SPEED OF TRANSITION OF MUNICIPAL SOLID WASTE FROM LANDFILL TO WASTE-TO-ENERGY

Lappeenranta–Lahti University of Technology LUT

Master’s Programme in Sustainability Science and Solutions, Master's thesis

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ABSTRACT

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In recent years, much effort has been invested in the development of waste-to-energy (WTE) technologies that might reduce the volume of waste and ameliorate its negative environmental consequences. This study examines the countries which have the fastest transition from landfilling to WTE treatment and the impact of this transition on the environment during their transition period. The study evaluates the speed of the transition by calculating the annual percentage of WTE treatment and decrease of landfilling and assesses the environmental impact in terms of greenhouse gas (GHG) emissions and dioxin emissions. The results showed that the annual rate of WTE progress based on the share of WTE of the total generation (and treatment) of municipal solid waste (MSW) was in the range of 0,53 %-points/a, and 5,26 %-points/a, also the annual rate of landfill reduction was in the range of 0,63 %-points/a, and 6,63 %-points/a. Furthermore, the results in the environmental impact showed that the changes of GHG emission caused by each kt of treated MSW by WTE and landfill methods were in the range of -0,924 kt $_{CO2eq}$/kt $_{(landfill +WTE)}$ and 0,421 kt $_{CO2eq}$/kt $_{(landfill +WTE)}$. Furthermore, the GHG emission factor for WTE plants was determined for the countries over the period. Finally, the dioxin emission changes were roughly estimated for the case countries, and the dioxin emission changes were decreasing and between 3,14 mg-TEQ/kt $_{landfill+WTE}$ and 7,61 mg-TEQ/kt $_{landfill+WTE}$. The study concludes that WTE is a quick and effective method to reduce environmental impact and generate energy and can be a valuable opportunity for other countries to improve their waste management systems.

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Abbreviations

DOC	Degradable Organic Carbon
EU	European Union
EUR	Euro
GHG	Greenhouse Gas
IEA	International Energy Agency
kt	Kilo Ton
LFG	Landfill Gas
MCF	Methane Correction Factor
MSW	Municipal Solid Waste
MSWI	Municipal Solid Waste Incineration
Mt	Million Ton
OX	Oxidation Factor
SRF	Solid Recovered Fuel
TEQ	Emissions Toxicity of Dioxins
TWh	Terawatt Hour
UK	United Kingdom
USA	United states of America
WPP	Waste Prevention Program
WTE	Waste To Energy
DEI	Dioxin Emission from Incineration
DEL	Dioxin Emission from Landfill fire

EMT	Total Amount of Dioxin Emission per kt of Landfill and WTE
OCG	Overall Changes in GHG emissions
RGE	Rate of GHG emissions
SLC	Speed of Landfill Changes
SLT	Speed of Landfill Transition
SWC	Speed of WTE Changes
SWT	Speed of WTE Transition

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1. Introduction

Municipal solid waste (MSW) is described in the EU's Landfill Directive 1999/31 as "waste from households, as well as other waste which, because of its nature or composition, is similar to waste from households". Also, according to Directive 2018/851, municipal waste means "mixed waste and separately collected waste from households, including paper and cardboard, glass, metals, plastics, bio-waste, wood, textiles, packaging, waste electrical and electronic equipment, waste batteries and accumulators, and bulky waste, including mattresses and furniture" (European Commission, 2022). It should be considered that MSW is mostly produced by households, however, it is also generated by commercial and institutional establishments such as businesses and government buildings. MSW is collected by or on behalf of municipalities and disposed of through waste management facilities (OECD, 2022).

According to the world bank trends in solid waste management (2016), Every year, humans produce 2,01 billion tonnes of MSW, with at least 33 % of that amount not being properly and safety treated. Waste produced daily per person ranges from 0,11 to 4,54 kilograms globally, with an average of 0,74 kilos. High-income countries produce 34 % of the world's waste (683 million tonne (Mt), although having just 16 % of the world's population. By 2050, it is predicted that worldwide waste would have increased to 3,40 billion tonnes, or more than double the growth in population during the same time.

In many countries around the world, MSW is mostly disposed of by landfilling. The high cost of treatment and disposal of MSW is a major reason for the use of landfills, especially in developing countries (Brunner and Fellner, 2007). Landfill has been used more so that roughly 37 % of waste is dumped in a landfill, with just 8 % going to sanitary landfills that also include landfill gas collecting systems. About 31 % of waste is thrown away in open dumps, 19 % is recovered through recycling and composting, and 11 % is incinerated (World bank, 2016).

It is possible to count landfill gas (LFG), which is created as a by-product of activities involving the treatment of MSW, as a source of GHG, most notably methane (Tozlu et al.,

2016). Landfills are responsible for the production of several by-products, including carbon dioxide, water vapor, and trace quantities of oxygen, nitrogen, hydrogen, and organic molecules that are not methane. If these gases are allowed to build up uncontrolled, they can not only contribute to climate change but also produce significant air pollution (Kayla Vasarhelyi, 2021). Additionally, the burning of waste from households, whether on purpose or by accident, in open fires at municipal or illegal landfills is the most significant contributor to the production of dioxins. These kinds of fires happen rather regularly, particularly during the dry and hot summer months (Vassiliadou et al., 2009).

Recycling and waste to energy (WTE) are two viable options that can serve as suitable replacements for landfills, which are known to have less adverse impact than landfilling on the surrounding ecosystem. Despite the fact that both recycling and WTE can have negative effects on the environment, these methods have the potential to protect, preserve, and improve the quality of the environment and human health, ensure the prudent and rational use of natural resources; promote a more circular economy; increase resource efficiency; and ensure valuable waste as a resource (Malinauskaite et al., 2017).

The process of recycling begins with the separation of waste and recyclables at homes and continues all the way to recycling industries. The recyclables are processed into secondary materials, when returned to the production chain. Recycling has some advantages such as: help to save energy, cut emissions, reduce the extraction of natural resources, create jobs and income necessary for the social inclusion of low-income families, and avoid the waste of natural resources (Lino et al., 2010; Lino & Ismail, 2013).

Lino and Ismail (2015), Jones (2010), and others defend the concept of thermal decomposition of MSW for energy production and recycling to ensure the long-term sustainability of the planet, given that the problem of the century is to reduce GHG emissions by transitioning to lower-carbon fuel sources and implementing sustainable technologies. Recycling and WTE schemes are complementary waste treatment strategies that redirect waste from landfills and minimize GHG emissions. WTE systems also contribute to recycling standards by consuming wastes that are inappropriate for recycling (Lino & Ismail, 2017; M Lino et al., 2015). It should be considered that good results have been achieved in both Asia and Europe through the utilization of thermal treatment of MSW in conjunction with energy generation and recycling (Defra, 2013; Karagiannidis, 2012; Lino & Ismail, 2011; Preface | Environment at a Glance 2013).

The current global situation is marked by a steady rise in population growth, energy generation, and consumption, which has resulted in a corresponding increase in GHG emissions. In response, considering MSW as a supplementary source of energy has been proposed as a potential solution. Furthermore, implementing WTE technology is suggested as an effective means of converting MSW into energy. By doing so, it is possible to address the challenge of waste disposal and to help satisfy the growing demand for energy. Therefore, the utilization of WTE technology is seen as a promising pathway for sustainable development and for mitigating environmental concerns (Lino & Ismail, 2017).

MSW incineration (MSWI) is crucial not just for handling this growing volume of waste, but also for energy recovery that may complement the existing supply (Cucchiella et al., 2017). More than 80 percent of the world's primary energy is generated from fossil fuels, meanwhile, WTE can play a vital role in balancing reliance on fossil fuels and boosting the proportion of renewable energy while also contributing to waste treatment. Globally, there are around 1179 WTE facilities with an installed input of over 700,000 metric tonnes per day (Makarichi et al., 2018; Lu et al., 2017).

The European Union (EU) has set a goal for itself to become the first climate-neutral continent by the year 2050. Concerns about climate change have led to an increase in efforts made by EU countries to mitigate the effects of climate change (Monni, 2012; EU Climate Policy - Ministry of the Environment, 2021). In addition, EU rules encourage the recovery of materials and energy from waste and the creation of energy from renewable sources. In the energy system, the substitution of an additional energy production plant with a WTE reduces overall emissions (Monni, 2012). Multiple studies have demonstrated that transitioning from landfilling to WTE reduces total GHG emissions and hazardous pollutants such as dioxins (Luckow et al., 2010; Chandel et al., 2014; S. T. Tan et al., 2015; Y. Wang et al., 2017a).

