MSW in all three studied landfills (Table III).

The amount of landfilled MSW showed an increase till 2007 (Table II). Since 2008, due to the economic decline, MSW depositing decreased in all landfills. In the coming years a higher waste recovery rate and MSW incineration (a waste incineration plant will start to operate in 2012 in the Tallinn area) will significantly influence the amount of deposited MSW. It may be assumed that the amount of MSW to be directed to landfills will drop from the current 76 percent to ca 5 percent by 2015.

4. Characterization of leachate and LFG formation

4.1 Landfill leachate

Landfill leachate formation is the result of the removal of soluble compounds by the non-uniform and intermittent percolation of water through the waste mass. Leachate contains organic and inorganic contaminants including humic acids, ammonia nitrogen, heavy metals and inorganic salts, and need to be treated before discharge to the environment due to the toxicity or unfavourable affect on the environment. The quantity of leachate generated is site specific and a function of water availability and weather conditions as well as the landfill geometry, surface and landfill engineering (Binner, 2003). Leachate composition is highly dependent upon the age (stage of fermentation) of the landfill and composition of landfilled waste.

Leachate generation was studied in the three studied landfills based on measurements obtained from field works in 2007 (Loigu, 2010). In parallel, the annual leachate generation was calculated based on Estonian long-term annual average precipitation and evaporation data (precipitations 650 mm and evaporation 450 mm) (Table IV).

Table III. Share of biodegradable waste fractions in landfilled MSW at the studied landfills 2007/2008

Category	Väätsa	Uikala	Jõelähtme
Kitchen waste	57	55	53
Garden waste	5.19	9	9
Textile	3.2	1.3	3.3
Paper and cardboard	32	31	31
Wood	1	1.2	1
Other organic waste	1.61	2.5	2.7
Share of biodegradable waste in landfilled MSW	52	60	55

Note: In percentage by mass
Source: Moora and Jürmann (2008)

Table IV. Measured and calculated leachate generation at the studied landfills

Landfill	Measured leachate amount (m^3/daily)	Calculated leachate amount (m^3/daily)
Jõelähtme	34.7	32.9
Uikala	25.9	16.4
Väätsa	13.2	12.8

The difference between the measured and calculated amounts of leachate generation at Uikala landfill can be explained by major rainfalls at Ida-Viru Region in 2007 in comparison with the Estonian average.

Normally, the amount of leachate is given as a percentage of rainfall. It can be assumed that the amount of leachate generated in Estonian conventional landfills is ca 30 percent of rainfall. This figure is in accordance with reports of other authors (Krümpelbeck, 2000; Plinke *et al.*, 2000; Wallmann, 1999) who showed the amounts of leachate in a range from 25 to 60 percent of rainfall.

Taking into account the basic parameters that characterise leachate all the studied landfills can be classified as stabilised old landfills (Table V).

As landfill age increases, organics concentration (COD) in leachate usually decreases and ammonia nitrogen concentration increases (Abbas *et al.*, 2009). The existing relationship between the age of the landfill and the organic matter composition should be taken into account when selecting a suitable leachate treatment technology.

A more detailed composition of leachate at the studied landfills is presented in Table VI. High concentrations of BOD, COD and nitrogen were found in leachates (exceeding the Estonian legal limit values). Concentration of phosphorus was within the limits or showed slight fluctuations exceeding the limits. Concentrations of heavy metals were relatively low. One of the reasons could be a very high content of sulphur in landfill and also high conductivity that makes it sufficient to precipitate metals from leachate water. Additionally, a toxicity test was carried out. Results showed that phenols in Uikala and Jõelähtme landfills are on an extremely high level. Such high-phenols content can be explained by the fact that in both landfills a significant amount of ash or residues from incineration have been used for covering the waste layers.

4.2 Landfill gas

The decomposition of organic waste under anaerobic conditions results in the formation of LFG which contains mainly methane (CH_4) and carbon dioxide (CO_2). The LFG contains also numerous other constituents, e.g. non-CH_4 volatile organic compounds and a small amount of nitrous oxide (N_2O), nitrogen oxide (NO_x) and carbon monoxide. LFGs, and especially CH_4, contribute significantly to global warming effect (IPCC, 2006).

The predicted amount of gas emissions from landfills can be estimated by the content of organic matter in waste. Different calculation methodologies are used on LFG estimations (Scharff and Jacobs, 2006). The gas potential related to the disposed waste (fresh matter, FM) is of importance in calculations.

	Young	Medium	Old	Jõelähtme	Väätsa	Uikala
Age (year)	<1	1-5	>5	6	9	7
pH	<6.5	6.5-7.5	>7.5	7.1-8.1	7.6-7.8	7.5-8.2
BOD/COD	0.5-1.0	0.1-0.5	<0.1	0.2-0.3	0.2-0.3	0.3
COD (g/l)	>15	3-15	<3	<9.1	<2.3	<4.8
NH_3—N (mg/l)	<400	NA	>400	<974[a]	<332[a]	<864[a]
TOC/COD	<0.3	0.3-0.5	>0.5	0.2-0.7	0.3-0.5	0.2-0.6
N_{tot} (g/l)	0.1-2	NA	NA	0.2-1.3	0.1-0.4	0.7-1.1

Note: [a]Calculated as a sum of $NH_3 + NH_4$ per nitrogen
Source: Hector *et al.* (2004)

Table V. Landfill leachate classification based on age of landfill and data from studied landfills

Name	Unit	Väätsa	Jõelähtme	Uikala	Legal limit value[a]
pH		7.6-7.8	7.1-8.1	7.5-8.2	6.0-9.0
El. conductivity	mS/cm	5,500-9,600	4,000-16,000	11,200-16,800	–
Floating substance	mg/l	280-690	280-870	270-520	15
BOD_7	mgO_2/l	300-1,663	100-5,500	500-1,800	15
COD	mgO/l	580-7,900	600-9,000	2,800-4,800	125
TOC	mgC/l	140-680	150-2,500	800-1,300	–
NH_4	mgN/l	50-330	100-980	440-680	–
N_{tot}	mgN/l	130-470	150-1,350	720-920	75
P_{tot}	mgP/l	2-7	3-10	3-6	2
HCO_3	mg/l	2,000-4,700	1,300-7,900	4,700-8,200	–
SO_4	mg/l	500-700	150-480	150-650	–
Cl	mg/l	200-550	300-1,760	1,500-1,730	–
1-phenols	mg/l	0.0025-0.125	0.004-67	1.25-3.9	0.1
2-phenols	mg/l	< 0,01-0.48	< 0.01-5.8	0.175-3.6	15
Oil prod.	mg/l	< 0.02-0.057	< 0.02-4.5	< 0.02	1
Fe(II) val.	mg/l	1.6-2.6	1.6-6.6	1.1-1.2	–
Fe(III) val.	mg/l	1.5-12	4.5-26	8.0-21	–
Hg	mg/l	< 0.05-0.25	< 0.05-0.13	< 0.05	50
Ag	mg/l	< 0.01-0.01	< 0.01	< 0.01	0.2
Cd	mg/l	< 0.02	< 0.02	< 0.02	0.2
Cr	mg/l	0.03-0.3	0.05-0.5	0.2-0.3	0.5
Mg	mg/l	98-117	60-240	220-360	–
Mn	mg/l	0.25-0.3	0.5-1.0	0.1-0.6	–
Na	mg/l	450-800	300-1,300	1,200-1,600	–
Ni	mg/l	0.07-0.1	0.04-0.2	0.05-0.08	1.0
Pb	mg/l	< 0.04	< 0.04	< 0.04	0.5
Zn	mg/l	< 0.1	< 0.1	< 0.1	2.0
Cu	mg/l	0.03-0.1	< 0.02-0.2	< 0.02	2.0

Note: [a]Estonian legal limit values are in line with EU values based on Directive 91/271/EEC

Table VI.
Composition of leachate at the studied landfills

For estimating the LFG generation in the studied landfills the following calculation model was used (Tabasaran and Rettenberger, 1987). Mathematically, it can be described as an empirical equation, where the share of CH_4 is 55 percent, CO_2 35 percent and air 10 percent:

$$G = 1.868 \times C_{org} (0.014d + 0.28) (1 - 10^{-kt}), \quad (1)$$

where:

G	specific LFG production until time t (m^3/t).
C_{org}	total organic carbon (kg/t).
d	temperature inside the landfill (°C).
k − 0.035	degradation coefficient.
t	the landfill operation time.

Table VII presents the calculated LFG potential of the studied landfills. LFG potential varies from 220 to 280 m^3/t of FM. It is in a range with the reports of other authors (Tabasaran and Rettenberger, 1987; Plinke *et al.*, 2000) estimations of LFG generation rates (120-300 m^3/t FM). However, as such the LFG generation in Estonian landfills is twice as much as that in average modern conventional landfills in Western-European countries (Obersteiner *et al.*, 2007). Such high gas potential can be explained by high content of organic matter in deposited MSW.

Additionally, the distribution of CH_4 generation in time was calculated for the studied landfills (Figure 1). The calculation takes into account the amount of MSW that is deposited during the most active period of landfilling from the opening of the landfill till 2015. It can be assumed that the amount of MSW directed to all Estonian landfills will drop significantly by 2015 (Section 3). Based on these assumptions it can be expected that the peak of the LFG generation in terms of CH_4 will be achieved around 2012.

5. Leachate and LFG treatment

5.1 Leachate treatment technologies

Landfill leachate needs to be pre-treated on site to meet the limit values for its discharge into the sewer or direct disposal into surface water. Various leachate treatment processes have been used in different landfills in Estonia.

There are three most common approaches for leachate treatment:

(1) *Combined treatment with municipal wastewater*. Collected leachate is piped out into the municipal sewer system to be treated with domestic sewage. This treatment option is used in Jõelähtme landfill. It is preferred for its easy maintenance and low operating costs. However, this option has been increasingly questioned due to the presence in the leachate of organic inhibitory compounds of low biodegradability and heavy metals that may reduce treatment efficiency and increase the effluent concentrations (Cecen and Aktas, 2004).

Landfill	Gas potential (m^3/t FM)
Jõelähtme	221
Uikala	255
Väätsa	281

Table VII. Calculated LFG potential at the studied landfills

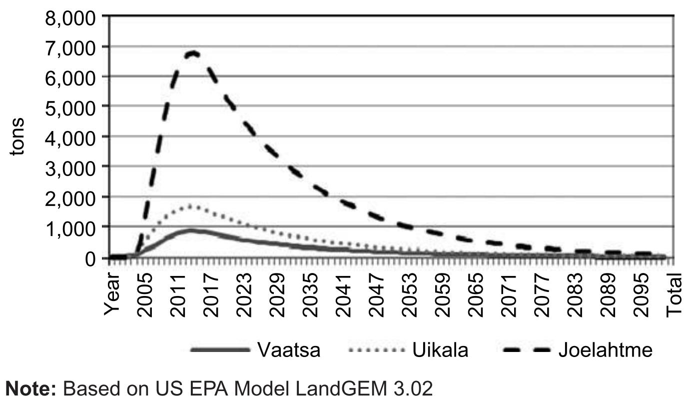

Figure 1. Methane generation at the studied landfills through long-term distribution

(2) *Aerobic treatment based on activated sludge processes*. This treatment method allows partial abatement of biodegradable organic pollutants and also achieves ammonium nitrogen nitrification. This treatment option is used in Väätsa landfill. It is seen as a low-cost method for removing pathogens, organic and inorganic matter. However, aerobic treatment based on activated sludge process has proved to be inadequate for handling landfill leachate treatment. This method is ineffective due to high concentrations of contaminants whose removal is ineffective in the case of one stage cleaning by activated sludge. This is very well illustrated by the results of tests made at Väätsa landfill (Table VIII).

(3) *Membrane processes based on reverse osmosis (RO)*. RO is a high-pressure membrane separation process that is one of the most efficient methods for landfill leachate treatment. This method is applied in Uikala landfill. Based on test results made in Uikala, it can be concluded that RO method allows to achieve very high removal rates of different pollutants (COD removal rate in Uikala varies between 90 and 96 percent) (Table VIII).

The results of a comparative analysis of efficiencies of aerobic treatment and RO membrane technology in Väätsa and Uikala landfills are presented in Table VIII. The efficiency of heavy metal removal is not taken into account due to their very low

Leachate treatment option	Sample	BOD_7 (mgO_2/l)	COD (mgO_2/l)	P_{total} (mgP/l)	N_{total} (mgN/l)
Aerobic-activated sludge (Väätsa landfill)					
Before treatment	1 March 2007	250	1,800	5	603
	1 May 2007	150	1,600	8.3	420
	23 September 2007	1,663	8,730	4.5	200
	18 October 2007	366	988	4.66	394
After treatment	1 March 2007	75	1,960	4.9	424.7
	1 May 2007	45	680	3.8	200
	23 September 2007	525	1,009	3.9	395
	18 October 2007	209	956	5.13	246
Effectiveness of leachate treatment (%)		63	37	17	30
RO (Uikala landfill)					
Before treatment	15 January 2007	220	960	2.7	430
	4 April 2007	171	130	0.15	45
	8 October 2007	1,851	4,840	6.06	720
	22 October 2007	541	2,870	2.89	920
After treatment	15 January 2007	6.5	60	0.21	23
	4 April 2007	11	40	0.04	0.71
	8 October 2007	73	80.5	0.187	79
	22 October 2007	13	30.6	0.029	16
Effectiveness of leachate treatment[a] % (aerobic activates ludge)		63	37	17	30
Effectiveness of leachate treatment[a] (%) (reverse osmosis)		96	90	90	95

Note: [a]Effectiveness of leachate treatment accounted relating to the Estonian normative for landfill water

Table VIII. Comparison of efficiency of leachate treatment technologies concentrations in leachate water. It is difficult to estimate the efficiency of leachate treatment in Jõelähtme landfill because leachate is treated together with municipal wastewater.

5.2 LFG management technologies

According to the landfill directive LFG shall be collected from all landfills receiving biodegradable waste and the LFG must be treated and used. If the LFG collected cannot be used to produce energy, it must be flared.

All operational Estonian MSW landfills have installed gas collection and flaring systems. LFG is extracted from landfills through extraction wells. Two most common types of LFG collection systems are used – vertical wells (Jõelähtme) and horizontal pipe system (Väätsa, Uikala). As experience with Estonian landfills shows that vertical wells are less expensive and suitable for installation in active filling areas. Horizontal collector wells could be used to improve the efficiency of LFG extraction systems from low yielding, shallow and laterally extensive landfill sites.

LFG is extracted and piped to a main collection header, where it is sent to be flared in open high-temperature flares. There is scarce information about gas collection efficiency in the studied landfills. According to the measurements made in Väätsa landfill, collected and flared LFG amount was about 350,000 m^3 in 2008, which makes only 8.5 percent of the total estimated LFG generation. It may be expected that gas collection will improve in the future. According to the literature, the LFG collection rate with efficient LFG collection system can vary between 50 and 75 percent of the total LFG generation (Sundqvist, 1999; Niskanen *et al.*, 2009; Del Borghi *et al.*, 2009).

All three studied landfills have started to develop LFG utilisation projects for producing electricity from LFG by using gas engines (planned capacity in Jõelähtme landfill 1.9 MW_{el}, Uikala landfill 0.3 MW_{el}, Väätsa landfill 0.35 MW_{el}).

6. Environmental assessment of leachate and LFG management options

6.1 Methodology

To evaluate and compare the environmental performance of installed or future leachate and LFG treatment technologies a special landfill version of LCA model for waste management planning WAMPS (Moora, 2009) was applied. This model allows scenario analysis of different waste management technologies and systems. It enables to learn how changes in the system affect its environmental and economic impacts. This model has been adjusted to the Estonian conditions.

This LCA model was developed by the Swedish Environmental Research Institute and is based on a more in-depth LCA model ORWARE (Björklund, 2000; Eriksson *et al.*, 2000; Sundqvist *et al.*, 2002).

WAMPS landfill model calculates emissions during a surveyable time period of about 100 years until the most active processes (especially, LFG generation) in landfill have ended (Sundqvist, 1999).

The main processes in the studied landfill system (LCA boundaries) are waste disposal at landfill site, on-site operations, gas collection, gas treatment by flaring, gas utilisation for energy production, leachate collection and treatment.