Because of these reasons, the transition from landfill to WTE can have a significant impact to mitigate the climate crisis. Landfills generate the highest level of toxic emissions because the majority of MSW goes to the landfill as earlier mentioned. To control the rate of climate change and the emissions of toxic gases from the landfill, it is required to have a transition from landfill to recycling and WTE in the future to mitigate substantially climate change and the factors which have effects on it. Furthermore, the transition from landfill to WTE systems constitutes a significant measure in advancing towards a sustainable future. This move has

the potential to mitigate our ecological footprint and decrease our reliance on finite resources. A comprehensive assessment of the speed of transitioning from landfill to WTE is imperative to gauge the level of progress achieved in pursuing these objectives. This analysis can enable the identification of specific areas that require further attention and improvement, with the ultimate goal of establishing a waste management system that is both sustainable and efficient.

Although according to the statistics, the amount of landfill and WTE changes is known, until now, the speed of transition and the factors affecting it have not been evaluated. Therefore, the focus of this thesis is on the investigating speed of this transition as well as the factors influencing it. Although some previous studies have involved the analyses of environmental impacts in some countries, there are lacking studies focused on speed of transition from landfill to WTE and the reasons for it. Furthermore, this study aims to investigate the estimated environmental impact of transition from landfill to WTE and evaluate the changes in GHG emissions and dioxin emissions during the transition period in the countries that have significant changes during this process. According to which the main research questions that will be analysed in the study include:

I. What has the speed of transition been and what have been the reasons for the transition?
II. What is the estimated environmental impact of the transition (GHG emissions, dioxins)?
III. How fast are the changes in GHG emissions and dioxin emissions during the transition period?

2. State of the art

2.1. Speed of transition research

As an alternative to traditional methods of waste management, the incineration and burning of solid waste has become increasingly popular in recent years. To reduce waste and dispose of it efficiently, the authorities responsible for regulating this process are considered this option as a potential strategy (Kollikkathara et al., 2009; Richter & Johnke, 2004).

Li et al (2015) stated in their research the WTE is gradually replacing landfills as the primary waste disposal option due to the large amount of land occupied by landfills. According to their analysis, During the "Thirteenth Five-Year Plan," China's rate of MSW incineration would continue to rise. Also, the expansion of waste incineration was anticipated to increase in the central and western regions of China. In the current market for WTE plant construction, China is one of the major players. Furthermore, between 2011 and 2018, the number of WTE plants increased from 109 to 311 in China (Schuler et al., 2012). World Bank (2015) reported that China became the largest generator of MSW in 2004 after surpassing the United States. As a result, many of its cities are experiencing an unprecedented trash crisis. The lack of land to continue landfilling MSW has led China to develop WTE facilities (The Current Worldwide WTE Trend | MSW Management, 2019). Hence, WTE capacity has steadily increased from 14 Mt in 2007 to nearly 102 Mt by 2018 (National Bureau of Statistics of China, 2022).

During the 1990s, the WTE industry in the USA saw significant setbacks, which resulted in the construction of no new WTE facilities between 1995 and 2014. Due to the expiry of financial incentives, strong public resistance to facility siting, and the Supreme Court's decision in Carbone regarding MSW flow control, several communities in the USA have chosen to move their MSW to less costly regional landfills. Recently, the Supreme Court decided on flow control, reinstating towns' power to establish flow control legislation and route their MSW to waste treatment facilities. In response to these reasons, several WTE plants have lately extended their operations by adding additional processing lines to their current operations (The Current Worldwide WTE Trend | MSW Management, 2019). Furthermore, Mukherjee et al. (2020) assessed current MSW WTE trends in the USA,

highlighting regional differences in technology adoption and distinct difficulties for each technology. They reported that just 13 % of MSW is utilized for energy recovery in the USA, whereas 53 % is landfilled.

In Italy, in 2020 about 6,2 Mt of MSW were treated in plants with energy recovery. Also, the trend of waste treatment is shown that in the period 2011-2020, the energy recovery, was carried out by the incineration plants that mainly treat urban waste. In particular, it is noted that the quantity of electricity produced in the period 2011-2020 shows a progressive increase, going from 4 TWh in 2011 to over 4,5 TWh in 2020. Thermal energy, produced exclusively by plants located in the North, it rose from almost 1,3 TWh in 2011 to over 2,3 TWh in 2020. Hence, Italy has invested in the development of WTE because it needs this power plant for its energy supply (Dedichiamo Questo Rapporto a Fabrizio Che Ci Guarda Dalla Vetta Più Alta, 2021).

Additionally, Malinauskaite et al. (2017) stated in their research that the WTE sector in Poland is currently experiencing rapid growth. It is planned to construct several modern MSW incineration plants in the near future. In the most recent national waste management plan, it is recommended that the number of newly developed facilities be reevaluated so as not to overstate the processing capacity.

2.2. Speed of transition in different countries (based on statistics)

Recently, most waste streams in European countries have been diverted from landfills, especially household waste. In accordance with the landfill directive, the fraction of MSW disposed of through landfilling shall be decreased to no more than 10 % of the total mass of MSW generated by 2035. A reduction of 21 %-points in the proportion of MSW that entered landfills was achieved by 2017, and 11 of 37 European countries were able to reduce their MSW landfilling rates by over 40 % and 10 had disposed of less than 10 % of their MSW; however, 15 countries still disposed of more than 50 % of their MSW (European Environment Agency, 2021). In addition, WTE contributes to achieving the EU's 2030 targets for reducing GHG emissions (minimum 40 % decrease from 1990 levels) and for increasing renewable energy sources (minimum 32 % share) (Paul de bruycker and Ella

stengler, 2019).

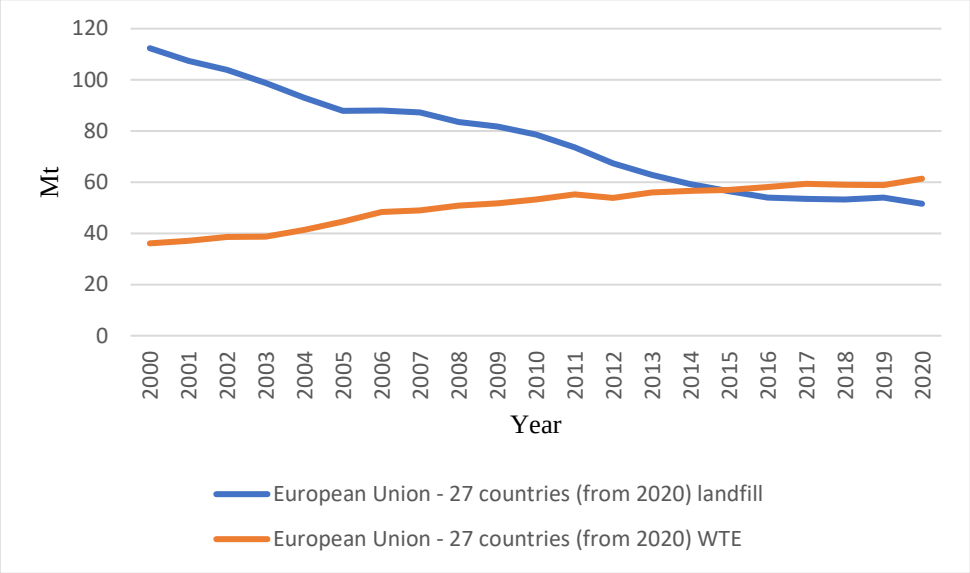

Figure 1. Comparison of MSW Landfill and WTE in EU from 2000 to 2020 Data from (Eurostat - Data Explorer, 2022)

A comparison of the trend of changes in WTE and landfills in Figure 1 demonstrates that over the last two decades, the EU has been able to reduce landfill volume significantly while increasing WTE utilization and progressing towards recycling and recovery. Trends in waste management have also transitioned. It has been reported that the rate of MSW disposal in landfills has declined by 43 % between 2008 and 2017, while energy recovery from MSW has risen by 72,1 %, material recycling has increased by 22,5 %, and composting and digesting has increased by 18 % (European Environment Agency, 2021). However, the trend for the USA as shown in **Figure *2*** has increased for both landfill and WTE from 1960 to 2018.