The LCA assessment takes into account the two most relevant impact categories for MSW landfilling: global warming and eutrophication of water. For each impact category emissions are weighted together with characterization factors (Sundqvist *et al.*, 2002; Hejungs *et al.*, 1992):

(1) *Global warming*. All emissions are expressed as CO_2 equivalents: 1 kg of CH_4 is equal to 25 kg of fossil CO_2 and 1 kg of N_2O is equal to 310 kg of fossil CO_2.
(2) *Eutrophication of water*. All emissions are expressed as PO_4^{3-} equivalents: 1 kg of phosphorus (P) is equal 3.06 kg of phosphate $\left(PO_4^{3-}\right)$, 1 kg of nitrogen (N) is equal to 0.42 kg of $\left(PO_4^{3-}\right)$, 1 kg of NH_3/NH_4 is equal to 0.34 of $\left(PO_4^{3-}\right)$, 1 kg of COD is equal to 0.022 kg of $\left(PO_4^{3-}\right)$.

Toxicity-related impacts were neglected in this study because there are still many uncertainties in modelling toxicological impacts (Reap *et al.*, 2008).

The net emissions from the studied system are calculated according to:

$$E_{net} = E_{waste} - E_{Backround} \tag{2}$$

E_{net} net emission (t/year or kg/year).

E_{waste} emission from a waste process that produces a certain amount of product (t/year or kg/year).

$E_{Background}$ emission from the same amount of alternative virgin production in the background system (t/year or kg/year).

This calculation can give negative emissions. This means for example that energy produced from LFG substitutes energy production in the background system.

The landfill system studied reflects the main parameters of Väätsa landfill. The total long-term leachate generation and LFG emissions (on the basis of a time horizon of 100 years) were calculated based on MSW amount and composition deposited at Väätsa landfill in 2000-2015. Taking into account the future developments of MSW management in Estonia it can be assumed that after 2015 only a very small amount of MSW will be landfilled (Section 3).

The local site-specific data on leachate and LFG generation and management that were collected during the research in Estonian conventional landfills was used to develop the core features of alternative treatment options.

It was assumed that partial oxidation of the uncollected LFG in the final soil cover not more than 15 percent (Sundqvist, 1999; Niskanen *et al.*, 2009; Manfredi *et al.*, 2009).

Emissions of CH_4 and CO_2 were calculated according to the carbon balance of a conventional MSW landfill (Sundqvist, 1999):

$$CH_4 = (1-\gamma)*(1-\varepsilon)*\alpha*\beta*0.99*16/12 \text{ kg } CH_4 \text{ kg } C_{in} \tag{3}$$

$$CO_2 = (1-\varepsilon)*(1-\beta)*\alpha*0.99*44/12 + \gamma*(1-\varepsilon)*\alpha*\beta*0.99*44/12 \text{ kg } CO_2 \text{ kg } C_{in} \tag{4}$$

α degradation yield kg °C kg C_{in}.

β molar (or volume) ration kmol CH_4 kmol (CO_2—CH_4).

γ oxidation yield of CH_4 in soil cover kg oxidised CH_4 kg CH_4 transported through soil.

ε part formed CH_4 that is recovered kg recovered CH_4 kg formed CH_4.

6.2 Comparison of treatment options

6.2.1 Leachate treatment. The environmental performance of three leachate treatment options were assessed and compared (Table IX). The worse case or reference situation (option 1) presents the typical landfilling situation before year 2000 where no treatment of landfill leachate was applied. Option 2 presents the situation where one-stage aerobic treatment based on activated sludge process is used. In option 3, leachate is treated by membrane technology based on RO.

6.2.2 LFG treatment. The environmental performance of four options for LFG management was assessed and compared (Table X). Reference option characterises the earlier situation in Estonian MSW landfills without any collection and utilization of LFG, where emissions of LFG were primary directed to the atmosphere. The current LFG management (option 1) in Estonian landfills is based on a rather moderate LFG collection (8.5 percent according to the data from Väätsa landfill) with further flaring. The current gas management situation is compared with three potential future options. An assumption was made that the gas collection rate will reach its maximum (75 percent). In option 2, flaring has been continued as the only treatment option. In option 3, the collected gas will be utilised for combined heat and power generation (electricity production efficiency is 35 percent and that of heat 60 percent). In option 3, only electricity is produced from the collected gas (electricity production efficiency is 39 percent).

6.3 Results

The results of life-cycle-based environmental impact assessment are shown in Figures 2 and 3. It should be kept in mind that the results are based on the chosen sample landfill (Väätsa landfill). Moreover, the results do not express the total environmental impact of the entire landfill system, but only the potential impacts related to the studied leachate and LFG management options.

Leachate treatment option	Treatment efficiency – removal rate of main pollutants (%)			
	BOD_7	COD	P_{total}	N_{total}
Option 1 – no treatment of leachate	0	0	0	0
Option 2 – aerobic treatment with activated sludge	63	37	17	30
Option 3 – RO	96	90	90	95

Table IX. Compared leachate treatment options

LFG management	Reference conditions	Option 1 (flaring)	Option 2 (flaring)	Option 3 (heat/electricity generation)	Option 4 (electricity generation)
Collection (% of generated)	0	85	75	75	75
Flared (% of collected)	0	100	100	0	0
Utilized (% of collected)	0	0	0	Electricity 35/ heat 60	Only electricity production 39

Table X. Comparison of LFG management options

6.3.1 Environmental impact of leachate tratment. Figure 2 shows the results of LCA with a focus on the main leachate parameters (pollutants) emitted to the environment. Untreated leachate (option 1) can be taken as a reference for evaluating the efficiency (environmental impact) of the studied leachate treatment options. Aerobic treatment with activated sludge reactor (option 2) minimises the emissions of nitrogen, ammonium and phosphorus, however, the total long-term environmental impact in terms of eutrophication is only less than half smaller than in the case of no treatment.

The results of LCA support the fact that leachate treatment with RO has the best environmental performance compared to other leachate tratment technologies. The long-term environmental impact when using RO is very small in case of nitrogen, ammonium and other pollutants.

6.3.2 Environmental impact of LFG management. Figure 3 shows emissions of main greenhouse gases (GHGs) in LFG expressed in kg of CO_2-ekv. As the results show, the current LFG management system with a relatively low gas collection rate (option 1) does not lead to a significant reduction of GHG emissions from landfill. If we assume that gas collection will be improved and the collection rate will reach the maximum of the total potential generation (75 percent), all the compared gas management options (flaring – option 2, electricity and heat generation – option 3, electricity production – option 4) lead to a substantial reduction of GHG emissions. However, the flaring option has a lower environmental performance, as no LFG is here utilised for energy generation. LFG utilisation for energy recovery leads to saved emissions and avoided global warming impact potential. The produced heat and electricity substitute the heat

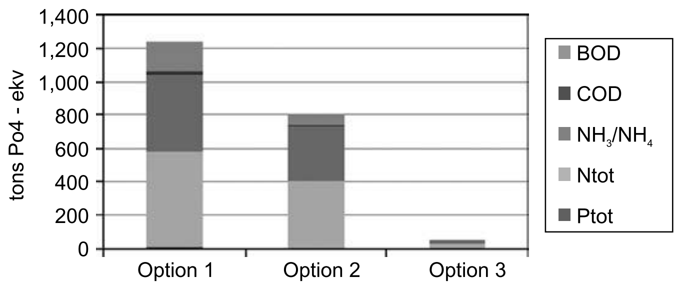

Figure 2.
Emissions of selected eutrophicating substances of studied leachate treatment options

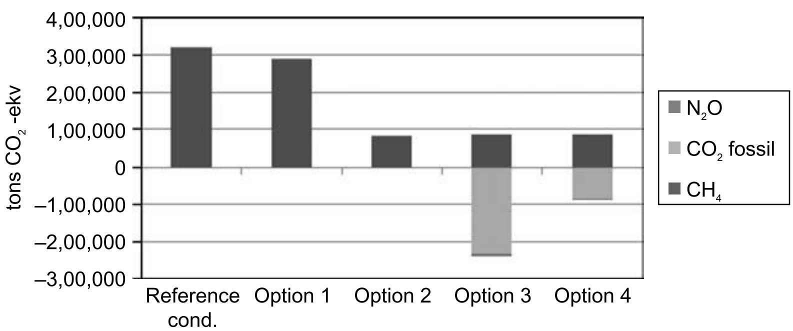

Figure 3.
Emissions of GHGs of the studied LFG management options

and electricity produced from fossil fuels – natural gas and oil shale – in the background system and thus result in the negative impact (avoided impact). The net benefit is proportional to the energy recovery efficiency achieved. As the results show the best result is achieved when LFG is used for heat and electricity production (option 3). Then the avoided impact from energy recovery is greater than direct impacts of GHG emissions from landfill.

7. Discussion and conclusions

The environmental impact of the emissions from landfills depend largely on local factors such as the location of landfill, the amount and composition of waste landfilled, the technical design of landfill and the choice of technologies for LFG collection and treatment of leachate. Owing to the very high share of biodegradable fraction in the landfilled waste and relatively humid weather, the Estonian conventional MSW landfills generate a significant amount of leachate and LFG rich in CH_4.

Taking into account the basic parameters that characterise leachate all the studied landfills can be classified as stabilised old landfills. Very high concentrations of BOD, COD and nitrogen were found in the leachate of the studied landfills. In addition to age (stage of fermentation) and waste composition, the co-disposal of certain industrial waste could influence the composition of leachate. The extremely high level of phenols can be explained by the fact that a significant amount of ash or residues from incineration have been used in landfills to cover the waste layers.

Results of both direct measurements in the studied landfills and LCA modelling support the fact that leachate treatment with RO has the best environmental performance compared to other leachate treatment technologies. This fact is supported by several studies of other authors (Renou *et al.*, 2008; Li *et al.*, 2009). However, when designing the treatment technology it should be taken into account that leachate should be collected and treated properly for a relatively long-time period (up to 40 years). Here also the economic aspects of different treatment technologies should be considered.

LFG potential in Estonian MSW landfills varies from 220 to 280 m^3/t FM. As such it is twice higher than in average modern conventional landfills in Western-European countries. It can be assumed that the amount of MSW directed to Estonian landfills will drop significantly by 2015 due to increasing recovery and opening of a waste incineration plant. Based on these developments it can be expected that the peak of LFG generation in terms of CH_4 will be achieved in all operational MSW landfills at around 2012. Therefore, in the coming years much focus has to be put to improving the LFG collection and treatments systems.

As the available data show the collection efficiency of LFG in the studied landfills is currently relatively low. In order to improve the overall environmental performance of the current LFG management (based on flaring), the LFG collection rate should be improved.

The results of LCA show that when the gas collection rate is increased to the maximum (75 percent), all the compared gas management options (flaring, electricity and heat generation and electricity production) lead to a substantial reduction of GHG emissions. LFG utilisation for energy recovery is an essential part of the treatment system since it leads to saved emissions and avoided global warming impact potential. The results of the study show that avoided impact from energy recovery can be even greater than direct impacts of GHG emissions from landfills. Therefore, the measures which combine LFG collection with energy generation should be preferred to treatment in flare. The environmental as well as economic benefits multiply when utilising LFG for both heat and electricity production. However, the possibility to utilise produced heat is rather limited, since landfills are located far from potential users of heat. Results of the study are inline with similar investigations of other authors (Bovea and Powell, 2006; Niskanen *et al.*, 2009; Manfredi *et al.*, 2009; Moora *et al.*, 2009). It should be noted that the results of this study do not express the total environmental impacts of the entire landfill system, but only the eutrophicating impacts and global warming related to the studied leachate and LFG management options. Therefore, it is recommended that further LCAs investigate also other relevant impact categories. It is especially relevant for toxicity-related impacts, since it was found that the content of toxic substances in leachate was relatively high; however, this can change significantly over a longer period.

It can be expected that due to the increasing recovery and waste incineration, the amount and composition of waste will change dramatically in the near future. Thus, it is necessary to study how these changes will influence the potential environmental impact and sustainable management options of Estonian landfills.

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PAPER III

Moora, H.; **Voronova, V**.; Uselyte, R., 2012. Incineration of Municipal Solid Waste in the Baltic States: Influencing Factors and Perspectives. Avraam Karagiannidis (Toim.). Waste to Energy: Opportunities and Challenges for Developing and Tsansition Economies (237 - 260). London: Springer-Verlag London Ltd

Incineration of Municipal Solid Waste in the Baltic States: Influencing Factors and Perspectives

Harri Moora, Viktoria Voronova and Rasa Uselyte

Abstract The three Baltic States are in the stage of changing their municipal waste management systems since they have to comply with the principles and targets of the European Union waste policy and directives. Over the past years, thermal treatment of municipal waste has been discussed more intensely in these countries as one of the waste management option that could help to reach the legal targets in a relatively short time. In general, the Baltic States have similar socio-economic characteristics, waste and energy sector developments and geographical conditions that form similar frameworks for the development of a waste management infrastructure, including possible waste-to-energy options. However, as experience from recent studies and projects shows, there are several local and regional factors that could significantly influence the economic success of large scale waste incineration. The paper attempts to identify and discuss these main influencing factors and perspectives for MSW incineration in the Baltic States. The main focus is on conventional mass-burn incineration. The specific issues in terms of technical, economic and environmental aspects are presented in the form of an illustrative case study based on the design and performance data of the first waste-to-energy facility in Estonia.

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1 Introduction

The three Baltic States—Estonia, Latvia and Lithuania—have recently experienced rapid economic growth, resulting in a significant increase of municipal solid waste (MSW) quantities, while their waste management systems still require much effort to be adjusted to the European state-of-the-art. A wide variety of technological options, increasingly diverse waste fractions, environmental restrictions and European Union (EU)-wide recovery targets require the decision makers to well consider the steps to be made. The solutions to municipal waste management should not only be environmentally sustainable but also cost-efficient and socially accepted. Therefore, waste management has become one of the key issues in governments, in the waste management sector as well as among the general public in all new EU Member States including the three Baltic States.

In spite of its lowest priority in the European waste management hierarchy, landfilling has been the predominant method for municipal waste management in all the Baltic States. The fact that landfilling is the worst option for MSW treatment is generally accepted. However, the choice of the most optimal waste management solution has been under heavy discussion.

Over the past years, thermal treatment of MSW has been discussed more intensely in the Baltic States. There exist several plans to build waste to energy (WtE) facilities in the Baltic States. At least one mass burn incineration is under construction in Estonia and the construction of the first incineration facility in Lithuania will start soon. Several other projects are in the preparatory phase.

Even though the 27 EU Member States are directly governed by the same overall legislation, including that on waste management, disposal or incineration, the importance of incineration differs widely from one EU member state to another [24, 25]. This is because the issue of waste incineration is complex and the success of this MSW treatment option depends largely on the framework conditions characteristic for a specific country or region [23, 26].

Energy recovery by waste incineration has a double function as a waste treatment method and a supplier of electricity and/or heat, thereby linking the systems of energy and waste management. Both systems are undergoing great changes in the Baltic States. There are also several other influencing factors (e.g. waste generation and content, waste and energy sector/market developments, environmental impacts and public opinion) that have to be studied carefully before starting to develop any plans for MSW incineration [26]. The Baltic States waste sector/market is relatively small in size. Therefore the already existing plans to build large-scale WtE facilities have resulted in many discussions and debates among other waste management actors. As the experiences from other European countries have shown, waste incineration could have a significant impact on the existing waste management system [1, 13, 14, 22–24].

This paper attempts to identify and discuss the main influencing factors and perspectives for MSW incineration in the Baltic States. The main focus is on conventional mass-burn incineration as it is probably the most suitable large-scale

WtE technology for the management of mixed municipal waste under the conditions in the Baltic States.

The specific issues in terms of technical, economic and environmental aspects related to large scale mass-burn incineration of MSW are presented in the form of an illustrative case study based on the design and performance data of the first WtE facility in Estonia. The environmental impacts of the new WtE facility were assessed by using the life cycle assessment (LCA) software tool WAMPS [17].

The information contained in this paper was derived from a series of earlier research projects, the aim of which was to analyse the environmental impacts and economic costs of planned WtE projects as well as evaluating possible alternative scenarios for MSW management in the Baltic States [1, 16, 17, 28, 31, 32].

The discussion on influencing factors of MSW incineration in this paper focuses mainly on the Estonian context. This is because of availability of data and a more established waste management system. However, most of the examples and discussions are also relevant for the other Baltic States which have a similar socio-economic structure and waste management and energy sector development as Estonia.

2 Plans for MSW Incineration

There is no experience in large scale MSW incineration in the Baltic States. However, for some years now, cement factories in the Baltic States have used refuse derived fuel (RDF) imported from other EU Member States and on a smaller scale produced in the first local mechanical–biological treatment (MBT) facilities.

All three Baltic countries have developed national waste management plans that foresee a place for possible municipal waste incineration. However, concrete projects to build waste incinerators are in very different stages of development in these countries (Table 1).

The construction of the first mass-burn incineration facility in the Baltic States started in 2010 and the new waste incineration unit of the Iru power plant close to the Estonian capital Tallinn is expected to begin generating electricity and heat from MSW in 2013. This WtE facility with annual capacity of 220,000 tons is supposed to incinerate MSW from all Estonia. Furthermore, plans for a second waste incineration plant in Tartu (central part of Estonia) with a 100,000 t/a capacity are under discussion. The city of Tartu initiated discussions about this plant because the regional landfill was closed in 2009. However, the project has been postponed due to uncertainties related to financing and the waste market.

There have been discussions for many years to build a waste incinerator in Riga, the capital of Latvia. However, currently there are no concrete plans for municipal waste incineration projects in Latvia.

There exist two projects to build mass-burn waste incineration plants in Lithuania. The Klaipeda region municipal waste management plan for 2010–2019 foresees that the mixed municipal waste collected from the Klaipeda waste

Table 1 Plans for WtE facilities in the Baltic States

Country	Number of projects/plants	Capacity (t/a)	Status
Estonia	Iru WtE unit	220,000	Under construction
	Tartu WtE plant	100,000	Planned
Latvia	–	–	–
Lithuania	Klaipeda WtE plant	245,000	Construction will start in 2011
	Vilnius WtE plant	250,000	Planned

management region (7 municipalities) will be incinerated. The construction of Klaipeda WtE plant at the Lypkiai local boiler house area will start in 2011 and it is expected to be completed by 2013. The total capacity of this plant is 245,000 t/a including up to 130,000 tonnes of MSW, 75,000 of biofuel and 50,000 of industrial waste. Klaipeda WtE plant is a co-operation project between the Finnish energy company Fortum and the local energy company Klaipėdos energija, mainly controlled by the Klaipeda city municipality.

A second WtE plant is planned to be constructed in Vilnius, the capital of Lithuania. A private company Regioninė komunalinių atliekų deginimo gamykla (Regional Municipal Waste Incineration Plant) had an intention to build a WtE plant in Vilnius (next to the current combined heat and power plant CHP-3 in Vilnius). The local people strictly opposed to this project. As a result of the Environmental Impact Assessment (EIA) process, the Vilnius Region Environmental Protection Department did not allow to build the plant, because the involvement of Vilnius municipality, especially in relation to the engagement of the public, was considered not sufficient. After this experience, the municipalities of Vilnius waste management region (8 municipalities) have decided to start a new a tendering process for the construction and operation of a waste incineration plant.

3 Factors Influencing MSW Incineration

Experiences in other EU Member States show that the role of waste incineration differs widely from one EU member state to another. This is because waste incineration is a highly complex waste treatment option, which involves large investments and that depends largely on the framework conditions characteristic for a specific country or region. In the EU both waste management and energy production are subject to extensive regulations. The legislation aims to set a suitable policy framework at the EU level, with specific targets, while leaving the choice of pathways and technological development to the individual players in the Member States (authorities and private sector).

In addition to the legal framework that is based on the same general requirements the three Baltic States have similar socio-economic characteristics, waste and energy sector developments and geographical conditions that form similar framework for the development of waste management infrastructure including possible WtE options. However, as the experiences from feasibility studies and

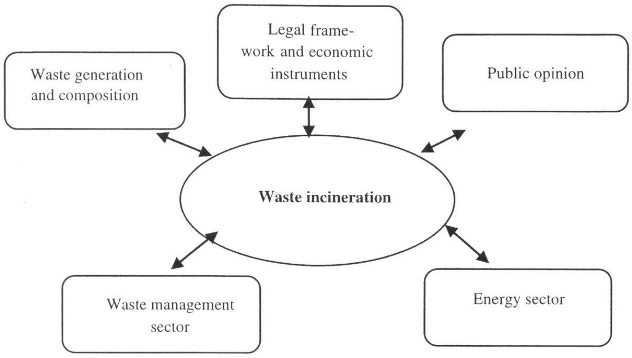

Fig. 1 Main influencing factors for municipal waste incineration

first WtE projects in these countries indicate, there are several local and regional factors that could significantly influence the success of waste incineration. Also the possible future trends and developments of these influencing factors have to be carefully studied. Given a service life of over 30 years, a waste incineration technology must also be able to function efficiently under the changing conditions in the future. The main framework factors that could influence the economic success of the new WtE facilities in the Baltic States are presented in Fig. 1.

3.1 Legal Framework and Economic Instruments

National waste policy and legislation in the Baltic States, as in all other EU-27 Member States, is governed by the EU policy and legislation.

The EU legislation on waste management is based on the Waste Framework Directive 2008/98/EC [8], which among others provides a definition of waste and sets out a general ranking of waste management methods, the so-called waste hierarchy. According to waste hierarchy waste generation should be prevented or reduced, and what is generated should be recovered by means of reuse, recycling and other recovery operations, thus reducing disposal/landfilling. A strong driver for improving the energy performance of waste-to-energy facilities is the Waste Framework Directive's new provision that allows high efficiency installations to benefit from a status of "recovery" rather than "disposal".

Recognising that not all waste can be prevented or recycled, the EU has also adopted directives on waste incineration and landfilling: Waste Incineration Directive 2000/76/EC [7] and Landfill Directive 1999/31/EC [6].

The Landfill Directive is arguably one of the most influential documents of the portfolio of the EU waste management regulations with direct influence on the development of waste recovery (including WtE) options [2, 10, 21, 29]. It sets

Table 2 Landfill taxes and bans (2010)

	Average gate fee for landfilling euro/tonne	Landfill tax in euro/ tonne	Ban of landfill of unsorted MSW
Estonia	45	12	Yes
Latvia	24.2	4.27	–
Lithuania	17.5	Planned in 2013, 22 euro/tonne	–

progressive targets for the reduction of the biodegradable fraction of MSW going to landfills to 75% of their 1995 baseline levels by 2006, 50% by 2009 and 35% by 2016. The Baltic States as other new Member States that rely heavily on landfilling, have made use of the allowance to postpone these targets by 4 years. Therefore they need to meet the respective diversion targets by 2010, 2013 and 2020.

The diversion of the biodegradable fraction of MSW places major challenges on all new EU Member States. Taking into account the current situation in the MSW management, it can be expected that the biodegradable waste diversion targets (especially the targets for 2013 and 2020) will be very challenging for the Baltic States. Consequently, there is an urgent need for action. Municipal waste incineration is one of the most realistic options that could help to reach these targets.

Many EU Member States with high waste recovery rates have facilitated the EU waste policy implementation by taxes on waste landfilling and landfill bans. Estonia introduced a pollution charge for municipal waste disposal (landfill tax) already in 1990. Until 2005, the rate was very low at EUR 0.10–0.20 per tonne. In 2006 it rose to EUR 7.8 per tonne and increases every year, reaching EUR 29.84 in 2015. Due to the landfill tax the landfilling fee has increased considerably over recent years. Estonia has also introduced a ban on the landfilling of untreated waste (including mixed municipal waste) (see Table 2).

Latvia has also introduced a landfill tax, applied as a natural resource tax. Lithuania intends to introduce a landfill tax in 2013. A ban on the landfilling of untreated waste has not yet been implemented in Latvia and Lithuania, due to lack of an alternative waste treatment capacity.

Based on the experience of Estonia, the legal requirements together with economic instruments such us landfill tax have resulted in favourable conditions for the development of new recovery facilities including waste incineration.

3.2 Municipal Waste Generation and Composition

The economic success of waste incineration depends directly on the available waste amount and waste composition. Investments in waste incineration presume a steady fixed stream of waste to ensure financial viability. The waste supply should be fairly stable in the whole life span of a WtE facility (up to 30 years).

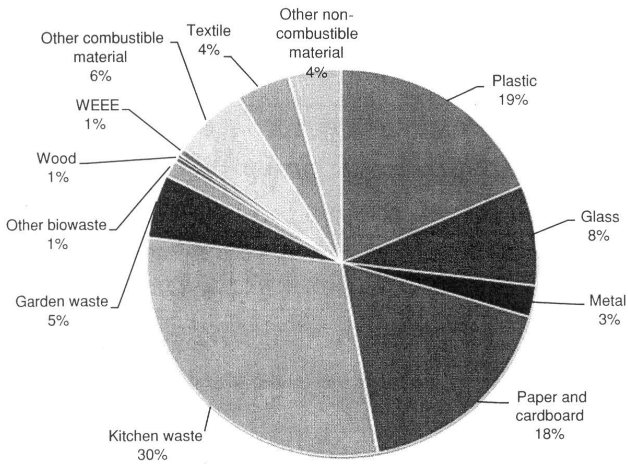

Fig. 2 Mixed municipal waste (landfilled) composition in Estonia (2008)

The energy content of waste, the so-called calorific value, depends on the composition of the waste and preferred to be as high as possible.

However, waste composition may change in time because of either additional recycling or changes in the socio-economic situation in the collection area. Both changes can significantly alter the amount of waste and its calorific value. Therefore, data on waste generation and composition as well as forecasting these waste trends are essential for the planning and development of a waste incineration project.

The availability and quality of data on MSW generation and composition in the Baltic States have been quite poor. To specify and validate the mixed municipal waste composition data in Estonia, a country-wide sorting analysis of mixed municipal waste was carried out in 2008 [18].

The results of the sorting study indicate that even at a relatively high rate of recycling the landfilled mixed municipal waste has a relatively high calorific value (10.5 MJ/kg) due to the high share of combustible materials such as plastic and paper (see Fig. 2).

The quantity of MSW in the Baltic States has rapidly risen as a result of economic growth and increasing consumption. In Estonia approximately 540,000 tonnes of municipal waste (400 kg per person) were generated in 2008. The respective figures in Latvia and Lithuania 1.2 million tonnes (407 kg per person). Earlier forecasts show that MSW will continue to grow rapidly; generation of waste was projected to increase by approximately 50% from 2005 to 2020 [12].

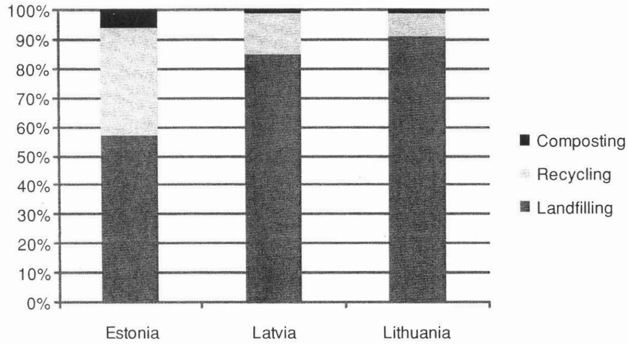

Fig. 3 MSW treatment methods in the Baltic States at 2008 [10]

However, fluctuations in the economic situation could lead to changes in waste generation. This is well illustrated by the impact of the unexpectedly serious global economic decline that has significantly influenced the municipal waste generation rate and made earlier waste generation forecasts questionable. Recent indicators show that MSW generation in Estonia dropped during 2008 and 2009 in correlation with the Gross Domestic Product (GDP) by almost 25%. It can be anticipated that municipal waste generation in Estonia and in the other Baltic States will start to grow along with the recovery of economy. However, it is very difficult to predict the growth rate and time line. Since the number of population is expected to remain roughly the same in all Baltic States, a possible economic development will be the key driving force for changes in waste volumes in the next decade.

Since earlier waste generation forecasts cannot be used anymore, the smaller amount of available mixed municipal waste may significantly influence the number, capacity and financial costs of the possible waste incineration projects in all three Baltic States.

3.3 Waste Management Sector Developments

The success of waste incineration depends largely on the development of the waste management sector including the commercial competition with other waste management options. Here the interest and ability of public sector waste management authorities to control and regulate the local waste management market plays important role.

Landfilling has traditionally represented the easiest and the cheapest option for MSW management. This is the reason why landfilling is still dominating MSW treatment option in the three Baltic States (see Fig. 3). In 2008, the lowest share of municipal waste landfilled was in Estonia (57%) and the highest in Lithuania (91%). Most of the new landfills in the Baltic States are owned by the local municipalities or Regional Waste Management Centres (RWMCs) established by municipalities. All new regional landfills that are established have received financial support in terms of EU funding.

Since the municipal waste management system must comply with the principles and targets of the European waste policy and directives, the role of landfilling will decrease significantly in coming years. Estonia has already achieved a relatively high share of MSW recycling. The main driver for the development of a separate waste collection and recycling systems have been the recovery targets of the EU Directive on Packaging and Packaging Waste 94/62/EC [5] and increasing landfilling gate fees driven by the growing landfill tax [11].

As the Estonian experiences show, the material recycling has been also influenced by the recent fluctuations of market prices of virgin and recycled materials. The high share of impurities in the source separated waste is another limiting factor of an efficient recycling system. The same aspects influence the recycling of collected organic waste. The experiences of composting of organic waste from households show that the quality of compost is low [17]. Another limitation for environmentally beneficial use of compost is the very low market demand for such product. In Estonia, today most of the municipal biodegradable waste based compost is used as a filling material and for landscaping the landfills.

Mechanical–biological treatment as an alternative treatment option for mixed municipal waste has gained recently a lot of attention in the Baltic States. Since there are many problems in running separate collection systems of different municipal waste streams, the MBT is seen as a relatively easy and low cost alternative for treatment of mixed municipal waste. Many private waste management companies as well as municipalities have started to invest into new MBT facilities. However, as the experiences of the first simple and low-cost MBT facilities in Estonia show, the quality of the produced RDF is relatively low. The remaining residual fraction contains significant amount of heavy metals and other harmful or disturbing substances and therefore this should be landfilled.

MBT facilities and conventional mass burn WtE plants compete for the same mixed municipal waste that is available in the region. In Estonia MBT facilities planned or under construction alone already exceed the available municipal waste amount. In Lithuania and Latvia investments to MBT are also a subject of State support that disturbs a fair competition on the waste management market.

In this new situation the public authorities play crucial role. In all three Baltic States the local municipalities are responsible in planning waste management system and organising the waste collection and treatment. Compared to Latvia and Lithuania majority of the municipalities in Estonia are small and therefore they are not able to manage the waste treatment tasks that have been imposed on them. The increased liberalisation and free competition pressure from the government has lead to the situation where the waste management market in Estonia is in a high content controlled by the private sector. Municipalities have very limited ability to direct the waste to certain waste treatment facilities. This has occasionally caused legal problems regarding the ownership of the waste. The extremely liberal waste management market has lead to intense commercial competition between different waste recovery facilities and companies. This also could influence the economy of the possible waste incineration. The WtE facility under the development in Estonia have not managed to sign any waste delivery contracts with municipalities.

In Latvia and Lithuania the municipalities have formed regional waste management centres that plan and develop the waste management on the regional level. They also participate directly in the MSW management by delegating certain functions of waste management, e.g. planning of new waste management infrastructure, operation of regional landfills, civic amenity sites, green waste composting sites, organising tenders for municipal waste collection, collection of fees from waste holders etc. [32]. Therefore, they have stronger waste flow control and general influence over the waste management sector.

3.4 Energy Sector Developments

Large scale WtE facilities, such as municipal waste incineration in combined heat and power plants, could generate significant amounts of energy and are therefore important players in the local energy markets—especially in small countries. It is thus important to establish whether an incineration facility can be integrated into the local legal framework and infrastructure of energy sector. The potential for heat utilisation, defined by the availability and accessibility to the district heating network, is also a very important aspect.

The Baltic energy markets are all undergoing a transition towards new sources of energy and significant changes will take place in the coming years [19]. The biodegradable fraction of MSW is a part of biomass definition, thus it counts as a renewable energy source. The use of MSW for energy production can contribute to achieving the 20% renewable energy goal and the 20% reduction in CO_2 emissions agreed upon at the EU level. Heat and electricity from waste are replacing the energy generated by conventional power plants which still predominantly use fossil fuels.

The Baltic States have all committed themselves to the EU energy goals of 20–20–20 by 2020 (goals which commit EU countries to reduce their energy consumption and emissions of greenhouse gases by 20% and ensure that 20% of energy consumption is covered by renewable energy by 2020) [3]. Twenty percent is the European average but individual goals have been set for each country in order to reach this total average of 20%. By 2020, the percentages of renewable energy in final energy consumption for the Baltic countries should be 25% for Estonia (currently 18%), 23% for Lithuania (currently 15%), and 42% for Latvia (currently 31.4%) [15]. All three countries have introduced policy instruments and support schemes to promote renewable energy generation.