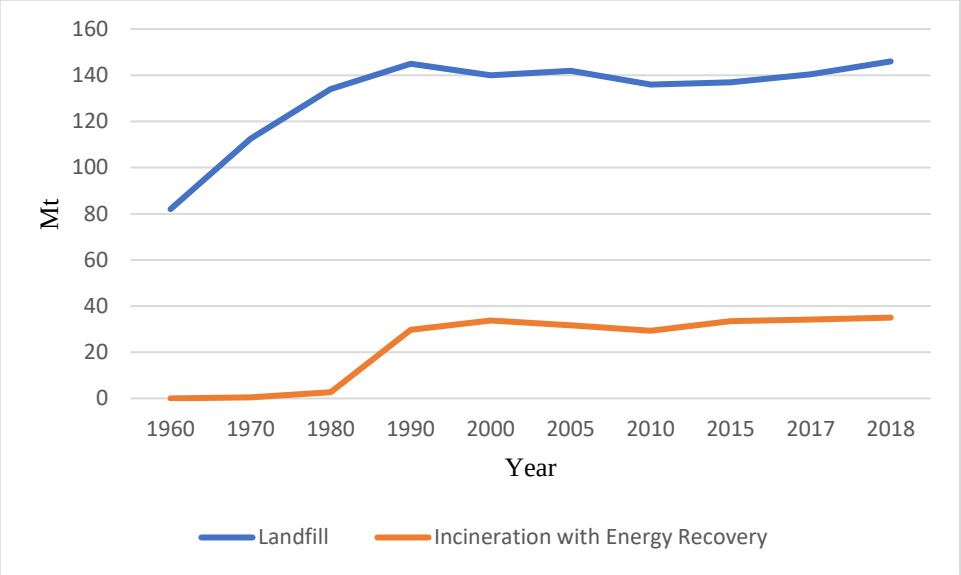

Figure 2. Comparison of MSW Landfill and WTE in the U.S. from 1960 to 2018 (US EPA, 2018)

In the last two decades, the European Union has made great strides in reducing the amount of MSW in landfills and progressing towards recycling and recovery. Trends in waste management have also transitioned. It has been reported that the rate of MSW disposal in landfills has declined by 43 % between 2008 and 2017, while energy recovery from MSW has risen by 72,1 %, material recycling has increased by 22,5 %, and composting and digesting has increased by 18 % (European Environment Agency, 2021). However, the trend for the U.S. has been grown for both, landfill and WTE from 1960 to 2018.

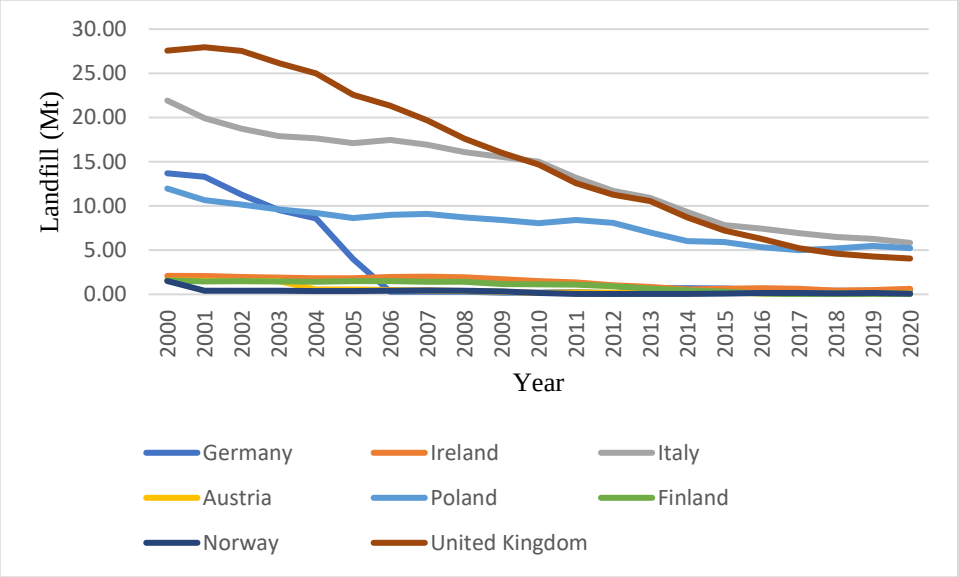

Figure 3. Landfill trend in eight European countries

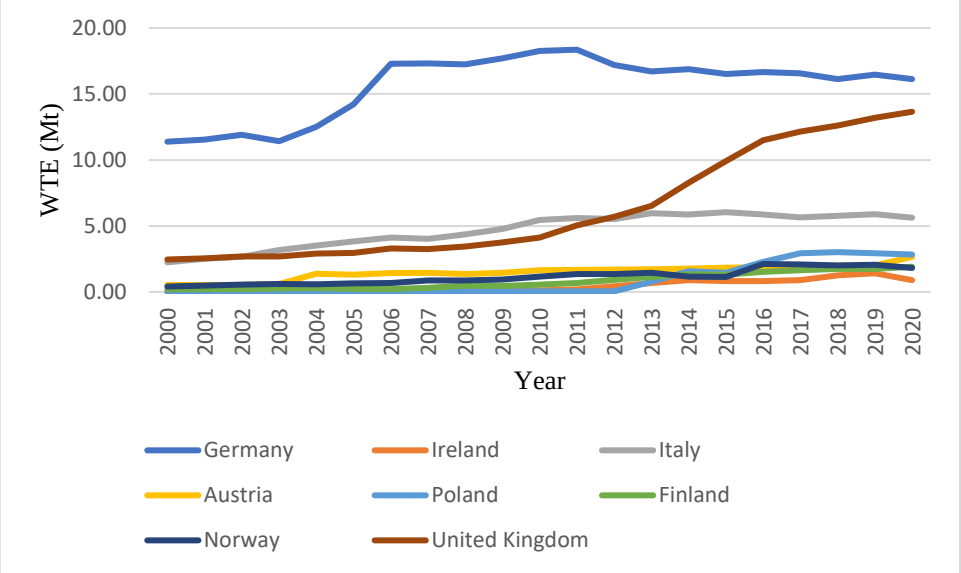

Figure 4. WTE trend in eight European countries

In light of Europe's commitment to climate mitigation, the examination of individual European countries becomes crucial. To this end, this study shows the trends in landfill and WTE changes in European countries in Figures **Error! Reference source not found.** and**Error! Reference source not found.**. The transition towards climate mitigation in Europe has prompted policymakers to take a more comprehensive approach to waste

management practices (Alshuwaikhat & Abubakar, 2008). As a result, the trends in landfill and WTE provide valuable insights into the progress of these practices over time. The data presented in Figures **Error! Reference source not found.** and**Error! Reference source not found.** offer a detailed overview of the changes in landfill and WTE trends in European countries, thus contributing to a better understanding of the effectiveness of waste management strategies in the context of climate change mitigation.

2.3. Environmental impact

2.3.1. GHG emissions from landfill and WTE

By utilizing waste products for energy production, rather than sending them to landfills, it is possible to avoid the emission of methane gas (Chaliki et al., 2016). Methane is a significant contributor to climate change as it is the second most crucial GHG, following carbon dioxide. Its impact on global warming is greater than that of carbon dioxide, with a global warming potential of 28 times on a 100-year timescale and 84 times on a 20-year timescale. Given this, methane emissions are of utmost importance when it comes to meeting the climate objectives set for 2050 (European commission, 2022).

Based on functioning facilities in the USA, Psomopoulos et al. (2009) focused on WTE, particularly its condition and the impact of transitioning from landfilling to WTE plants in terms of GHG, dioxin, and mercury emissions reduction, energy generation, and land conservation. The results of the research indicate that the WTE facilities produce much lower levels of GHG than electricity production facilities using fossil fuels (except natural gas). As a result, the GHG emissions from landfills will be reduced further, while fossil fuel dependence for power generation will be decreased as well. Also, Aracil et al. (2018); Bosmans et al. (2013); Cole-Hunter et al. (2020); Wang et al. (2017) and others defended these results in their studies.