In Estonia, the development within the renewable energy sector is mainly focused on wind energy and biomass/waste based CHP. As a support scheme, producers of renewable energy can sell electricity to a state owned energy company at a fixed high feed-in tariff. There is also support available for CHP and heat production to switch to renewable fuels.

Like the other Baltic countries, Latvia is also obliged to increase the proportion of renewable energy, which could be achieved by installing more biomass/waste based energy production. Several projects are outlined for the modernization of the district heating segment and for making changes in the composition of the fuel utilised there. The emphasis is on the implementation of renewable energy based technologies, especially biomass/waste type solutions.

The Lithuanian energy sector is already in the reorganization stage in order to cope with the shutdown of the Ignalina NPP. The government has launched several initiatives such as the liberalization of the energy market, increasing energy efficiency, broader use of renewable energy sources and promotion of small energy producers. Promotion of renewable energy is one of the main goals of the energy policy in Lithuania. Particular focus is laid on the generation of thermal energy, and 48 million Euros have been allocated for the construction and renovation of bio-fuel/waste boilers and CHP power plants until 2013. In addition, the government has approved a special policy to buy green electricity. The price is by 50–60% higher for biomass energy compared to the current average level of traditional resources, and these prices are state guaranteed until 2020. It is obligatory to buy the supplied green energy and there is a possibility to receive a 40% tax advantage for connecting to the electricity grid.

To have high energy efficiency, waste incineration in a large-scale CHP should take place in large district heating networks where the WtE facilities can function as base load heat providers with both diurinal and seasonal variations.

As in many other Eastern European countries, district heating has a relatively large share in the Baltic States. Most major cities have big district heating networks dating back to the Soviet time. For example in Estonia the share of district heating in heat consumption is approximately 70% [17]. Most of the district heating systems are, however, small and heat is supplied from relatively small-scale boilers (in Estonia 80% of the boilers are less than 1 MW). The largest district heating networks in the Baltic States are located in larger cities (e.g. Tallinn, Riga, Vilnius, Klaipeda, Kaunas), where the share of district heating is close to 90%. The predominant fuels used in district heating systems vary. However, most of the large scale power plants still use fossil fuels (oil, natural gas).

Energy policy and the increasing costs of fossil fuels have given an impetus to local heat producers to transfer from their sometimes rather old and inefficient production technology to the modern CHP technology fuelled by renewable fuels including waste.

3.5 Public Perception

Changes in waste management arrangements in local areas are gaining more attention in media. As a result of greater publicity and higher awareness, many people and organisations are opposing the new waste management facilities. In areas of no public awareness of waste incineration plants, there is usually resentment and distrust towards the environmental and technical performance of such a facility. The experiences in the Baltic States show that the distrust to decision makers and developers may lead to a situation where new waste management facilities of any type are rarely welcomed by the residents close to where the facility is to be located. Public opinion on waste management issues is widely varied and can often be at the extreme ends of the scale. The public opposition to the development of a waste incinerator in Vilnius is a good example of this. Due to the failure to discuss with the public, the project was turned down.

Waste incinerators are still generally perceived as great pollutant sources. Public perception may not be so strong when the WtE facilities are planned to be built in the territory of already existing energy facilities. However, if the same plant has caused environmental problems earlier, the opposition could even stronger.

Therefore, it is important to communicate the waste incineration technology, as well as local and global environmental/health impacts, in a trustworthy and detailed manner. Here the development of the Iru WtE unit could be taken as a positive example. The public, especially the people living in the neighbourhood, was involved in an early stage by using public information meetings and hearings. In addition to EIA, several other studies were ordered from independent institutions to provide comprehensive and independent information about the impacts of the planned WtE unit. This all ensured that the opposition to the Iru WtE was low among the public.

4 The Case of Iru Waste-to-Energy Unit in Estonia

In 2006, the Estonian state owned Energy Company Eesti Energia AS started preparations for the construction of a waste-to-energy unit located in the outskirts of Tallinn, the capital of Estonia.

The main aim was to diversify the production portfolio of Eesti Energia as the company currently produces most of its energy from fossil fuels with rather high CO_2 emissions. In addition there was a need to replace and add to the rather old and oversized energy production units that are not flexible enough to meet the needs of the changing heat and electricity market.

The WtE unit is an extension of the existing Iru CHP plant which has generated electricity and heat in the same location for over 30 years. This CHP plant has been one of the main suppliers of district heat for Tallinn and the city of Maardu. It currently uses natural gas as its main fuel and oil as a reserve fuel.

The new unit will annually burn up to 220,000 tonnes of the mixed municipal waste currently deposited in landfills. This will replace nearly 70 million m^3 of natural gas currently used each year. The facility will use municipal waste mainly from the Tallinn region (the city of Tallinn and the surrounding municipalities).

Table 3 Main parameters of the Iru WtE unit

Thermal treatment unit	Mass burn—MARTIN grates
Number of units	1 × 2.7 tonnes per hour
Capacity	220,000 t/a
Energy efficiency (R_1)	1.28
Heat	50 MW
Electricity	17 MW

More than 50% of the total municipal solid waste in Estonia is generated in this region. However, since the territory of Estonia is small (45,226 km^2), waste can also be collected from other parts of the country.

The construction of the waste incineration unit of the CHP plant started in 2010 and the unit will start to generate electricity and heat from waste in 2013. As the first plant for the thermal utilisation of municipal waste in the Baltic States, the Iru WtE facility can be considered as a pilot project for similar WtE projects in the region.

4.1 Incineration Technology

Modern mass burn waste incineration technology was chosen for the WtE unit as the most reliable, economically feasible and proven technology for commercial use. This type of incineration technology is flexible enough to burn different waste streams without pre-treatment and for producing heat and electricity at a price that is acceptable and competitive in the market. The mass burn technology is the most common commercially used waste incineration solution in the neighbouring Scandinavian countries and there are over 400 similar plants around Europe [9, 27]. A WtE unit that runs on technology similar to that of Iru will soon be completed in the Finnish capital Helsinki.

The available mixed municipal waste with the average calorific value of 10.5 MJ/kg and ranging from 8 to 15 MJ/kg will be incinerated on a modern air-cooled moving grate. At a waste throughput of 27.5 tons per hour (220,000 t/a) it converts about 82% of the energy in the waste into electricity and heat. The thermal energy capacity of the WtE plant will be 50 MW, while the electricity generating capacity is planned to be 17.3 MW. This should complement and partly replace the existing capacities of the Iru CHP plant.

According to the new Waste Framework Directive energy efficiency calculation formula R_1 the energy efficiency of the Iru WtE unit is as high as 1.28. As such the WtE unit complies with the energy efficiency criteria of the directive and can be considered as a R_1 recovery operation of waste (Table 3).

The main components of the energy block are as follows:

- Waste receiving hall and storage bunker
- Automatic feeding and mixing system (crane and waste feed hopper)

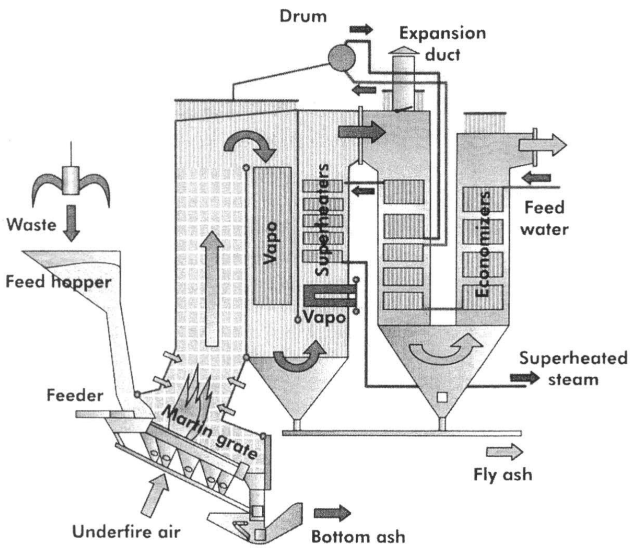

Fig. 4 The main components of the energy unit of the Iru WtE facility [35]

- Combustion unit (MARTIN type reverse-acting grate)
- Vertical heat recovery boiler (CNIM)
- Steam turbine and generator
- Pollution control system including flu gas treatment plant (CNIM/LAB semidry system) (Fig. 4).

The MARTIN reverse-acting grate is the key component of the combustion system. It consists of several parallel runs inclined at an angle of 26°. Each grate run has its own drive and feeding device. Grate bars made from a wear and temperature-resistant chromium-steel alloy are assembled to form grate steps. Alternating fixed and moving grate steps make up a grate run.

The reverse-acting movement ensures that the grate surface is always covered by a protective layer of waste or ash. Thermal wear due to heat irradiation from the furnace does not occur, and consequently grate bar life times are long. Water cooling is not needed, even with very high waste heating values.

The combustion air is divided into underfire air, which passes the grate surface, and overfire air, which is injected into the furnace above the grate. Each grate run is divided into several under grate air zones. The underfire air is distributed as needed locally—it is not required to cool the grate bars.

There is a defined pressure drop as the underfire air passes through the grate surface, providing a uniform distribution within each zone. The air gaps between the grate bars are constantly kept clean by means of a relative movement of the grate bars with regard to their adjacent bars at the end of each stroke.

The heat released from the combustion of waste is recovered in a water tube boiler which forms an integrated unit with the grate. The boiler is of the vertical type, top-supported, and includes one steam drum and five vertical passes. The superheated steam produced by the boiler feeds a turbo-generator set of the back pressure type. Three steam bleeds are provided to ensure both the feeding of the district heating network and the feeding of internal consumers (air pre-heater, deaerator, etc.)

Many of the fittings of the new unit will be installed in the existing plant. The unit will use the 202.4 m chimney of the Iru plant. The turbine and the generator will be installed in the existing building and the existing office buildings and other auxiliary rooms will be used as well.

The pollution control system of this incinerator includes a flue gas cleaning process, a wastewater treatment unit, an odour and noise control system and an ash management system.

Waste management will be conducted in sealed rooms to prevent the spread of offensive odours beyond the unit. Trucks will drive into the building and dump their loads directly into a deep bunker. The air needed for combustion of the waste will be drawn from the waste unloading room and the waste bunker. This will ensure that there will constantly be a low air pressure in these rooms and that an inward draught will be created when the door is opened.

The flue gas cleaning process comprises an active carbon and semi-dry lime scrubbing process followed by particle removal in a fabric filter followed by a two-stage wet scrubbing process. The waste scrubbing process will remove a vast majority of HF, HCl, SO_2 and Hg left from the semidry stage. In order to avoid wastewater from the flue gas cleaning process, the small amount of wastewater from the wet process is evaporated in the boiler.

Reduction of dioxin takes place by adding activated carbon to the flue gas prior to the fabric filter, where dioxin and activated carbon are collected together with fly ash and FGT-residues.

Reduction of NO_x from the combustion process will take place in a selective non-catalytic reduction (SNCR) process by injecting ammonia water (NH_4OH) into the first pass of the boiler, thus securing compliance with the Waste Incineration Directive 2000/76/EEC.

Since the facility will utilise a semi-dry flue gas cleaning system, it is designed with zero wastewater discharge. This is accomplished via reuse of wastewater produced by the facility.

Separate systems will be implemented for the drainage, treatment and discharge of rainwater, including roof water, so that it does not mix with the potentially or actually contaminated wastewater streams. Surplus rainwater which cannot be stored on site, will be discharged to the public sewer.

The facility will utilise water from the Pirita River and to a lesser extent ground water. Under normal operating conditions, the water consumption is approximately 6.5 m^3/hour.

The main solid waste streams generated by the WtE unit are bottom ash, fly ash and flue gas treatment residue.

Bottom ash constitutes the largest percentage of solid waste resulting from the combustion process. After burnout of the waste at the end of the grate, the combustion bottom ash falls down the bottom ash chute into the water bath of the wet ash extractor. Bottom ash consists of inert materials from the combustion process such as glass, metal, earth and other fractions. The bottom ash is magnetically screened to recover ferrous metal. Separated metallic scrap will be sent to recycling. Ash is stored in a separate bottom ash bunker with sealed surfaces. The bottom ash bunker offers a temporary storage capacity of approximately 900 tonnes. This is equivalent to the amount of bottom ash produced over a period of approximately 4 days. It is planned to dispose of bottom ash at a landfill. Also possibilities to use it as a construction aggregate substitute are studied.

The flue gas treatment residue containing fly ash, calcium-based salts, lime and activated carbon (or coke) is collected in the hopper(s) of fabric filters. The flue gas treatment residue is transported pneumatically to two fully enclosed silos/steel tanks. It is planned to send the flue gas treatment residue either to a special hazardous waste landfill or to Germany for filling up old coal mines.

4.2 Economic Aspects

The economic cost of a WtE facility is influenced by local circumstances related to the size and design of the factory, legal aspects, labour cost, the cost of consumables, potential for heat utilisation, market price for energy, etc.

The total investment costs of the Iru WtE unit are approximately 98 million euros. Although the technology supplier carries out all the engineering, procurement and construction works (EPC contract) providing a fully-equipped facility ready for operation ("turn of the key"), the capital costs are relatively low—445 euros per tonne of installed capacity. Based on the experience of other similar WtE facilities the capital costs are generally 600–900 per tonne of installed capacity [17, 30]. The low investment costs have been attained mainly by integrating the new energy unit tightly with the existing power plant. Capital costs lower than that have been achieved in Europe only by very experienced energy companies who already own or operate several plants and have the in-house competence for construction and management of waste incineration plants.

The capital costs can be split into different components. In Table 4 the total capital costs are split into four main components. For each main component, the percentage of the total capital costs related to the specific component is shown.

It is difficult to estimate the operational costs of the WtE unit since they depend on several variable cost items such as cost for residue disposal, maintenance, salaries, etc. It could be expected that the operational costs will be about 50–70 euros per tonne of waste. The European average operational costs per tonne fall

Table 4 General distribution of the total capital costs of Iru WtE unit

Component	Percentage of capital costs (%)
Thermal processing equipment and flue gas treatment (grate, boiler, etc.)	60
Energy production equipment and electrification (steam turbine, generator transformers, etc.)	15
Civil works	10–15
Miscellaneous	10–15

Table 5 General distribution of the operational costs of Iru WtE unit

Component	Percentage of operational costs (%)
Labour and consumables	25–35
Maintenance	20–30
Residues (management and disposal)	40–50

into the same range (45–70 euros per tonne of installed capacity) [17, 30]. The operational costs can be split into different components as indicated in Table 5.

Sale of energy is a significant element in the economy of waste incineration. The potential energy production and income from energy sale depend heavily on waste composition (calorific value), the potential for heat utilisation and market price for energy (heat and energy).

The average market price for electricity (Nord Pool) was about 45 euros/MWh in 2010 and the competitive heat price was about 30–35 euros/MWh. Additionally, there is a subsidy for electricity produced in CHP (32 euros/MWh).

The estimated income from energy sale covers up to 80% of the total costs. In Europe, the average is about 40% [26].

Net treatment costs can be calculated based on the estimates of costs and potential income from sale of energy. Using the costs presented above a rough estimation of the net costs of waste incineration shows that the gate fee for MSW treatment at the Iru WtE unit is approximately the same as the current average landfill gate fee in Estonia (45 euros/tonne). In the case of higher energy prices in the future and other favourable conditions the gate fee for MSW incineration could be even lower (ca 20%). However, if one or more of the critical preconditions fail (especially waste supply, calorific value of waste or energy prices), the actual net treatment costs may be severely influenced.

4.3 Environmental Impacts

A number of environmental impacts are linked to the incineration of waste. In addition to site- specific impacts that are studied usually in the frame of EIA, indirect impacts/emissions should be taken into account when assessing the net impacts of WtE facilities. As incineration of municipal waste should fit into the overall waste management system of the region, it should be compared with alternative waste management options. Life cycle based environmental assessment methodologies can help to identify an overall, optimal environmental solution for managing MSW, without risking that the decision (e.g. to build an incineration plant) will result in a more negative overall impact [4, 33, 34].

Since the Iru WtE unit will be built as an extension to the already existing plant, the local site-specific environmental impacts will be relatively small. The summary of an EIA report indicates that the potential increase in direct air emissions is 0.01–1%. However, the indirect atmospheric emissions will notably fall. This is because municipal waste will replace the current fossil fuels such as natural gas and oil. Waste incineration with energy production will partly offset the emissions that occur when energy (both electricity and heat) is produced from fossil fuels. This is especially important concerning the climate change impact in terms of greenhouse gas emissions.