According to Assamoi & Lawryshyn (2012), their investigation included a comparison of two scenarios on how to deal with the residual waste after diversion in Toronto. Initially, the

residual waste is landfilled completely in the first scenario, while in the second scenario, approximately 50 % of the residual waste is incinerated and the rest is landfilled. Based on the displacement of power plant emissions, the scenario incorporating incineration performed better environmentally and contributed significantly to a reduction in GHG emissions. While, based on the studies conducted by M. Faizal et al. (2018), the potential carbon avoidance from incineration in Malaysia is estimated at 22 500 tons CO_2/day, which is a higher claim than LFG recovery at 632 tons CO_2/day.

Overall, the aforementioned research demonstrates that WTE is an environmentally preferable alternative to landfills. In WTE, waste is regarded as a useful substance that returns to the consumption cycle as energy. Table 1 compares between 1990 and 2018 the quantity of CO_2 emissions from landfills and WTE in Europe. It is evident that an increase in the utilization of WTE technology during the specified period led to a decrease in the overall amount of GHG emissions stemming from both fossil fuel sources and landfills (Table 1).

Table 1. Comparison of CO_2-eq emissions from landfill and WTE (Janek Vahk, 2020)

Data	Data Area	CO_2 from WTE (kt) (Biogenic sources)	CO_2 from WTE (kt) (Fossil Fuel Sources)	Total CO_2 WTE (kt)	Total CH_4 from Landfill (kt) (CO_2eq)	Total CO_2 (kt) (Fossil Fuel Sources and Landfill)
1990	EU27+UK	2 600	2 700	5 500	189 500	192 200
2018	EU27+UK	43 500	52 100	95 500	99 500	151 600

In a study conducted by Beylot & Villeneuve (2013), the environmental impact of residual MSW incineration in France was investigated using a life cycle approach. The authors discovered that incinerating one ton of MSW had potential implications on climate change ranging from a benefit of -58 kg CO2eq to a burden of 408 kg CO2eq, with an average impact of 294 kgCO2eq.

Guo et al. (2018) evaluated the ecological impacts of urban solid waste incineration within a traditional Chinese business park. The results of their analysis indicated that the in-use reserves of MSW incineration provided positive environmental benefits in terms of climate impact, human pollution, and acidification. However, they also observed a negative impact on eutrophication potential. Furthermore, Nabavi-Pelesaraei et al. (2017) performed an

environmental life cycle assessment to evaluate the incineration of MSW in Iran. The study revealed that incineration was able to mitigate the negative effects related to toxicity through the production of electricity and phosphate fertilizers.

2.3.2. Dioxin emissions from landfill fire and WTE

The term "dioxins" refers to a group of similar chlorinated organic compounds known as polychlorinated dibenzo-dioxins (PCDDs) and polychlorinated dibenzo-furans (PCDFs). The difference between them is the number and arrangement of chlorine atoms in their chemical structure. These chemicals are generated as a result of activities such as burning fuel, incinerating waste, burning of landfill sites, making pesticides, and bleaching pulp and paper using chlorine. They can also be produced by natural events such as forest fires and volcanic eruptions. (Cheng & Hu, 2010a).

Dioxins are typically produced in a variety of combustion processes, including landfill fire and waste incineration (Kulkarni et al., 2008; Ruokojärvi et al., 1995). Until last decades, incinerators have traditionally been identified as one of the most significant sources of toxic emissions, including not only dioxins but also heavy metals (Kim et al., 2008; Shibamoto et al., 2007; J. B. Wang et al., 2009). Among the harmful substances emitted by waste incinerators and landfill fires, dioxins have generated a substantial amount of controversy since they are among the most toxic compounds in the environment (Ruokojärvi et al., 1995; Schuhmacher & Domingo, 2006).

World Health Organization (WHO) (2016) stated that humans exposed to high quantities of dioxins for a short period of time may develop skin lesions, such as chloracne and patchy darkening of the skin, and impaired liver function. Long-term exposure is associated with impaired immunological, neurological, endocrine, and reproductive systems.

Numerous in-depth investigations have been conducted on a range of dioxin emission sources, which include waste combustion sources, chemical-industrial sources, and other thermal sources (Mari & Domingo, 2010)

Due to the harmful effects of dioxins emissions, the mass of dioxins should be controlled in MSW incineration. According to Dwyer & Themelis (2015) research, "dioxins" can be classified into two categories based on their sources: controlled industrial sources and open burning sources including landfill fires. An estimation has been made by Dwyer & Themelis (2015) to determine the amount of dioxin emission in 1987 and 2012 in the USA. The total emissions from controlled sources reduced by 95,5 % in 2012, from 14 kg TEQ in 1987 to 0,6 kg TEQ in 2012. Furthermore, the emissions of open-burning sources raised up from 2,3 kg TEQ in 1987 to 3 kg TEQ in 2012.

Furthermore, Wei et al. (2022) indicate that China does not have a current regional inventory of dioxin emissions from MSWI, despite the fact that MSWI has grown rapidly in recent decades. They compiled a dioxin emissions inventory based on information collected from the official government and enterprise websites in mainland China, resulting in MSWI releasing 22,56 g-TEQ of dioxins in 2020.

An investigation conducted by the French Ministry of Environment in 1998 revealed that dioxin emissions from the MSW incinerator in Besançon, France were 16,3 ng-TEQ Nm^{-3}, which exceeded the European guideline value of 0,1 ng-TEQ Nm3 (Floret et al., 2007; Viel et al., 2011).

Floret et al. (2007) and Viel et al. (2011) conducted studies to investigate the nature of dioxin soil contamination in the vicinity of the Besançon incinerator. It was intended to determine whether dioxins could be attributed to more than one source of emission. Nevertheless, due to the sampling site selection process, the high similarity in the congener profiles, and the absence of other polluting industries in the study area, it was concluded that the presence of dioxins in the study area was due solely to the MSW incinerator. As a result, following the closure of the most polluting furnaces and the replacement of an advanced Air Pollution Control system-equipped new furnace, a gradual decline in dioxin concentration in soils is anticipated in the study area.

3. Methodology

The present study undertook an extensive examination of the waste treatment processes through the WTE method in various countries worldwide, with a particular emphasis on European nations. Furthermore, the study sought to evaluate the statistical data spanning from 1995 to 2020 to identify the countries that have experienced significant changes in their WTE waste management practices. Consequently, the selected countries, including the United Kingdom (UK), Austria, Poland, Ireland, Norway, Finland, Lithuania, and China, were chosen for a comprehensive review. By analysing the waste management strategies and trends in these countries, the study aimed to provide insights into the effectiveness of the WTE approach in waste management. Therefore, speed of transition and environmental impacts in the process in these countries will be investigated and estimated.

The objective of this chapter is to provide an in-depth analysis and discussion of all the formulas and data used in the present study. By presenting the formulas and data in detail, this chapter aims to provide a comprehensive understanding of the methodology used in the study, as well as the reliability and validity of the results obtained.

3.1. Determining speed of transition

One of the main goals of this study is to calculate the speed of transition from landfill to WTE. Therefore, this study aims to show the progress of WTE in the selected countries During the most significant transition period, while also showing the decrease in landfill treatments. The speed of transition for each case country is calculated with respect to the trend of WTE and landfill changes in a period. First, it should be mentioned that in this study two parameters namely, speed of transition and speed of change are defined. The first one is the overall speed of transition which shows the percentage of changes in WTE treatments per year (see Eq. (2)). The second one is a speed of changes in WTE treatments based on per kt of MSW generated per year, which is expressed as the percentage of changes in the

amount of MSW treated by WTE divided by total mass of MSW generated per year (see Eq. (7)). The speed of change does not depend on the level of starting conditions of WTE and landfill treatments for each case country. Hence, it can be used as a comparable basis for all the case countries for comparison purposes. Also, these methods are applied for landfill changes (see Eq. (4) and (10)).

In order to determine the percentage of WTE development, Eq. (1) is applied to assess the overall progress of WTE during the transition period. The overall speed of transition of WTE is calculated using Eq. (2), where SWT (overall speed of transition of WTE) represents the percentage increase of WTE per year over a selected period (%/a).