Large scale municipal waste incineration has to be discussed within the context of the overall waste management strategy. As part of the environmental impact assessment an additional Life Cycle Assessment study was carried out [20]. The aim of this study was to evaluate on how MSW incineration in the Iru WtE unit will influence the life-cycle based environmental impacts of the Estonian municipal waste management system. The incineration-based scenario was compared with an alternative scenario where legal waste management targets in Estonia are achieved by intensive material and biological recycling of municipal waste. For the environmental impact assessment the waste management situation in 2000 was taken as a starting point or a base scenario. The LCA model for waste management planning WAMPS was applied for assessing the environmental impacts of the studied waste management scenarios [17]. In this paper the results of the LCA study are expressed in terms of a climate change impact (GHG emissions) as the most important global environmental impact category of waste management.

Both studied scenarios are in compliance with the legal requirements and recycling targets of the relevant EU directives. It was assumed that waste composition remains the same during the studied period (2000–2020). It was also assumed that after the economic crisis the amount of MSW will continue to increase. For both scenarios it is also assumed that all landfills in Estonia will be equipped with a landfill gas collection system by 2010 at the latest and the landfill gas recovery rate will increase by up to 50% by 2020. Before 2010 the collected gas is flared and after 2010 it is used for electricity and heat production, which is substituting oil shale based electricity and natural gas based heat used for district heating. The energy produced in waste incineration will replace the electricity produced from oil shale that has a very high climate change impact in terms of CO_2 emissions, and heat from natural gas.

Base scenario (scenario 0)—In 2000, waste management in Estonia primarily involved landfilling of MSW (92% of the total MSW). There was no landfill gas collection in landfills at that time. Only a small amount of packaging waste

Table 6 Municipal Solid Waste management scenarios

Scenario	Material recycling (%)	Biological recycling (composting) (%)	Incineration (%)	Rest waste (landfilling) (%)
2000 Base scenario	4	4	0	92
2020 Scenario 1	27	15	45	13
2020 Scenario 2	27	37	0	36

(mainly PET-bottles and cardboard) was collected separately and sent to recycling. There was no centralised collection system for biodegradable waste. Approximately 17,000 tonnes of biodegradable waste (mainly garden waste) were composted by the households (4% of the total MSW). It is assumed that the share of home composting will remain the same till 2020.

Material recycling with intensive incineration (scenario 1)—This scenario is a projection for 2020, where the dominant option of MSW management in Estonia is incineration. 45% of total MSW generated in Estonia is incinerated in the Iru WtE unit. A large amount of the generated heat could be utilised since Tallinn has large dwelling areas with district heating system. In this scenario increased amounts of recyclable materials (mainly packaging, paper, cardboard and metals) are collected separately and recycled to meet the recycling targets of the EU Packaging Directive. About 30% of the waste material is expected to be recycled. As incineration is already contributing to the reduction of biodegradable waste, the share of biological recycling is not expected to exceed 15% of the total MSW. Centrally collected kitchen waste is composted by using the static composting method with forced aeration. Collected garden waste is composted in open windrows. Intensive material recycling and incineration leads to a relatively small amount of rest waste, which is landfilled (13% of the total MSW).

Material recycling with biological recycling by composting (scenario 2)—This scenario is a projection for 2020, where the legal targets are achieved by material and biological recycling. Also in this scenario material recycling is expected to amount to up to 30% of the total MSW. The Landfill Directive requirement to divert biodegradable waste away from landfilling is met by increasing composting to 37% of the total MSW. An increased amount of wet biodegradable waste is composted by using the centralised reactor-composting method (without gas collection and energy recovery). It is assumed that the remaining waste will be deposited in a landfill (Table 6).

The results of the scenario analysis regarding net GHG emissions are shown in Fig. 5. The diagram shows net GHG emissions from the waste management system minus saved emissions in the background system. When the emissions from the studied waste management scenario or waste management practice are lower than the saved emissions in the background system then net result is negative.

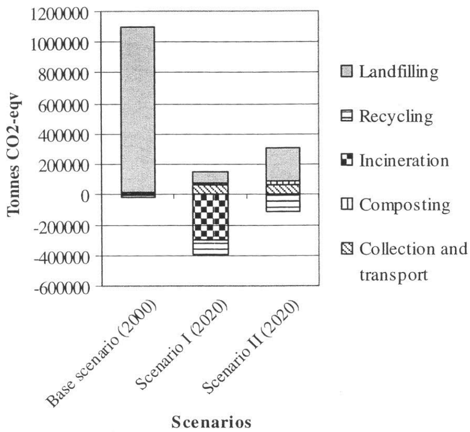

Fig. 5 Emissions of net GHG from the studied waste management practices and scenarios, 2000–2020 (CO_2-equivalents, tonnes)

When comparing the two scenarios we can see that the incineration scenario (scenario 1) has a higher climate protection potential than the alternative scenario (scenario 2). In the incineration scenario where high rates of recycling and incineration with energy recovery are attained, the net emissions of CO_2-equivalents are even negative. The reason for the negative net GHG emissions is a relatively low amount of waste sent to landfills as well as a high share of material recycling (avoided primary production of materials) and the recovered energy in Iru WtE unit (avoided emissions as a result of replacing heat and electricity produced from natural gas and oil shale in the background system). In Estonia electricity produced from waste replaces oil shale based electricity which has a high climate change impact in terms of CO_2 emissions.

Incineration gives approx. 75% and recycling almost 25% of the total avoided emissions. In scenario 2 GHG savings are attained mainly due to material recycling and the avoided emissions from landfilling. As in this scenario composting without energy recovery is applied, the net GHG emissions are higher than in the incineration scenario.

Direct emissions from landfills continue to be a major source of GHG emissions till 2020 despite of the fact that the landfilling rate will significantly decrease and a relatively high share of landfill gas is recovered in both studied scenarios. GHG emissions from waste collection and transport will increase until 2020 due to increased recycling. In scenario 2 a higher collection rate of biodegradable waste causes slightly more emissions of CO_2-equivalents. However, collection and transport of waste accounts for a relatively small amount of the estimated net GHG emissions in both future scenarios.

5 Conclusions

The three Baltic States are in the stage of changing their municipal waste management systems since they have to comply with the principles and targets of the European waste policy and directives. Over the past years, thermal treatment of MSW has been discussed more intensely in these countries as one of the waste management options that could help to reach the legal targets in a relatively short time.

In general, the Baltic States have similar socio-economic characteristics, waste and energy sector developments and geographical conditions that form similar frameworks for the development of a waste management infrastructure, including possible WtE options. However, as experience from feasibility studies and the first WtE projects shows, there are several local and regional factors that could significantly influence the economic success of waste incineration. Since the service life of a waste incinerator is usually over 30 years, the possible future trends and developments of these influencing factors have to be carefully studied before a decision for waste incineration is made.

The legal framework on waste management in the three Baltic States is based on the same general requirements. However, it is important how the policy implementation is facilitated. Estonian experience shows that a relatively high landfill tax together with a ban for landfilling of unsorted MSW have resulted in favourable conditions for the development of new recovery facilities, including waste incineration. Calculations of the economic costs of the new Iru WtE unit in Estonia indicate that incineration of waste has already today a competitive advantage in terms of a lower gate fee compared to landfilling and other new mixed waste recovery options such us MBT.

Investments in waste incineration presume a steady fixed stream of waste with high calorific value, to ensure financial viability. The results of recent sorting studies show that mixed municipal waste in the Baltic States has a relatively high calorific value due to the high share of combustible materials. However, municipal waste composition as well as the amount may change in time because of either additional recycling or changes in the socio-economic situation in the region. This is well illustrated by the impact of the recent economic crisis that significantly reduced the mixed municipal waste generation rate and made earlier waste generation forecasts questionable. This has lead to the situation that there might be not enough mixed municipal waste for all the planned waste recovery facilities. For example, the planned Iru WtE unit will treat most of the mixed municipal waste that today is landfilled (approximately half of the total municipal waste generated per year). As such it will significantly influence the economic performance of all other municipal waste management options. However, conventional mass-burn incineration plants have a competitor in the form of MBT facilities, because they compete for the same mixed municipal waste that is available in the region.

In this new situation with a more liberal waste market and the ever increasing commercial competition between different recovery facilities, the ability and willingness of public authorities to regulate and control the waste sector plays a crucial role. In Estonia, local municipalities have less control over the waste market than in Latvia and Lithuania where municipalities have formed regional waste management centres that participate directly in MSW treatment services and infrastructure projects.

Due to the EU energy policy goals the Baltic energy markets are undergoing a transition towards new sources of energy. Municipal waste contains a large amount of biological and renewable materials, and is therefore a promising source of renewable energy. As a consequence, WtE option is becoming more interesting as a potential contributor to energy security and diversification and matches the growing demand for renewable energy. All three Baltic States have introduced policy instruments and support schemes to promote renewable energy generation. Another favouring factor is the high potential for heat utilisation of WtE facilities due to rather cold climate and existing big district heating networks in larger cities. As the studied Iru WtE unit show, the relatively high value of district heat and support schemes for RES make the average energy revenues much higher in the Baltic States than in many other European countries.

Public perception is an important factor that should be taken into account when developing WtE facilities. The experience in the Baltic States shows that distrust of decision makers and developers may lead to a situation where new waste management facilities of any type are not welcomed by the residents close to where the facility is to be located. Waste incinerators are still generally perceived as great polluters and it is very difficult to convince the public that health and environmental risks are under control. Therefore, it is important to well plan and conduct the public involvement process and to communicate the technology related, as well as the environmental and health impacts, in a trustworthy and detailed manner.

From the decision-makers perspective, when developing sustainable waste management plans at the national or regional level, it is also important to take into account the full life-cycle and the related environmental and economic benefits/trade-offs associated with alternative waste management options for achieving the targets. As the results of the LCA study of Iru WtE unit show, waste incineration with energy recovery can partly offset the emissions that occurred when energy was produced from fossil fuels. This is an important aspect in the context of climate change, since the oil shale combustion technology currently used in Estonia to generate electricity has a very high climate change impact. This means that the municipal waste management scenario where a high share of recyclable waste fractions are sent to material recycling and the maximum amount of rest waste is incinerated with energy recovery should be preferred from the environmental impact point of view.

In general, it can be concluded that thermal treatment of MSW in large scale WtE facilities has a relatively good outlook in the Baltic States. Although the initial investments are relatively high, the favourable conditions in the energy sector allow the WtE facilities to treat municipal waste at a relatively low cost. Therefore, it can be expected that WtE provides an environmentally and economically efficient way to meet the stringent EU waste management targets. However, large scale municipal waste incineration has to be discussed within the context of an overall waste management strategy, rather than as a single option.

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PAPER IV

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Assessment of the applicability of the Pay As You Throw system into current waste management in Estonia

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Abstract

Purpose: This paper assesses the current waste management situation in Estonian municipalities and outlines the main constraints hindering the implementation of the Pay-As-You-Throw (PAYT) system into the existing waste management model.
Design/methodology/approach: Data pertaining to the treatment methods of municipal solid waste (MSW) and the ability to implement the PAYT system were gathered from 150 of the 226 local municipalities, whilst statistical data related to the amounts of MSW generated and separately collected at a municipal level were obtained from the Estonian Environmental Information Centre.
Findings: The results of the study showed that 39% of the municipalities sort waste before landfilling. To increase the sorting ability of inhabitants, 43% of those municipalities that responded to the questionnaire suggested enhancing awareness among people in regard to waste handling. It was found that people are not economically motivated to sort their waste due to the fact that differences in charges between separately collected and unsorted waste are negligible. It was estimated that implementing the PAYT system in one rural municipality would increase cost of emptying containers by approximately 20–45%.
Practical implications: Results of our study can be used in countries with a comparable economic situation to improve their current economic and legislative context in the field of sustainable waste management.
Originality/value: The novelty is that authors aimed to assess the possibility of implementation of Pay-As-You-Throw system in practice, using Estonian municipalities as a case area, including economic feasibility and willingness of stakeholders to apply the system.

Keywords: municipal solid waste management; weight-based collection; pay-as-you-throw.

1. Introduction

The approach of Pay-As-You-Throw (also known as variable rate pricing, unit pricing, and differentiated tariff system) in waste management is to realise the concept of sustainable development – "polluter pays principle" – in a fair manner by charging people according to the amount of waste they actually generate. In contrast to the common, prevailing approach of waste charging whereby collection services are invoiced in the form of a fixed recurring fee and/or in association with payments calculated on the basis of living space, the number of household members or certain other supplies such as water and electricity, households under the PAYT scheme pay a variable amount depending on the quantity/quality of waste generated by them and the corresponding service they obtained for its disposal. Such a direct form of unit pricing for waste primarily aims to stimulate households to divert an increased portion of their waste away from traditional means of disposal to recycling, a priority waste management approach that is believed to be less harmful to the environment and more cost effective in the long run.

Skumatz (2008) reported that 25% of the population was covered by the PAYT scheme in the US in 2006. The system decreased residential municipal solid waste (MSW) disposal by about 17% and increased the rate of recycling by 8–11%.

In Europe, the rate of PAYT implementation varies greatly. For instance, in 2005, in Western Europe (excluding new EU member states) the rate varied between 'largely without PAYT experience' in Spain, Portugal and Greece and 'area wide implementation' in Sweden, Germany and France (Bilitewski, 2008). In Sweden, municipalities with pay-by-weight schemes collected 20% less household waste per capita than other municipalities (Dahlen and Lagerkvist, 2010). Similar positive results such as an increase in sorting ability and the minimisation of landfilled waste appeared in France and Germany (Bilitewski, 2008; Le Bozec, 2008; Reichenbach, 2008).

Among the new EU Member States, Czech Republic has approved PAYT applicability in some municipalities (Sauer *et al.*, 2008). Citizens there separate more waste, while producing less residual waste. In Estonia, in 2009–2011, the HEC-PAYT project assessed the potential viability of PAYT. Legislation, financial tools, socio-economic and cultural aspects and practices, charging and billing mechanisms, and the security of cost recovery options were considered along with the current status of solid waste management practices. Results showed that the current situation in Estonia does not facilitate implementation of the PAYT system. Estonia has just constructed a new MSW recovery facility, a mass-burn incineration plant, which will work as the main MSW management solution in the near future. Similarly, Lithuania is constructing a mass-burn incineration plant, while another is planned. The development of a national waste management policy primarily depends on the choice made by the decision-makers at national and local level as well as on the type of business.

The authors assumed that PAYT system implementation for large and small municipalities would be different, as it was difficult to find any specific information on this issue. In the study, the possibilities of PAYT implementation in a larger city were assessed by analysing factors that influence the success of PAYT. The economic feasibility of PAYT implementation has been evaluated in a selected rural municipality in Estonia.

Batlewell and Hanf (2008) demonstrated that PAYT characteristics depend on the implemented waste management system, which are determined by socio-economic, political, and cultural systems, which, in turn, depend on natural systems (Fig. 1). The successful implementation of the PAYT model therefore requires the favourable conditions of these systems.

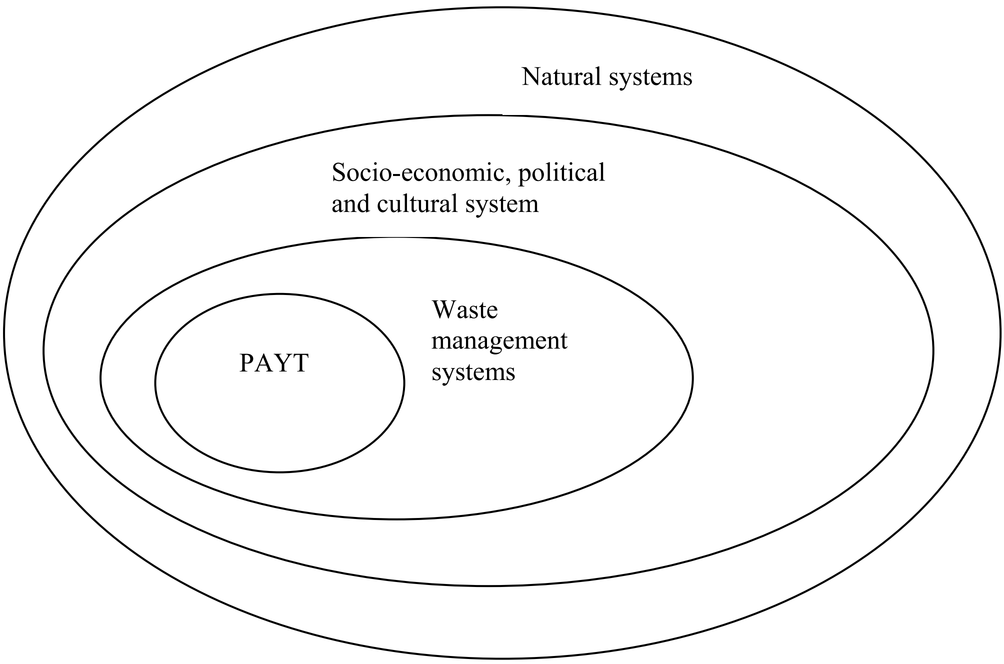

Fig. 1:Contextualisation of PAYT systems (redrawn from Batlewell & Hanf, 2008)

As the PAYT model only charges for the amount of residual waste, it can achieve environmental improvement if, in parallel, waste separation, minimisation, recycling and reuse programmes are undertaken and effectively implemented (Batlewell & Hanf, 2008; Bilitewski, 2008; Chowdhury, 2009, Reihenbach, 2008; Zotos *et al*., 2009). Therefore, the main aim of this study was to evaluate the interest of municipalities in the implementation of the PAYT system and their practical ability of its application in their region. To evaluate the results, we compared the factors of the successful implementation of PAYT with those in the city of Dresden, Germany, where an electronic bin identification system was implemented in 1995 and its two-component charging system (basic fee and service fee) for residual and organic waste collection in 2003.