Also, the general equation and concept of speed of transition has been used from Orton book (1984) to calculate SWT and percentage of development in WTE. To determine the transition period where WTE had fundamental and significant changes for these different countries, the trend of WTE changes was examined from 1995 to 2020. Subsequently, the years with major transition were selected to be further analysed. The purpose of Eq. (1) is to examine the overall increase of WTE during the transition period and Eq. (2) is to assess the annual percentage increase in WTE adoption during the selected period.

$$pecentage\ of\ development\ in\ WTE = \frac{Y_2 - Y_1}{Y_1} \times 100 \quad (1)$$

$$SWT = \frac{Y_2 - Y_1}{Y_1 \times (X_2 - X_1)} \times 100 \quad (2)$$

Where:

X_1: start point of the transition period (year)

X_2: End point of the transition period (year)

Y_1: Mass of WTE (kt) of MSW in the start point of transition period

Y_2: Mass of WTE (kt) of MSW in the end point of transition period

SWT: Overall speed of WTE transition (%/a) which shows the average percentage of increase of WTE in each year

Since landfill reduces significantly because of the transition to WTE in most of the selected countries, percentage of landfill reduction is also evaluated. Two other equations are defined in this regard. In Eq. (3) is the percentage of reduction in landfill, which shows the percentage of landfill reduction over the transition period (%). Also, Eq. (4), SLT (overall speed of landfill transition) is expressed as a percentage of landfill reduction per year (%/a). The purpose of Eq. (3) is to show the percentage of the increase or decrease of the landfill during the selected period and Eq. (4) is to evaluate the annual decrease or increase percentage of the landfill over the selected period.

$$pecentage\ of\ reduction\ in\ landfill = \frac{Z_2 - Z_1}{Z_1} \times 100 \quad (3)$$

$$SLT = \frac{Z_2 - Z_1}{Z_1 \times (X_2 - X_1)} \times 100 \quad (4)$$

Where:

Z_1: Mass of landfill (kt) of MSW in the start point of transition period

Z_2: Mass of landfill (kt) of MSW in the end point of transition period

SLT: Overall speed of landfill transition (%/a) which shows the average percentage of decrease of Landfill in each year

Using the above equations, the Overall speed of WTE and landfill transition have been shown. Notably, the countries under consideration have witnessed a significant upsurge in the adoption of WTE technology. Therefore, comparing countries is desirable, Eq. (5) and

is used to show how much percentages of MSW generated is treated by the WTE method at the beginning of transition period, whereas Eq. (6) represents how much percentages of MSW generated is treated by the WTE method at the end of transition period. Then Eq. (7) is defined as the speed of WTE changes, and it is used to represent the percentage of WTE changes based on the amount of MSW generated per year. Or in other words, how many percentages of the amount of MSW generated is treated by WTE per year during the transition period. Finally, all the case countries will be compared with each other in the next step. The purpose of this calculation is to obtain the percentage of increase or decrease of WTE compared to the amount of MSW generated and collected in each year during the transition period of each country.

$$A_1 = \frac{mass\ of\ WTE\ in\ Year\ 1}{mass\ of\ MSW\ generation\ in\ Year\ 1} \times 100 \quad (5)$$

$$A_2 = \frac{mass\ of\ WTE\ in\ Year\ 2}{mass\ of\ MSW\ generation\ in\ Year\ 2} \times 100 \quad (6)$$

$$SWC = \frac{A_2 - A_1}{(X_2 - X_1)} \quad (7)$$

Where:

A_1: The percentage of mass of MSW treated by WTE divided by MSW generated at the start point of period

A_2: The percentage of mass of MSW treated by WTE divided by of MSW generated at the end point of period

SWC: The percentage of WTE changes per year divided by of MSW generated during the transition period per year (%-points/a)

Furthermore, this section aims to calculate the speed of landfill changes based on the amount of MSW generated per year. Hence, Eq. (8) and (9) are used to calculate the percentage of

the mass of MSW treated by landfill divided with total mass of MSW generated at the beginning and end point of the transition period. Then total mass of MSW landfilled divided with total mass MSW generation per year. Finally, Eq. (10) is used to show the speed landfill changes based on the MSW generated each year.

$$B_1 = \frac{mass\ of\ landfill\ in\ Year\ 1}{mass\ of\ MSW\ generation\ in\ Year\ 1} \times 100 \quad (8)$$

$$B_2 = \frac{mass\ of\ landfill\ in\ Year\ 2}{mass\ of\ MSW\ generation\ in\ Year\ 2} \times 100 \quad (9)$$

$$SLC = \frac{B_2 - B_1}{(X_2 - X_1)} \quad (10)$$

Where:

B_1: The percentage of mass of MSW treated by landfill divided by of MSW generated at the start point of period

B_2: The percentage of mass of MSW treated by landfill divided by of MSW generated at the end point of period

SLC: The percentage of landfill changes per year divided by of MSW generated during the transition period per year (%-points /a)

3.1.1. Speed of transition data

In the speed of transition section, according to the formulas used in the previous chapter (3.1) it was required to provide statistical data regarding the amount of MSW treated by the WTE as well as the landfill. In addition, data on the amount of MSW generated in selected countries were needed. These data were derived for European countries from Eurostat - Data

Explorer (2022), and for China from the China statistical yearbook (2020), which are shown in **Table 2**.

Table 2. Mass of MSW generation, WTE and landfill treatment (kt) (Eurostat - Data Explorer ,2022; China statistical yearbook 2020)

Countries	Time period	MSW generation (kt)		WTE (kt)		Landfill (kt)	
		Start point	End point	Start point	End point	Start point	End point
Ireland	2008 to 2014	3224	2619	82	893	1939	537
Lithuania	2013 to 2020	1280	1350	92	349	798	220
Austria	2005 to 2020	4732	7438	1310	2652	535	137
Poland	2012 to 2020	12084	13117	51	2823	8085	5218
Finland	2010 to 2019	2519	3123	556	1908	1136	30
Norway	2006 to 2013	2140	2518	675	1446	390	52
United Kingdom	2005 to 2018	35 212	30 786	2942	12615	22569	4613
China	2010 to 2017	158048	215209	23000	102 000	95000	120 000

3.2. GHG and dioxin emissions analysis

3.2.1. GHG emissions

GHG emissions is known as the sum of emissions of various gases: carbon dioxide, methane, nitrous oxide, and smaller trace gases such as hydrofluorocarbons (HFCs) and sulfur hexafluoride (SF6) (World Bank, 2022).

The present study did not consider the impact of the energy generated and the substitution of other energy sources with WTE technology. While such factors could contribute to additional reduction in GHG emissions, their substitution may differ among countries. It is worth noting that the avoided emissions of energy production were not incorporated in the GHG calculations of this study.

One of the goals of this study is to investigate the changes in GHG emissions caused by landfill and WTE during the transition period. Also, this study aims to evaluate whether WTE is a suitable and sustainable solution to reduce GHG emissions or not. Therefore, by using Eq. (11) and Eq. (12) the amount of GHG emissions caused by WTE is calculated at the beginning and end points of the transition period for each country. In addition, it should be mentioned that the amount of GHG emissions caused by the landfill was taken from the Eurostat - Data Explorer (2022), which was calculated based on the IPCC method. Upon reviewing the GHG emissions data on this website, it was discovered that the amount of emissions shown each year is determined by the accumulated mass of the landfill in the past years. Also, since the amount of LFG collection was not available for all countries, it was preferred to rely on Eurostat-statistic (2022) for our calculations and avoid landfill GHG emissions calculations.

For the calculation of GHG emissions from WTE plants, the IPCC (2006) guidelines was followed. The Kyoto protocol lists carbon dioxide (CO_2) and nitrous oxide (N_2O) as part of the GHG inventory from incineration. MSW releases CO_2 and small amounts of other GHGs, such as N_2O, when burned for energy (Ryu, 2012). The CO_2 emissions were calculated using Eq. (11):

$$CO_2 emissions = \sum(SW_i \times dm_i \times CF_i \times FCF_i \times OF_i) \times \frac{44}{12} \quad (11)$$

Where:

SW_i : The waste volume in wet waste (kt/a)

dm_i : Dry matter content in each waste constituent

CF_i : Total carbon content in dry matter

FCF_i : Fraction of fossil carbon in the total carbon

OF_i : Oxidation factor, which equals the combustion efficiency of the WTE plants

$\frac{44}{12}$: Conversion factor from carbon to CO_2

Based on IPCC recommendations, the N_2O emission rate from MSW incineration was determined at 50 g N_2O/t wet MSW (Lou et al., 2017). the total GHG emissions from WTE plants were calculated using Eq. (12):

$$Total\ GHG\ emissions = CO_2 emissions + N_2O\ emissions \quad (12)$$

The input parameter values are presented in chapter 3.2.2 in Table 4 and Table 5.

According to the formulas applied in this section, results obtained in Ireland was unexpected. Thus, these results prompted this study to perform further analysis of GHG emissions caused by landfills. Hence, the amount of GHG emissions from landfills was calculated according to the IPCC method for Ireland. It should be noted that this method shows the amount of GHG released from the amount of landfilled waste in a year and is not cumulative. It is also very approximate because the amount of LFG collection is not considered.

The IPCC guidelines (2006) published as method to reach the amount of GHG emissions by landfill as LFG. The calculation of LFG emissions is based on the IPCC mass balance method. The overall method equation is shown below:

$$GHG_{Landfill} = \left(MSW_T * MSW_F * MCF * DOC * \mathrm{DOC}_F * F * \frac{16}{12} - R\right) \times (1 - OX) \times 28 \quad (13)$$

Where:

$GHG_{Landfill}$: GHG emissions from landfill (kt/a)

MSW_T: Total MSW generation amount (kt/a)

MSW_F: MSW fraction disposed in landfills

MCF: Methane correction factor for aerobic decomposition in the year of deposition, fraction

DOC: Degradable organic carbon in the year of deposition, fraction

DOC_F: Fraction of DOC that can decompose, fraction

F: Fraction of CH4 in generated LFG, volume fraction

R: recovered CH4, fraction

OX: Oxidation factor

28: the characterization factor of methane to convert it to CO_2-eq

The input parameter values are showed in Table 6 and Table **7**.

The most important aspect of this section is the analysis of the changes in GHG emissions caused by landfill and WTE during the transition period. Accordingly, Eq. (14) is used to calculate the percentage (%) of overall changes in GHG emissions from landfill and WTE during the transition period. After that, Eq. (15) and (16) calculate the overall changes in GHG emissions of landfill and WTE respectively. Furthermore, the annual rate of change in GHG emissions for landfilling and WTE (%/a) is calculated by using Eq. (17). Also, using Eq. (18) and Eq. (19) the annual rate of change in GHG emissions for WTE and landfill is calculated separately.

$$OCG = \frac{(L_2+I_2)-(L_1+I_1)}{(L_1+I_1)} \times 100 \quad (14)$$

$$OCG_W = \frac{I_2-I_1}{I_1} \times 100 \quad (15)$$

$$OCG_L = \frac{L_2-L_1}{L_1} \times 100 \quad (16)$$

$$RGE = \frac{\frac{(L_2+I_2)-(L_1+I_1)}{(L_1+I_1)}}{Y_2-Y_1} \times 100 \quad (17)$$

$$RGE_w = \frac{\frac{I_2-I_1}{I_1}}{Y_2-Y_1} \times 100 \quad (18)$$

$$RGE_L = \frac{\frac{L_2-L_1}{L_1}}{Y_2-Y_1} \times 100 \quad (19)$$

Where:

L_1: The amount of GHG emissions (kt) from landfill at beginning of transition period

L_2: The amount of GHG emissions (kt) from landfill at the end of transition period

I_1: The amount of GHG emissions (kt) from WTE at the beginning of transition period

I_2: The amount of GHG emissions (kt) from WTE at the end of transition period

OCG : The relative overall changes in GHG emissions from landfill and WTE during the transition period (%)

OCG_W: The relative overall changes in GHG emissions from WTE during the transition period (%)

OCG_L: The relative overall changes in GHG emissions from landfill during the transition period (%)

RGE: Annual rate of change percentage of overall changes in GHG emissions from landfill and WTE during the transition period (%/a)

RGE_w: Annual rate of change percentage of overall changes in GHG emissions from WTE during the transition period (%/a)

RGE_L: Annual rate of change percentage of overall changes in GHG emissions from landfill during the transition period (%/a)

As part of this analysis, it is also necessary to determine how much GHG emissions are emitted per kt of MSW treated by landfill and WTE in each year. Equation (20) shows the sum of GHG emissions from landfilling and WTE divided with the sum of landfilled and incinerated masses of MSW.

$$G = \frac{E_{LI}}{m_{LI}} \quad (20)$$

Where:

G : The amount of GHG emissions from WTE and landfill divided by total mass of landfill and WTE per year (kt CO_2eq/(kt $_{(landfill\ +WTE)}$)

E_{LI}: Total amount of GHG emissions (kt CO_2eq/a) from landfill and WTE

m_{LI}: Total mass of landfill and WTE (kt/a) in treatment of MSW

Evaluation of GHG emissions changes is also one of the major aspects of this study. To be able to compare the case countries with each other in this section. It was decided to calculate the changes in GHG emissions based on MSW treated by landfill and WTE during the transition period by Eq. (21).

$$\Delta G = G_2 - G_1 \quad (21)$$

ΔG: Changes in the amount of GHG emissions (kt CO_2eq/kt $_{(landfill\ +WTE)}$) divided by total mass of landfill and WTE during transition period

G_1: The amount of GHG emissions from WTE and landfill divided by total mass of landfill and WTE at the start point of transition period (kt CO_2eq/kt ($_{landfill\ +WTE)}$)

G_2: The amount of GHG emissions from WTE and landfill divided by total mass of landfill and WTE at the end point of transition period (kt CO_2eq/kt $_{(landfill+WTE)}$)

Also, this study aims to show the amount of GHG emissions in per kt of MSW treated by the WTE method at the beginning and end of the transition period. This calculation also considered for the landfill, but since the data related to the GHG emissions of the landfill is based on the accumulated mass of the landfill in the past years. consequently, this calculation is not logical for the landfill and cannot be scientific criteria. Equation (22) shows the amount of GHG emissions in per kt of MSW treated by the WTE.

$$GI = \frac{E_I}{m_I} \quad (22)$$

Where:

$GI_:$ the amount of GHG emissions of WTE divided by total mass of WTE per year (kt CO_2-eq/kt $_{(WTE)}$)

E_I: Total amount of GHG emissions (kt CO_2eq/a) from WTE

m_I: Total mass of WTE (kt/a) in treatment of MSW

3.2.2. GHG emissions data

In this section, according to the equations that are examined to show the trend of GHG emissions from landfill and WTE treatment of MSW, statistical data on GHG emissions are required in selected countries. This study aimed to calculate the amount of GHG emitted from landfill and WTE using the IPCC method. Since the amount of LFG collection is not known in the selected countries, it was decided to adapt the GHG emissions data in the landfill section from the Statistics Eurostat (2022) for European countries. Also, for China, official statistical information on the amount of GHG emissions caused by landfills were not available. Then the amount of methane emissions from landfill in 2010 and 2017 was taken

from an article by Bian et al. (2022). Finally, these emissions were converted to CO_2 equivalents. It should be noted that all these data were calculated by the IPCC method (**Table 3**).

Table 3. GHG emissions from landfill (Bian et al., 2022; Statistics Finland Greenhouse Gases, 2020; Statistics Eurostat, 2022).

Countries	Time period	Landfill CO_2eq (kt)	
		Start point	End point
Ireland	2008 to 2014	468	648
Lithuania	2013 to 2020	874	563
Austria	2005 to 2020	2 184	831
Poland	2012 to 2020	10 094	7 543
Finland	2010 to 2019	2 168	1 425
Norway	2006 to 2013	1 392	1 197
United Kingdom	2005 to 2018	43 740	14 207
China	2010 to 2017	95 480	151 256

On the other hand, the method recommended by the IPCC was used to calculate the GHG emissions caused by WTE. In this method which is represented in chapter 4 as Eq. (11), dry matter content, percentage of total carbon content in dry weight, percentage of fossil carbon fraction of total carbon, and oxidation factor are required. These values were adapted from the default values in Table 2.4 in Chapter 2 of the IPCC report (IPCC-TFI, 2006). **Table 4** shows the values for each fraction of MSW.

Table 4. Required values for GHG emissions calculation of WTE (IPCC-TFI, 2006)

Category	Dry matter content %	Total carbon Content in % of dry weight	Fossil carbon fraction in % of total carbon	Oxidation factor (combustion efficiency)
Paper	90 %	46 %	1 %	1
Textile	80 %	50 %	20 %	1
Food Waste	40 %	38 %	0 %	1
Wood	85 %	50 %	0 %	1
Plastic	100 %	75 %	100 %	1
Metal	100 %	NA*	NA*	0
Glass	100 %	NA*	NA*	0
Other, inert waste	90 %	3 %	100 %	1

* Metal and glass contain some carbon of fossil origin. Combustion of significant amounts of glass or metal is not common.