2 Study area and methods

Administratively, Estonia is divided into 15 counties and 226 local municipalities, of which 33 are towns and 193 are parishes.

In Estonia, the volume based municipal solid waste collecting scheme operates in all local municipalities. Municipalities were subdivided into size classes, according to the number of inhabitants (Table 1). Since 2005, according to the Waste Act (RT I 2004, 9, 52), every local authority in which the number of residents exceeds 1,500 must implement the organised waste collection system. This system aims to link all waste holders into a common waste management system to minimise illegal dumping and the misuse of waste bins. This volume of inhabitants was taken as a basis for the subdivision of municipalities into size classes, probably due to the limited capacity of small rural municipalities to implement an organised waste collection scheme. The number of smaller rural municipalities (class I) not obliged to join the organised waste handling remains high. One solution is to merge municipalities into bigger administrative units, which would help reorganise the existing waste management system. At present, this merging is a voluntary process; therefore, it is very slow.

Table 1: Size classes of local municipalities according to the population size

Class	Population size	No. of municipalities	Description
I	<1,500	90	Rural municipalities that are not obliged to join the organised waste collection system.
II	>1,500	103	Rural municipalities that are obliged to join the organised waste collection system.
III	>1,500	33	Cities that are obliged to join the organised waste collection system.

A more detailed study was conducted in two municipalities. The city of Tallinn was chosen as the larger city (class III) and Kuusalu rural municipality as a typical parish (class II) with several possible advantages for the implementation of the PAYT system.

2.1 City of Tallinn

Tallinn is the capital of Estonia, with a total population of about 400,000 inhabitants and an area of 159.2 km². The residents of Tallinn produced 142,550 tons (343 kg per capita) of municipal solid waste in 2011, which included residual waste, paper and cardboard, biodegradable waste, garden waste, packaging, large waste and waste electronics (Estonian Environment Information Centre, 2011). The management of waste is mainly regulated by the Waste Act, together with Regulation no. 303 of the Government of the Republic from 25 September 2001 on the "Procedure For Arrangement conducting Of Public Competition For Granting Special or Exclusive Rights". Tallinn's waste is transported to the collection point by the company who wins the public tender that has been organised by the Environmental Office of Tallinn (RT I, 2004, 9, 52). The price of waste management services is fixed following the bidding competition. However, the city council may amend the reference service prices if transportation costs or the cost of constructing, using, closing or continuing the maintenance of a waste handling facility increases. Tallinn City Government organises the separate waste collection of packaging waste, hazardous waste, used batteries, biodegradable waste, used house appliances, used clothing, old tyres, paper and cardboard, bulk waste, construction and demolition waste, garden and park waste by providing public containers.

2.2 Kuusalu rural municipality

Kuusalu rural municipality has about 6,880 inhabitants and an area of 708 km^2. An organised waste handling system is established in three small towns and 58 villages. An organised waste collection was initiated in Kuusalu in 2006. The rural municipality uses a four-level waste management system:

- first level – organised collection of MSW and biodegradable waste;
- second level – public containers for mixed packaging, glass, paper and cardboard ;
- third level – Kuusalu waste station, where hazardous waste, paper and cardboard, packaging, electrical and electronic equipment, sheet glass, construction and demolition waste and tyres are collected;
- Fourth level – household waste collection by companies.

2.3 Methodology

The analysis of the waste management system and the assessment of the main obstacles that hinder implementation of the PAYT system in Estonia were based on information collected from 150 of the 226 municipalities in Estonia. The collected information included questions about MSW treatment methods (recycling, reuse, incineration, composting, and mechanical-biological treatment, landfilling or other) and ranked them according to priority as well as the options for enhancing the sorting ability in their municipality and to assess possible implementation of the PAYT model. Descriptive statistics and regression analysis were used to structure and assess the data. In order to limit the number of observations, mean values were used to generate trends of MSW production and collection in all municipalities in Estonia. Linear regression was applied to assess the correlation between MSW generated and separately collected in Estonian municipalities.

The assessment was based on the methodology suggested by Bilitewski in 2004, which was to assess the factors that influence the success of PAYT system implementation through the symbiosis of technical, economical, political and social systems (Table 2). More detailed analysis of those systems was done in terms of assessment of the effectiveness of PAYT implementation in Tallinn.

Table 2: Factors influencing the success of PAYT system implementation (Bilitewski, 2004)

Factors	Description
Technical	The possibility of waste separation in terms of responsibility (who is responsible for the organisation of MSW collection and treatment) and administrative capacity (practical application of MSW collection and treatment) was assessed.
Political	The legal framework in the waste management field was evaluated.
Social	Population density and environmental awareness among people
Economic	Economic issues in terms of costs for households, calculation of billing and necessary investments were assessed in the case of one rural municipality.

Data concerning amounts of waste generated, recycled, landfilled and eliminated at national, county, and municipal levels were provided by the Estonian Environment Information Centre (EEIC). At a municipal level, information regarding total volumes of generated MSW was obtained for 75% of municipalities, including 136 rural municipalities and 33 towns.

3 Current situation in waste management in the Baltic States

3.1 Overview of MSW treatment methods

In 2010, landfilling remained the dominant treatment method for MSW in Estonia as well as in other Baltic states (Fig. 2). This is despite the fact that the majority of the old unsanitary landfill sites were closed between 2004 and 2010 in all three states. For modern landfill facilities, the need to meet the technical requirements of the European Union, such as leachate treatment and gas collection systems, still remains challenging. In addition to landfilling, Estonia operates four mechanical biological treatment plants (MBT) while Latvia runs five centralised operational composting facilities and one anaerobic digestion plant (Moora, 2011).

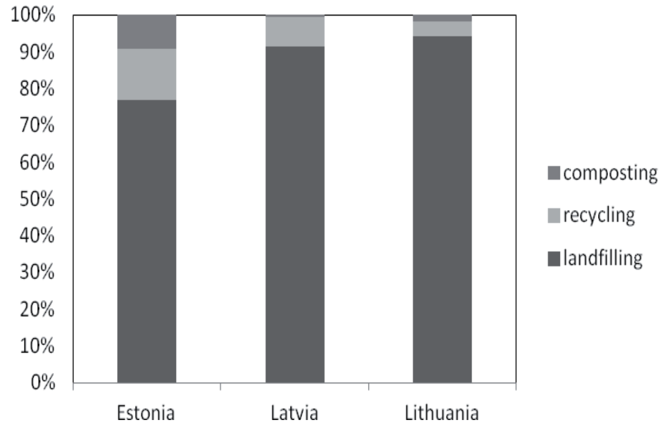

Fig. 2: MSW treatment methods in the Baltic states in 2010 (Eurostat, 2012)

In 2008, Estonia banned the landfilling of unsorted municipal waste. Municipalities must organise source separation of paper and cardboard, green garden waste and hazardous waste, as well as packaging waste. A 20% decrease in the landfilling of non-hazardous waste in 2008 was also due to higher landfill taxes and the economic downturn (EEA, 2010).

Latvia develops polygons and collecting systems for non-hazardous municipal waste as well as systems for the collection and treatment of hazardous waste. The country has also prioritised biogas collection and use for energy production from biodegradable waste and sludge (EEA, 2010).

Lithuania has banned the disposal of waste at landfill sites that do not comply with EU requirements since mid-2009. These sites were closed in 2011. Since 2007, Lithuania has mainly invested in the development of a municipal biodegradable waste management infrastructure (EEA, 2010).

3.2 Organised MSW handling in Estonia

The amounts of MSW produced in Estonia between 2006 and 2010 fluctuated (Fig. 3), most likely due to the rapid economic growth from 2007–2008, followed by the economic downturn since 2008 when reduced consumption significantly decreased the amounts of MSW. Most waste was produced in Tallinn and its densely populated catchment areas in Harju county (Fig. 3).

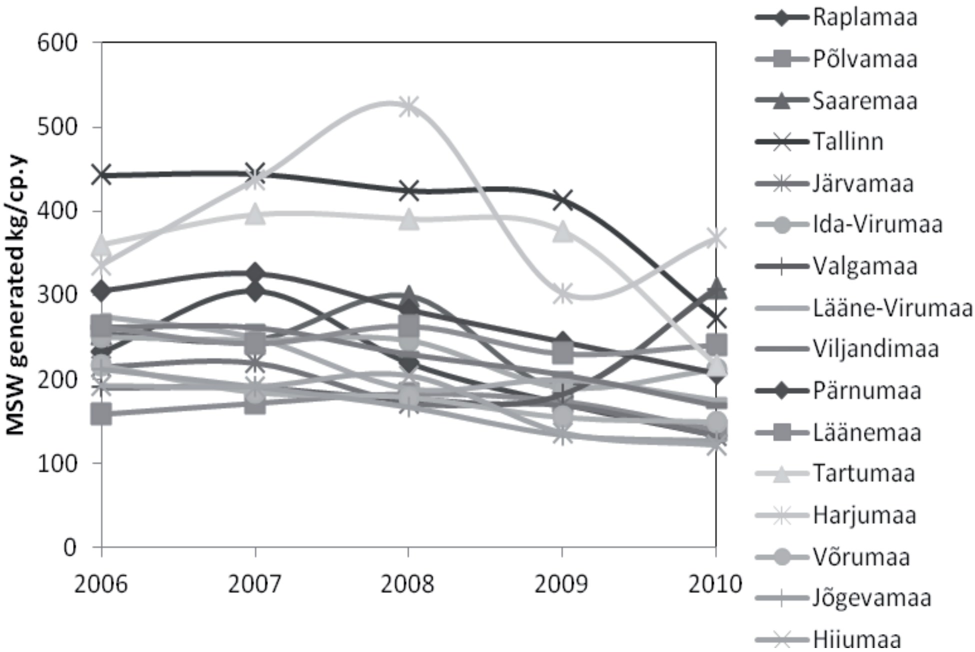

Fig. 3: MSW generation in Estonia by counties
(Estonian Environment Information Centre, 2012)

Each municipality with more than 1,500 inhabitants must have at least 95% of its residents within the organised waste handling system (Ministry of the Environment, 2012). However, actual involvement remains considerably lower. Moreover, some entire municipalities have remained outside of the organised collection system (Fig. 4), mainly because the total number of residents remains below the threshold level.

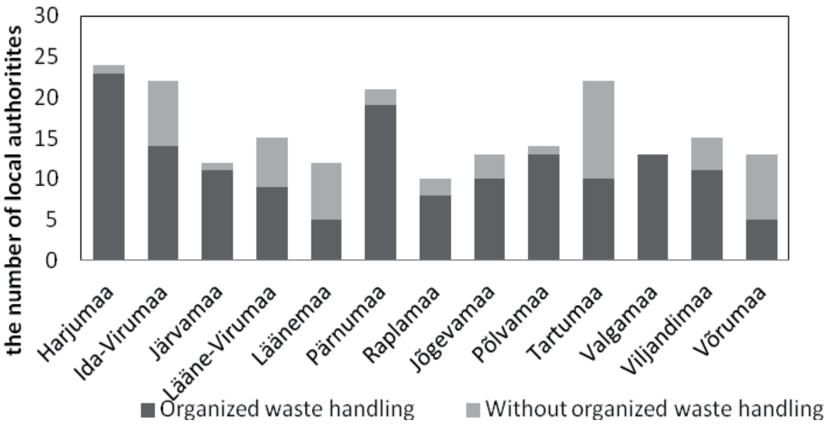

Fig. 4: Organised waste handling by local municipalities in 2009 (Ministry of Environment, 2012)

To facilitate separate waste collection, local Estonian municipalities must provide containers to residents for paper and cardboard, mixed packaging, glass, biodegradable waste and mixed municipal waste.

Local municipalities are obliged to organise a bidding competition between waste handling companies and set a marginal rate for waste handling. The company that offers the lowest price wins the public tender and serves the region for the next five years. All residents in this region are obliged to make an agreement only with this company. Another factor against PAYT is the shortage of financial resources in the budget of local municipalities. Their income depends on pollution tax from MSW landfilling. 75% of the pollution tax that is paid for MSW landfilling in the region goes towards the further development of waste management by the local municipalities. As the landfilling rate is decreasing, municipalities are struggling financially to reform the system.

4 Results and discussion

4.1 Survey among Estonian local municipalities

Trends for MSW generation and separate collection have been observed since 2004 in local municipalities in Estonia (Fig. 5). As is evident from the figure, cities and rural municipalities with populations greater than 1,500 people produced more waste compared to the smaller sized rural authorities. The separate collection of waste in those areas was also more stabilized. This can be explained by an obligation to be involved in organised waste collection system, which helps to systemise MSW collection and handling.

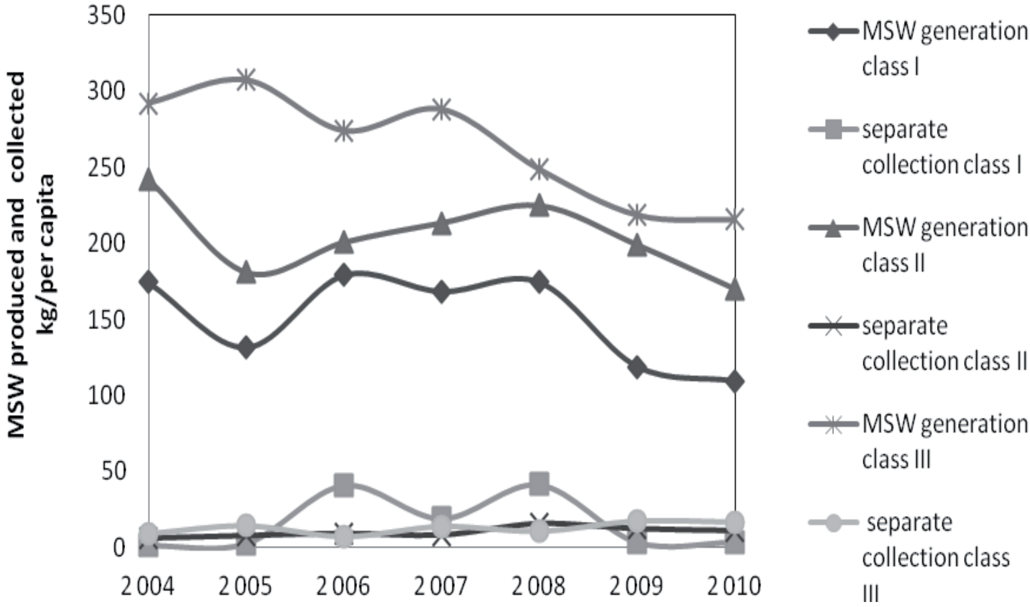

Fig. 5: Trends of MSW generation and separate collection in Estonian municipalitites

The results of linear regression analysis showed no correlation between the amounts of MSW generated and separately collected (Fig. 6). The coefficient of determination R^2 was 0.208 for class I and II municipalities and 0.279 for cities, class III. Most likely, the sorting rate primarily depends on an organised and operational waste collection system. Often, the sorting rate is very low in different municipalities and the minimisation of generated MSW amounts, which was observed in the past few years, was not reflected in the separately collected waste amounts.