In addition, to calculate the GHG emissions from WTE based on Eq. (11), the composition of MSW is required for each country. The composition of MSW for each country is compiled in Table 5. To calculate GHG emissions at the start and end of the transition period, this study endeavoured to determine the composition of MSW for years proximate to the beginning and end of the transition period in each country. Regrettably, such data was unavailable for many countries. Consequently, for some countries, a fixed composition was assumed during their transition period in the computations, while for others, dissimilar compositions were employed for the beginning and end of the period when such information was obtainable. Moreover, the composition of mixed waste was not accessible for all countries, except Finland, whereby the MSW composition was used instead of mixed waste composition for some countries. Also, the MSW composition for Ireland was assumed to be similar to that of the UK. In addition, the composition of MSW was not available for Norway, which was considered the average composition of European countries for this country.

Table 5. Composition MSW used in calculation GHG emissions

Countries	Year	Paper (%)	Textile (%)	Food Waste (%)	Plastic (%)	Metal (%)	Glass (%)	Wood (%)	Other, inert waste (%)	Reference
China	2017	16,9	2,3	53,7	13,6	0,6	1,5	2,2	9,2	(Ma et al., 2020)
Finland	2020	17	6,1	32,3	16,6	2,4	2,3	1,5	21,8	(KIVO, 2020)
Poland	2014	13,4	2,8	37,3	12,1	2,2	10,1	0,5	21,7	(Alwaeli, 2015)
United Kingdom and Ireland	2000	33,2	2,1	20,2	10,2	7,3	9,3	-	20,6	(Burnley, 2007)
United Kingdom and Ireland	2017	17,4	4,7	20,31	9,3	3,8	6,9	17,2	20,4	(Compositional waste in the United Kingdom, 2019)
Lithuania	2007	14	8,5	44	9	3	11	8,5	2	(Luoranen et al., 2009)
Lithuania	2019	8,6	7,9	34,8	12	1,7	4,4	0,93	29,7	(Stankevičiūtė, 2021)
Austria	2020	14,2	5,1	36,1	14,2	4,3	8,1	3,8	13,2	(Environment Agency Austria, 2021)
EU Average	2005	32,1	4,1	30,2	7,3	8,2	10,1	-	8,9	(Zhang et al., 2010)

According to the unexpected results obtained related to GHG emissions in two countries, this study was intended to calculate the GHG emissions caused by the landfilling per year according to the IPCC method. The necessary data for this calculation, which is needed for Eq. (13), is shown in Table 6Table 7. These data are taken from the IPCC guidelines by default.

Table 6. DOC values for each fraction (IPCC-TFI, 2006)

Waste Fraction	DOC (%)
Food waste	15
Garden	20
Paper	40
Wood and straw	43
Textiles	24
Disposable nappies	24
Sewage sludge	5
Industrial waste	15

Table 7. Required values to calculate GHG emissions from landfill (IPCC-TFI, 2006)

F (%)	50 %
OX (%)	0 %
DOC_f (%)	77 %
MCF (%)	60 %

3.2.3. Dioxin emissions from incinerator

According to the article of Cudjoe and Acquah (2021), Eq. (23) is used to calculate the amount of dioxin emissions from burning waste in incineration plants. It should be noted that two factors are essential in this calculation, one is the amount of MSW that is incinerated and the second is the value of the dioxin emissions factor. Article of Wei, Li, Liu, and Zhong (2022) was studied to obtain the dioxin emissions factors in each country. Then the amount of dioxin emissions was calculated for each country per year. For the countries that dioxin emissions factors were not available, the dioxin emission factor of a country with similar mass of WTE treatment of MSW was assumed.

$$DEI_{dioxin} = DF_I \times m_I \quad (23)$$

Where:

DEI_{dioxin}: The mass of dioxin emissions (mg-TEQ/a) from WTE plants

DF_I: Dioxin emissions factor from WTE plants (μg I-TEQ/t MSW)

m_I: Total mass of WTE (kt/a) in treatment of MSW

3.2.4. Dioxin emissions from landfill fire

The concept of Eq. (24) is taken from article of Dwyer and Themelis (2015) to calculate the amount of dioxin released from the landfill fire.

$$DEL_{dioxin} = NLF \times WBL \times DF_L \quad (24)$$

Where:

DEL_{dioxin}: Total amount of dioxin emissions (mg-TEQ/a) from landfill fire

NLF : Number of landfill fire in each year

WBL: The Mass of waste burnt per landfill fire (kt)

DF_L: Dioxin emission factor from landfill fire (ng TEQ/kg $_{burned}$)

To calculate the amount of dioxin emission based on the Eq. (24), the dioxin emissions factor, the number of landfill fires and the amount of waste burned in each landfill fire are required. It should be noted that data to calculate dioxin emissions were available only for the United States in 2011 and Finland in 1992. The data from the US article in 2011 was used because it is more up to date than that from Finland. Also, these data are assumed to be same for all case countries.

Since there is a lack of required data in the case countries, Eq. (25) was created to determine the ratio of MSW burned in landfill and total landfill waste. This ratio is used for other countries where data was not available. Also, in this equation to reach the total amount of MSW burned per year (*MLB*), the number of landfill fires should be multiplied by the amount of MSW burned in each landfill fire. $W_{waste\ burned}$ represents the total mass of MSW that is burned in the landfill fire process based on the total mass of landfill waste per year, and it is used to get the approximate amount of MSW burned in landfill fires in each country per year. It should be mentioned this estimation is very rough because the probability of landfill fires varies amongst nations due to diverse climatic circumstances, levels of MSW treatment, and levels of methane capture. However, better data has not been reported and studied in this regard. Hence, it is possible to use this kind of generalization to get the possible magnitude of the dioxin emissions in landfill fires.

Furthermore, regarding the obtained ratio from Eq. (25), $W_{waste\ burnt}$ is a constant number (0,0187) and it is used for all case countries. Also, the dioxin emissions factor of landfill fire which is given from article of Dwyer and Themelis (2015) is assumed to be constant number for all selected countries with a value of 700 ng TEQ/kg $_{burned}$. Finally, by multiplication of these two constant numbers, Eq. (26) is simplified with one constant number. Then we multiply this number for each selected country by the amount of landfilled waste in order to

have a very approximate estimate of the amount of dioxin emissions in landfill fires (Dwyer and Themelis, 2015).

$$W_{waste\ burned} = \frac{MLB}{MSW_{landfill}} = \frac{11000\ fire \times 225\ ton/fire}{131\ 770\ 000\ t} = 0{,}0187 \quad (25)$$

$$DEL_{dioxin} = 13{,}09 \frac{\mu g}{kt_{landfill}} \times MSW_{landfill} \quad (26)$$

Where:

$W_{\ waste\ burned}$: The ratio of amount of waste burned in landfill to the total amount of MSW treated by land fill in US per year which is constant number 0,0187 ($kt_{burned}/kt_{landfill}$)

$MSW_{\ landfill}$: Total mass of landfill (kt) in treatment of MSW per year

MLB: Multiplication of number of fires occur per year by the amount the mass of waste burned in landfill fire in each fire (Total mass of waste burned in landfill in each year)

$DEL_{\ dioxin}$: Total amount of dioxin emissions (mg-TEQ/a) from landfill fire

13,09: The constant number ($\frac{\mu g}{kt\ landfill}$) for each selected countries which is result of multiplication of dioxin emissions factor of landfill fire by $W_{\text{ waste burned}}$

Moreover, Eq. (27) is defined to estimate how much dioxin is released in per kt of MSW treated by landfills and WTE per year. By this estimation, it would be possible to analyze the trend of dioxin emissions during the transition period and show how increasing the utilization of WTE and avoiding landfills have affected dioxin emissions in a transition period.

$$EMT = \frac{DEl + DEI}{m_{LI}} \quad (27)$$

Where:

EMT: Amount of dioxin emissions (mg-TEQ/kt $_{(landfill+WTE)}$)