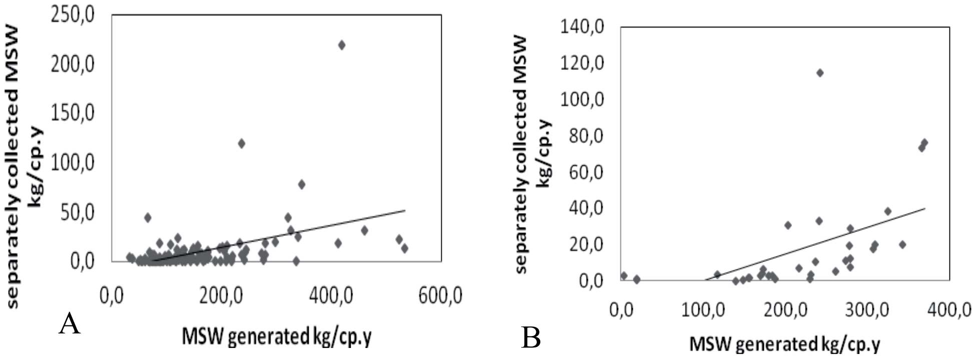

Fig. 6: Results of the linear regression between MSW generated and separately collected waste in rural municipalities class I and II (A) and cities, class III (B).

With regard to the treatment methods used in municipalities, the order of priority was found to be as follows: 1) sorting, 2) recycling, 3) landfilling, 4) mechanical – biological treatment (MBT) (Fig. 7A). Study results on the measures that can be used to raise the sorting ability of MSW in municipalities revealed that raising awareness among people was ranked highly by 43% of respondents (Fig. 7B). Such a high percentage indicates that the cost paid for MSW landfilling is marginal. Awareness can be raised through active information campaigns, the distribution of leaflets, involvement, and the participation of local residents in thematic seminars and education.

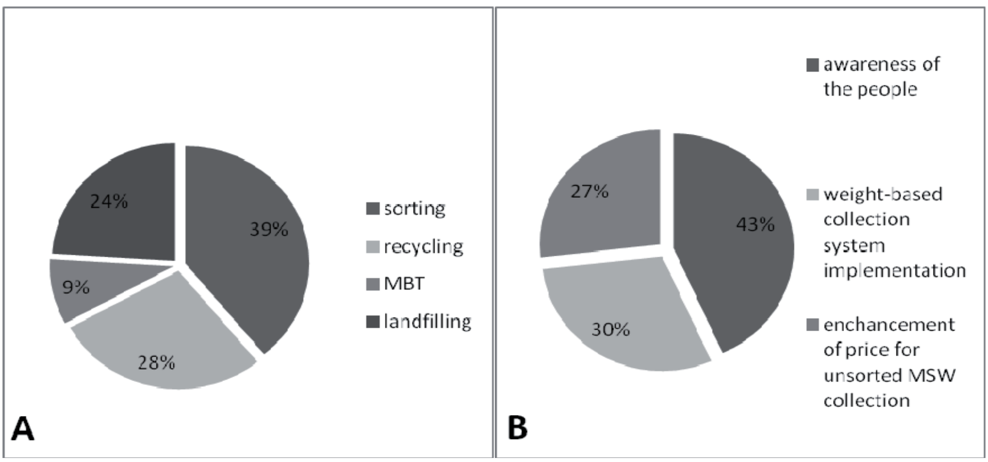

Fig. 7: Results of the study on the treatment methods used in municipalities in Estonia (A) and the options for increasing the sorting rate (B)

The results of the study from the municipalities showed that 50% of the respondent municipalities had not heard of PAYT systems. Other respondents mentioned that it was not yet the appropriate time for the implementation of the PAYT system in Estonia. This can be explained by the significant economic interest in constructing a mass-burn incineration plant, thereby making it unnecessary to enhance the sorting level in municipalities. Some of the respondents who agreed with the necessity of PAYT system implementation noted the importance of explanatory work within the community, the lack of financial resources, and the demand for change in current policies concerning the waste management sector. The main problem in small, sparsely populated municipalities when implementing the PAYT system is the small amounts of waste generated by local residents, particularly packaging waste and glass. Moreover, other types of waste such as paper and cardboard are often incinerated in a furnace and biodegradable waste is usually composted at home. The demand for PAYT system implementation in such areas is very low due to the high implementation costs and low municipal waste management charges in Estonia, if people pay almost the same fee for unsorted waste as they do for separately collected waste.

4.2 The case of Kuusalu rural municipality

New technical installations, additional equipment, and modifications to the collection sites are normally required to carry out weight-based waste collection (Bilitewski et al. 2004).

According to the building register, there are 3,800 immovable units with a dwelling house in Kuusalu rural municipality. Many of the dwellings are summer houses, as Kuusalu rural municipality is a popular recreational area. Implementation of the weight-based waste collection system for private houses in the municipality requires the purchase of new technical equipment. In order to record the weight of all waste, bins need to be labelled with radio-frequency identification tags (RFID). In addition, an investment in RFID equipment for trucks and an IT system for the office is required. Furthermore, on-board weighing systems must be installed, and other costs will arise in connection with the introduction of a new system and pay by weight business model. The staff will need to be trained and customers will need to be informed in various ways regarding the new tariff. The total investment cost for the two trucks that would be required to serve the region was estimated to be € 68,500. The analysis does not include the estimated cost of building fences around containers located by blocks of flats in urban districts, which will significantly increase the cost of the implementation of the system in these districts. Taking into account the income-expenditure of the waste transport service provider, the investments by the company would be earned back within 44 months.

Considering the average waste generation per capita in Kuusalu rural municipality, the total investment of the weight-based measurement system, and the average waste disposal fee for a given 140-litre container, the mixed municipal waste disposal fee would be 0.14 €/kg on the basis of the PAYT weight system (Table 3). The cost of emptying containers would increase by approximately 20–45% based on estimated investments over the coming three years, and assuming that the production of waste would decrease by 20% in the first year and 15% thereafter.

Table 3: Calculation of weight-based system service fee

Size of container, m^3	Price at the moment, €	Weight per container, kg	Investment needed per kg, €	Price now per kg, €	Price per kg for PAYT, €	Price per container for PAYT, €	Price increase, %
0.08	1.60	11.2	0.03	0.14	0.17	1.92	20.04
0.14	2.41	19.6	0.03	0.12	0.15	2.97	23.28
0.24	4.10	33.6	0.03	0.12	0.15	5.06	23.46
0.26	4.24	36.4	0.03	0.12	0.15	5.28	24.58
0.37	6.71	51.8	0.03	0.13	0.16	8.19	22.10
0.60	7.15	84.0	0.03	0.09	0.11	9.55	33.63
0.80	8.14	112.0	0.03	0.07	0.10	11.35	39.39
2.50	23.20	350.0	0.03	0.07	0.09	33.22	43.19
4.50	39.65	630.0	0.03	0.06	0.09	57.69	45.49

4.3 Analysis of factors influencing the success of PAYT implementation in the case of Tallinn city.

On the one hand, Tallinn City Government is responsible for organising MSW management, which includes separated waste collection and providing collection facilities and services for source separated waste, according to the MSW sorting procedures adopted by the Ministry of the Environment in January 2007 (RTL 2007, 9, 140). On the other hand, the municipality does not have the right to choose the place of utilisation of MSW, as it is collected from households by waste handling companies, which then decide on further waste treatment. For instance, in most European cities, the role of the local municipality is much broader with extended rights to make decisions regarding waste management and its powers are not simply limited to organisational issues (Karagiannidis *et al*., 2008, 2006; Reichenbach, 2008; Sauer *et al*., 2008). Such limitation can be viewed as an obstacle for the successful implementation of the PAYT system.

Since 1 June 2010, Tallinn has been divided into 13 waste transport districts (Tallinn City Council, 2008). At the beginning of 2010, Tallinn's waste was transported by six companies (Tallinn City Government, 2010).

If a waste management company violates contractual obligations, the city of Tallinn has the right to collect a contractual penalty or cancel the sole right for transporting the waste. To ensure the compensation of possible damage to the environment or a third party, transporting companies must have a liability insurance contract. To avoid urban air pollution, the Environmental Office also regulates the age and technical conditions of vehicles.

The Waste Act requires that organised municipal waste transport must cover all waste holders: private houses, summer houses, apartments, apartment associations, etc. To enter into a contract, waste transporters in the particular district must contact waste holders. The waste holder can specify the frequency and time for emptying the container, conditions of container rent, payments, additional services and other details concerning waste management.

Waste holders who reject a waste management contract must still pay the minimum service package. For a single family house, assuming that the container holds only slowly degradable waste, such a package means emptying a container with a volume of at least 140 L once a month. More infrequent emptying of the container, which is seen as the exception, requires justification and special agreement with the waste transporting company. Therefore, waste holders may utilise such an exception for a certain period only. A waste management company may accept such an exception if the waste holder is absent from the site of waste origin for a longer period (usually due to the dwelling being temporary or the residents travelling) or when a building is under construction or is otherwise unsuitable for living. After that period, the waste holder must report regarding its waste handling. In contrast to individuals, enterprises may handle their waste themselves.

In Dresden, for example, aside from the services for separate MSW collections, one container per 730 inhabitants was distributed for the drop-off of glass, recovered paper and other recyclable material (Reichenbach, 2010).

Tallinn produced 161,557 tons of MSW in 2009, 70% of which was landfilled and only 30% recycled (Tallinn City Council, 2011). In contrast, Dresden recycled 91% of the 178,826 tons of MSW it produced (Dresdner Amtsblatt, 2012).

4.3.1 Legal framework

The PAYT model can only be implemented under favourable legislation (Sauer *et al.*, 2008). If regulations prevent illegal littering or the random use of waste containers, waste producers are forced to use special containers. Waste transportation companies need legal rights and motivation to bill according to the PAYT model. Proper motivation comes from adequate financial regulations, which either bill waste transportation companies according to the provided waste amounts or reimburse their costs independently from produced amounts, or combine both of these approaches. The legal right of waste transportation companies to bill according to the PAYT model must accompany a legally accepted waste amount measurement system.

Although waste management policy and practice in EU member states has been harmonised to a certain extent, it has had a significant national and local element (Bilitewski *et al*., 2004).

For example, in Dresden, there is a general legal stipulation on the use of charges by municipalities, which has to be respected. Charges must encourage waste minimisation. Charge models may calculate volume according to the disposition of the household, pick-up frequency, number of household members, the amount or volume of waste and the intensity of the service requested as a basis. Waste charges can be levied for the sole specific purpose of waste collection and disposal.

In Tallinn, by contrast, the price of waste management services is fixed in the bidding competition (RT I, 2004, 9 52). Instead of encouraging recycling and waste minimisation, current legislation facilitates price competition between waste transporting companies, who must, in order to win public tenders, bid cheap services. In such situations, many people disregard waste transportation expenditure because it forms only a small fraction (3–5%) of communal service costs. In the case of only part-full containers, people tend to add recyclable waste to them. In contrast, containers in districts with large buildings often fill over. At the same time, transporting companies are interested in transporting higher waste amounts to receive more reimbursement from local governments.

Moreover, degressive pricing motivates citizens to order larger containers and fill them. For instance, while the reference price for emptying a 140 L container is € 5.60, a 240 L container costs € 6.06. Hence, the next 100 L costs only € 0.46. Such a small price difference between containers of different sizes explains well why people still, at least in theory, are not being economically punished for producing larger amounts of waste.

4.3.2 Social circumstances

Population density critically affects waste management. Low population density challenges cost-effective collection and manufacturing (Ventosa, 2008), while a dense population often generates a higher environmental impact, better social control over illegal littering, as well as higher public sensitivity and awareness about the issue.

The population is distributed more or less equally on a larger area such as in Dresden, despite the relatively low average population density of 1,594 inhabitants/km^2, which prevents "deserted" areas nearby and disfavours illegal dumping. The forming of solidarity groups (groups of residents who are given joint access to a chargeable waste service) has led to better social control and reduced the problem of anonymity of the waste generator.

By contrast, in Tallinn, the population density is higher – about 2,619 inhabitants/km^2 – and there are many deserted areas nearby, which become areas for illegal dumping. This problem seriously challenges the adoption of the PAYT system.

4.3.3 Environmental Awareness

Informing the public is crucial for acceptance of and participation in the system (e.g. Sauer *et al*.,2008; Ventosa, 2008). A potentially beneficial system will not become effective if people doubt its necessity. To sufficiently inform citizens, local governments should allocate more resources. Communities that have implemented PAYT programmes in Europe are nearly unanimous in listing education and community relations as the most important elements in successful implementation (Bilitewski, 2004). The information process initiated for this should start early; it should be programmed and continuous. Information processes have been found to be the single effective way to develop a general consensus among residents on the need for a PAYT scheme and community support has proven vital in ensuring the long-term success of these schemes. Achieving transparency among the public obviously helps to combat apprehensions about PAYT, such as concerns regarding increased illegal dumping, the perception that the introduction of PAYT will result in an additional financial burden for residents or their natural resistance to change.

4.3.4 Economical aspects

The marginal rate of MSW handling is established by local municipalities. In Tallinn, centralised waste handling and the marginal rate for waste handling companies for MSW collection from households had been applied since 2007 by Regulation No. 34 "Application of centralised waste handling in Tallinn city". The rate for single emptying of mixed MSW in Tallinn is relatively low when comparing for example to Dresden, (Table 4). In particular, the difference in the price between sorted and unsorted residuals in Tallinn is very small, which does not motivate residents to sort their waste.

Table 4: Marginal rates of collection of MSW per single emptying from households

Size of bin, l	Place	Mixed MSW, €	Paper and cardboard, €	Biodegradable wastes (mainly kitchen wastes), €
240	Tallinn	5.05	2.70	3.80
240	Dresden	13.30	2.50	5.25

Major differences between the two cities emerge when comparing waste management costs with the costs of other services e.g. electricity, water and transport.

In **Dresden**, in 2008, a family paid an average of € 380 for electricity and € 385 for water/ wastewater (Reichenbach, 2010). Waste management services cost an average of 0.11 €/m^2 of apartment, € 58.80 per inhabitant and € 235.20 per family, which consisted of 62% of the electricity cost, 61% of the water/wastewater cost, 72% of the local transport cost, 109% of the TV and radio cost, 65% of the phone and radio cost, and 100% of the daily newspaper cost. Hence, waste management formed a significant proportion of all major costs. Therefore, residents are economically highly motivated to decrease their waste management costs.

In **Tallinn**, waste management services cost annually only 0.02–0.10 €/m^2 of apartment, which forms less than one per cent of family income and only 3–5% of all communal costs. Hence, residents in Tallinn still remain economically little motivated to change the existing waste charging model.

5 Conclusions

The results of the assessment study showed that half of the respondents among local municipalities deemed the effective implementation of the PAYT system as a possibility, through significant explanatory work within the community. The lack of financial resources and demand for change in the policies concerning the waste management sector could hinder this process.

The results of the evaluation of the factors influencing the success of the PAYT system showed that the current situation in Estonia does not facilitate PAYT system implementation in the near future: legislation should create favourable conditions for switching to the weight-based collection system, citizens should be economically motivated to sort their waste, the social aspect in terms of avoiding illegal dumping should be accounted for, and the responsibility and administrative capacity of local authorities should be expanded. By analysing the waste management situation in other Baltic States, it can be assumed that PAYT system implementation will involve similar difficulties.

Economic feasibility plays a crucial role in implementing innovative techniques into the current waste management system. The difference in price between mixed MSW and sorted waste is negligible in Estonia where the citizens remain little motivated economically to change the existing waste charging model.

Our findings confirm the fact that current implementation of the PAYT system in Baltic States is hindered by political, economical and social factors.

The main obstacles are concluded below:

1. The need to make major changes in current waste management legislation and change the financing mechanism of local municipalities in the field of waste management.

2. Changes are necessary in MSW charging mechanisms i.e. switching from flat rate charges to a variable rate pricing system. The difference in price between unsorted MSW and separately collected waste amounts should be increased to motivate people to sort their waste.
3. Awareness of the PAYT system among local authorities is rather low. Therefore, more information has to be provided to local authorities. For instance, successful experiences from other countries should be disseminated.
4. Other factors include the high population density in Tallinn while there are many deserted areas nearby, which facilitate illegal dumping. To improve the situation, better spatial planning can be proposed.
5. No correlation was found between the total amount of waste generated and separately collected waste, which indicates that people in Estonia randomly sort their waste and that they are not economically motivated.
6. The billing mechanisms are not entirely fair at the moment; however, according to the calculations on behalf of one rural municipality, significant investments were required in order to switch to the weight-based waste collection system.

General trends in the waste management sector show that Estonia is currently mass-burn incineration orientated and less motivated to increase its recycling rate. In comparison, in Dresden, a similar situation was observed in the early 1990s when landfill amounts started to be reduced and more attention was paid to increasing recycling materials. The current situation in Dresden shows that they have succeeded with waste management reforms and they now operate a fair and transparent charging mechanism and weight-based collection system. Therefore, significant changes are required to create step-by-step favourable conditions for more efficient and sustainable waste management. More in depth investigation of the options for PAYT system implementation with its practical adoption in a pilot municipality is recommended.

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APPENDIX II CURRICULUM VITAE

1. Personal data

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Date and place of birth	26.01.1980, Russia

2. Contact information

Address	Valge 18-14, Tallinn, Estonia
Phone	+37253484084
E-mail	viktoria.voronova@ttu.ee

3. Education

Educational institution	Graduation year	Education (field of study/degree)
Tallinn University of Technology	2005	master degree - speciality: environmental management and cleaner production
Tallinn University of Technology	2003	bachelor degree - speciality: chemistry and technology of environmental protection
Kohtla-Järve Joint Gymnasium	1998	Secondary education

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

Language	Level
Russian	Native
Estonian	fluent
English	fluent
German	average

5. Special Courses

Period	Educational or other organisation
22-26 October, 2007	Wageningen University, Holland – Scenario development,

	understanding and applying multi-scale and participatory conept and tools.
13-17 Juuni, 2007	University of Tampere – training course for phD students – „Water Governance in Long-Term Perspectives”.
Febr. 2006 – Dets.2006	Hamburg University of Technology, scholarship „Environmental management systems in small and medium sized Enterprises“.

6. Professional Employment

Period	Organisation	Position
2012	Tallinn University of Technology, Department of Environmental Engineering	lecturer
2007-2012	Tallinn University of Technology, Department of Environmental Engineering	researcher
2005-2006	Emedical OÜ	specialist
2002-2004	Tallinn University of Technology, Department of Chemical Engineering	laboratory technician

7. Defended theses

Masterthesis: Catalityc ozonation as a means for degradation of resistant nitroaromatics, Tallinn University of Technology, 2005.
Bachelor: The application of catalysts for intensification of ozonation of m-dinitrobenzene, Tallinn University of Technology, 2003.

8. Main areas of scientific work/Current research topics

Biosciences and Environment, Research relating to the State of the Environment and to Environmental Protection (environmental management systems, life cycle assessment, waste management.)

9. Other research projects

2012 – Sustainable Uses of Baltic Marine Resources (SUBMARNIER), (INTERREG IVA)
2009 – 2011 – Development of pay-as –you-trhow system in Cyprus, Greece and Estonia, HEC-PAYT, (Program Life+ Environmental Policy and Governance)
2008 – 2010 - Research and analyses of various treatment technologies of landfill leachate. II etap
2007-2008 – Research and analyses of various treatment technologies of landfill leachate. Development of suitable leachate cleaning technology in Estonian conditions.

10. Scientific work

Publications and presentations:
Voronova, V., Iital, A. (2013). Coastal reed beds in the Baltic Sea and assessment of the potential for use. International Conference on the utilization of emergent wetland plants Reed as a Renewable Resource, poster presentation, February 14th-16th 2013, Greifswald, Germany.

Voronova, V.; Piirimäe, K.; Virve, M. (2013). Assessment of the applicability of the Pay As You Throw system into current waste management in Estonia. Management of Environmental Quality:
An International Journal, xx - xx. [ilmumas].

Iital, A.; Klõga, M.; Kask, Ü.; Voronova, V.; Cahill, B. (2012). Reed harvesting. A. Schultz-Zehden, M. Matczak (Toim.). Compendium. An Assessment of Innovative and Sustainable Uses of Baltic Marine Resources (103-124). Gdansk-Berlin: Maritime Institute in Gdansk.

Moora, H.; Voronova, V.; Uselyte, R. (2012). 'Incineration of Municipal Solid Waste in the Baltic States: Influencing Factors and Perspectives. Avraam Karagiannidis (Toim.). 'Waste to Energy:
Opportunities and Challenges for Developing and Tsansition Economies (237 - 260). London: Springer-Verlag London Ltd.
Iital, A..; Voronova, V..; Klõga, M. (2011). Development of water scenarios for large lakes in Europe: The case of Lake Peipsi. Journal of Water and Climate Change, 2(2-3), 154 - 165.

Voronova, V.; Moora, H.; Loigu, E. (2011). Environmental assessment and sustainable management options of leachate and landfill gas treatment in Estonian municipal waste

landfills. Management of Environmental Quality: An International Journal, 22(6), 787 - 802.

Moora, H.; Voronova, V.; Reihan, A. (2009). The impact of Municipal Solid Waste Management on Greenhouse Gas Emissions in Estonia. Walter Leal Fihlo (Toim.). Interdisciplinary Aspects of Climate Change (311 - 325). Frankfurt am Main: Peter Lang Publishers House.

Koroljova, A.; Voronova, V. (2007). Eco-Mapping as a Basis for Environmental Management Systems Integration at Small and Medium Enterprises. Management of Environmental Quality: An International Journal, 5, 542 - 555.

Voronova, V. (2006). Problems Related to Sustainable Heritage Conservation in Estonia . In: Conference Proceeding - Citizens and Governance for Sustainable Development: The 4th International Conference, Citizens and Governance for Sustainable Development, Vilnius, Lithuania, 28-30 September 2006. (Toim.) Leal Fihlo, W.; Dzemydiene, D.; Sakalauskas, L.; Zavadskas, E. K.. VGTU Press "Technika"; Mykolas Romeris University, 2006,
353 - 357.

Honours & Awards 2012: Viktoria Voronova; Outstanding paper Award, Emerald Literati Network.

APPENDIX III ELULOOKIRJELDUS

1. Isikuandmed

Ees- ja perekonnanimi	Viktoria Voronova
Sünniaeg ja -koht	26.01.1980, Venemaa
Kodakondsus	Eesti

2. Kontaktandmed

Aadress	Valge 18-14, 11415, Tallinn
Telefon	+37253404084
E-posti aadress	viktoria.voronova@ttu.ee

3. Hariduskäik

Õppeasutus (nimetus lõpetamise ajal)	Lõpetamise aeg	Haridus (eriala/kraad)
Tallinna Tehnikaülikool	2005	Magistrikraad – eriala keskkonnajuhtimine ja puhtam tootmine
Tallinna Tehnikaülikool	2003	Bakalaureusekraad – eriala keemia ja keskkonna-kaitsetehnoloogia
Kohtla-Järve Ühisgümnaasium	1998	keskharidus

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

Keel	Tase
Vene keel	Emakeel
Eesti keel	Kõrgtase
Inglise keel	Kõrgtase
Saksa keel	Kesktase

5. Täiendusõpe

Õppimise aeg	Täiendusõppe läbiviija nimetus
22-26 oktoober, 2007	Wageningen University, Holland – Scenario development,

	understanding and applying multi-scale and participatory conept and tools
13-17 juuni, 2007	University of Tampere - interdistsiplinaarne uurimuslik koolitus doktorantidele – „Water Governance in Long-Term Perspectives”
Veebruar 2006 – detsember 2006	Hamburg University of Technology, täiendkoolitus keskkonnajuhtimissüsteemid väikestes ja keskmistes ettevõtetes Saksamaal;

6. Teenistuskäik

Töötamise aeg	Tööandja nimetus	Ametikoht
2012	Tallinna Tehnikaülikool, KeskkonnatehnikaInstituut	lektor
2007-2012	Tallinna Tehnikaülikool, KeskkonnatehnikaInstituut	teadur
2005-2006	Emedical OÜ	arendusspetsialist
2002-2004	Tallinna Tehnikaülikool Keemia Instituut	laborant

7. Kaitstud lõputööd

Magistritöö: Katalüütilise osoonimise kasutamine aromaatsete nitroühendite lagundamiseks, Tallinna Tehnikaülikool, 2005.
Bakalaureusetöö: Katalüsaatorite kasutamine m-dinitro-benseeni osoonimise intensiivistamiseks, Tallinna Tehnikaülikool, 2003.

8. Teadustöö põhisuunad

Keskkonnaseisundit ja keskkonnakaitset hõlmavad uuringud (keskkonnajuhtimis-süsteemid, olelusringi hindamine, jäätmemajandus);

9. Uurimisprojektid

2012 – Sustainable Uses of Baltic Marine Resources (SUBMARNIER), (INTERREG IVA)

2009 – 2011 - Täpsema jäätmete maksustamise süsteemi väljaarendamine Kreekas, Eestis ja Küprosel HEC-PAYT, (Program Life+ Environmental Policy and Governance)
2008 – 2010 - Prügilavee uuringud ja erinevate puhastustehnoloogiate analüüs. Eesti oludesse sobiva puhastustehnoloogia väljatöötamine - II etapp
2007-2008 - Prügilavee uuringud ja erinevate puhastustehnoloogiate analüüs. Eesti oludesse sobiva puhastustehnoloogia väljatöötamine.

10. Teadustegevus

Publikatsioonid ja ettekanded:

Voronova, V., Iital, A. (2013). Coastal reed beds in the Baltic Sea and assessment of the potential for use. International Conference on the utilization of emergent wetland plants Reed as a Renewable Resource, poster presentation, February 14th-16th 2013, Greifswald, Germany.

Voronova, V.; Piirimäe, K.; Virve, M. (2013). Assessment of the applicability of the Pay As You Throw system into current waste management in Estonia. Management of Environmental Quality:
An International Journal, xx - xx. [ilmumas].

Iital, A.; Klõga, M.; Kask, Ü.; Voronova, V.; Cahill, B. (2012). Reed harvesting. A. Schultz-Zehden, M. Matczak (Toim.). Compendium. An Assessment of Innovative and Sustainable Uses of Baltic Marine Resources (103-124). Gdansk-Berlin: Maritime Institute in Gdansk.

Moora, H.; Voronova, V.; Uselyte, R. (2012). 'Incineration of Municipal Solid Waste in the Baltic States: Influencing Factors and Perspectives. Avraam Karagiannidis (Toim.). 'Waste to Energy:
Opportunities and Challenges for Developing and Tsansition Economies (237 - 260). London: Springer-Verlag London Ltd.
Iital, A..; Voronova, V..; Klõga, M. (2011). Development of water scenarios for large lakes in Europe: The case of Lake Peipsi. Journal of Water and Climate Change, 2(2-3), 154 - 165.

Voronova, V.; Moora, H.; Loigu, E. (2011). Environmental assessment and sustainable management options of leachate and landfill gas treatment in Estonian municipal waste landfills. Management of Environmental Quality: An International Journal, 22(6), 787 - 802.

Moora, H.; Voronova, V.; Reihan, A. (2009). The impact of Municipal Solid Waste Management on Greenhouse Gas Emissions in Estonia. Walter Leal Fihlo (Toim.). Interdisciplinary Aspects of Climate Change (311 - 325). Frankfurt am Main: Peter Lang Publishers House.
Koroljova, A.; Voronova, V. (2007). Eco-Mapping as a Basis for Environmental Management Systems Integration at Small and Medium Enterprises. Management of Environmental Quality: An International Journal, 5, 542 - 555.

Voronova, V. (2006). Problems Related to Sustainable Heritage Conservation in Estonia . In: Conference Proceeding - Citizens and Governance for Sustainable Development: The 4th International Conference, Citizens and Governance for Sustainable Development, Vilnius, Lithuania, 28-30 September 2006. (Toim.) Leal Fihlo, W.; Dzemydiene, D.; Sakalauskas, L.; Zavadskas, E. K.. VGTU Press "Technika"; Mykolas Romeris University, 2006,
353 - 357.

Teaduspreemiad ja- tunnustused 2012: Viktoria Voronova; Silmapaistva artikli auhind, Emerald LiteratiNetwork

DISSERTATIONS DEFENDED AT TALLINN UNIVERSITY OF TECHNOLOGY ON *CIVIL ENGINEERING*

1. **Heino Mölder**. Cycle of Investigations to Improve the Efficiency and Reliability of Activated Sludge Process in Sewage Treatment Plants. 1992.

2. **Stellian Grabko**. Structure and Properties of Oil-Shale Portland Cement Concrete. 1993.

3. **Kent Arvidsson**. Analysis of Interacting Systems of Shear Walls, Coupled Shear Walls and Frames in Multi-Storey Buildings. 1996.

4. **Andrus Aavik**. Methodical Basis for the Evaluation of Pavement Structural Strength in Estonian Pavement Management System (EPMS). 2003.

5. **Priit Vilba**. Unstiffened Welded Thin-Walled Metal Girder under Uniform Loading. 2003.

6. **Irene Lill.** Evaluation of Labour Management Strategies in Construction. 2004.

7. **Juhan Idnurm**. Discrete Analysis of Cable-Supported Bridges. 2004.

8. **Arvo Iital**. Monitoring of Surface Water Quality in Small Agricultural Watersheds. Methodology and Optimization of monitoring Network. 2005.

9. **Liis Sipelgas**. Application of Satellite Data for Monitoring the Marine Environment. 2006.

10. **Ott Koppel**. Infrastruktuuri arvestus vertikaalselt integreeritud raudtee-ettevõtja korral: hinnakujunduse aspekt (Eesti peamise raudtee-ettevõtja näitel). 2006.

11. **Targo Kalamees**. Hygrothermal Criteria for Design and Simulation of Buildings. 2006.

12. **Raido Puust**. Probabilistic Leak Detection in Pipe Networks Using the SCEM-UA Algorithm. 2007.

13. **Sergei Zub**. Combined Treatment of Sulfate-Rich Molasses Wastewater from Yeast Industry. Technology Optimization. 2007.

14. **Alvina Reihan**. Analysis of Long-Term River Runoff Trends and Climate Change Impact on Water Resources in Estonia. 2008.

15. **Ain Valdmann**. On the Coastal Zone Management of the City of Tallinn under Natural and Anthropogenic Pressure. 2008.

16. **Ira Didenkulova**. Long Wave Dynamics in the Coastal Zone. 2008.

17. **Alvar Toode**. DHW Consumption, Consumption Profiles and Their Influence on Dimensioning of a District Heating Network. 2008.

18. **Annely Kuu**. Biological Diversity of Agricultural Soils in Estonia. 2008.

19. **Andres Tolli**. Hiina konteinerveod läbi Eesti Venemaale ja Hiinasse tagasisaadetavate tühjade konteinerite arvu vähendamise võimalused. 2008.

20. **Heiki Onton**. Investigation of the Causes of Deterioration of Old Reinforced Concrete Constructions and Possibilities of Their Restoration. 2008.

21. **Harri Moora**. Life Cycle Assessment as a Decision Support Tool for System optimisation – the Case of Waste Management in Estonia. 2009.

22. **Andres Kask**. Lithohydrodynamic Processes in the Tallinn Bay Area. 2009.

23. **Loreta Kelpšaitė**. Changing Properties of Wind Waves and Vessel Wakes on the Eastern Coast of the Baltic Sea. 2009.

24. **Dmitry Kurennoy.** Analysis of the Properties of Fast Ferry Wakes in the Context of Coastal Management. 2009.

25. **Egon Kivi**. Structural Behavior of Cable-Stayed Suspension Bridge Structure. 2009.

26. **Madis Ratassepp**. Wave Scattering at Discontinuities in Plates and Pipes. 2010.

27. **Tiia Pedusaar**. Management of Lake Ülemiste, a Drinking Water Reservoir. 2010.

28. **Karin Pachel**. Water Resources, Sustainable Use and Integrated Management in Estonia. 2010.

29. **Andrus Räämet**. Spatio-Temporal Variability of the Baltic Sea Wave Fields. 2010.

30. **Alar Just**. Structural Fire Design of Timber Frame Assemblies Insulated by Glass Wool and Covered by Gypsum Plasterboards. 2010.

31. **Toomas Liiv**. Experimental Analysis of Boundary Layer Dynamics in Plunging Breaking Wave. 2011.

32. **Martti Kiisa**. Discrete Analysis of Single-Pylon Suspension Bridges. 2011.

33. **Ivar Annus**. Development of Accelerating Pipe Flow Starting from Rest. 2011.

34. **Emlyn D. Q. Witt**. Risk Transfer and Construction Project Delivery Efficiency – Implications for Public Private Partnerships. 2012.

35. **Oxana Kurkina**. Nonlinear Dynamics of Internal Gravity Waves in Shallow Seas. 2012.

36. **Allan Hani**. Investigation of Energy Efficiency in Buildings and HVAC Systems. 2012.

37. **Tiina Hain**. Characteristics of Portland Cements for Sulfate and Weather Resistant Concrete. 2012.

38. **Dmitri Loginov**. Autonomous Design Systems (ADS) in HVAC Field. Synergetics-Based Approach. 2012.

39. **Kati Kõrbe Kaare**. Performance Measurement for the Road Network: Conceptual Approach and Technologies for Estonia. 2013.