DEL $_{dioxin}$: Total amount of dioxin emissions (mg-TEQ/a) from landfill fires

DEI $_{dioxin}$: The mass of dioxin emissions (mg-TEQ/a) from WTE plants

m_{LI}: Sum of landfilled and incinerated waste (kt/a)

In the final step, Eq. (28) is created to show the total changes of dioxin emissions from landfill fire and WTE based on per kt of MSW treated by landfill and WTE during the transition period. Then Eq. (30) shows percentage of increasing or decreasing rate per kt of MSW treated by landfills and WTE per year.

$$\Delta D = D_2 - D_1 \quad (28)$$

ΔD: Changes in the amount of dioxin emissions (mg-TEQ/kt $_{(landfill+WTE)}$) from landfill fire and WTE during transition period

D_1: Total amount of dioxin emissions (mg-TEQ/kt $_{(landfill+WTE)}$) from landfill fire and WTE at the start point of transition period

D_2: Total amount of dioxin emissions (mg-TEQ/kt ($_{landfill+WTE)}$) from landfill fire and WTE at the end point of transition period

3.2.5. Dioxin emissions data

According to the formulas that are used for calculating the amount of dioxin emissions caused by landfill fire and incinerator, statistical data on the amount of MSW that is treated by landfill and WTE methods per year are shown in Table 2.

In addition, to calculate the amount of dioxin emissions caused by landfill fire and incineration, the dioxin emissions factor is required for each country. Using the article of Wei, Li, Liu, and Zhon (2022), dioxin emissions factors for incinerators are obtained in selected countries (Table 8), although for some case countries it was not available. Nevertheless, for the countries that were not available, it was assumed the dioxin emissions factor of the country that has a roughly similar the mass of WTE treatment of MSW to it. Consequently, in the case of Ireland and Lithuania, the dioxin emission factor value of Poland was utilized, while in Norway, the dioxin emission factor value of Austria was applied. Moreover, the dioxin emission factor value for the UK was assumed to be equivalent to that of China. The corresponding dioxin emission factors applied to each country are provided in Table 8.

Table 8. Dioxin emissions factors of incinerators (Wei, Li, Liu and Zhon, 2022)

Dioxin emissions factors of incinerators	μg I-TEQ/t MSW
Ireland	0,0525
Lithuania	0,0525
Poland	0,0525
Finland	0,3200
Austria	0,0440
Norway	0,0440
United Kingdom	0,1700
China	0,1700

On the other hand, since many studies have not been conducted on landfill fires, the value of the dioxin emissions factor is not available for all countries, and it is not reported in the annual statistics in each country. Hence, the data from the article of Dwyer and Themelis (2015) was used, especially the dioxin emissions factor as a constant factor of landfill fire for all selected countries. Furthermore, the number of landfill fires and the mass of landfill

fires per fire in the article of Dwyer and Themelis (2015) is used to create a constant ratio that is explained in sub-chapter (3.2.4). Table 9 shows the data required for calculating dioxin emissions.

Table 9. Data required for landfill fire dioxin emissions (Dwyer and Themelis, 2015)

Dioxin emissions factor of landfill fire	700	ng TEQ/kg burned
Number of landfill fire in USA in 2011	11 000	fire
Mass of landfill fire per fire	225 000	kg

4. Results and discussion

4.1. Speed of transition research

In this section, the speed of transition from landfill to WTE method is estimated in all the selected countries during the period with the most significant transition. The estimation is based on all the equations defined in sub-chapter 3.1 It is important to note that positive values in this section indicate the growth of WTE and landfill treatment, whereas negative values indicate a decrease in WTE and landfill treatment. It is also essential to mention that the landfilling rate is influenced not only by the WTE method but also by waste generation, recycling, and other material recovery practices. However, these factors are not considered in this thesis.

4.1.1. Ireland

The biggest change in recent years has been the transition from landfill disposal to energy recovery for residual waste. With a total licensed capacity to treat 835 kt of non-hazardous municipal waste annually, Ireland now has two incinerators that treat MSW for energy recovery. In addition, three cement kilns are authorized to accept solid recovered fuel (SRF) for co-incineration as a substitute for fossil fuels. In early 2016, it was determined that landfill capacity was severely limited and required additional capacity was authorized to avoid negative environmental impacts, such as stockpiling of wastes and illegal operations (National Waste Statistics Summary Report for 2018, 2020).

WTE technology has been used in Ireland since 2007 according to Statistics Eurostat (2022). As shown in Figure 5, during the period 2008-2014, the most significant changes occurred in the utilisation of WTE technology. The results of applying Eq. (1111) and (22) demonstrate that the percentage of development in WTE and the overall speed of WTE

transition have been estimated at 989 % and 165 %/a, respectively. While the results of Eq. (3) and (4) indicate that the, the percentage of landfill reduction has been 72 % during the transition period and overall speed of landfill transition has been -12 %/a. In summary, Ireland has been able to increase the use of WTE treatment by 989 % during the transition period. In addition, Ireland has reduced landfill treatment by 72 % during the same period. The trend of WTE and landfill treatment during the transition period is shown in Figure 5.

It is worth mentioning that in 2013 and 2015, there was no MSW incineration data available for Ireland. Therefore, to compensate for the missing data, an averaging method was used. Specifically, the incineration data for those years was estimated by taking the average of the data from the previous and subsequent years. This approach was adopted to ensure the completeness and accuracy of the data used in the study, as the missing data could have a significant impact on the overall results. It is essential to note that the averaging method used in this study is a commonly adopted approach in research studies to address missing data, and it is considered a reliable and valid method.

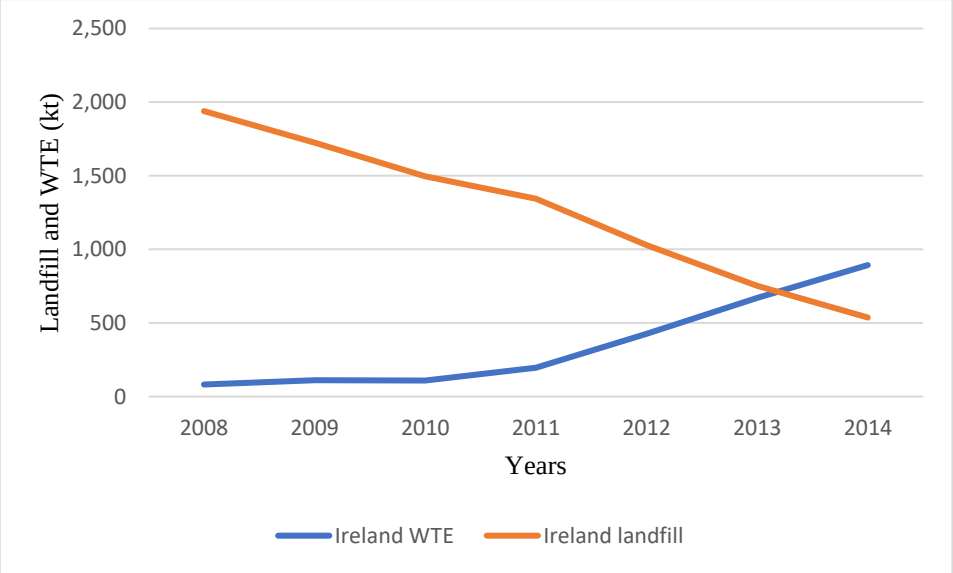

Figure 5. Amount of WTE and landfill in Ireland during the transition period (Eurostat - Data Explorer, 2022)

According to the selected period between 2008 and 2014 in Ireland, a step was taken further to compare MSW generation and WTE treatment. The results demonstrate that for the selected period between 2008 and 2014, WTE treatment significantly increased while the amount of MSW generated slightly decreased followed by a drastic decrease in landfill. Figure 6 below represents the trend of MSW generation and WTE and landfill treatment. The aim of defining A_1, A_2, B_1 and B_2 parameters (Eq. (5), (6), (8) and (9)) is to calculate the percentage of MSW treated by WTE and landfill methods based on MSW generation at the beginning and end of the transition period. Then speed of changes in WTE and landfill treatment based on per kt MSW generated in per year are determined by *SWC* (see Eq. 7) and *SLC* (see Eq. 10) parameters. Hence, in Ireland A_1 in 2008 was 3 % and A_2 increased to 34 % in 2014 at the end of the transition period. While B_1 in 2008 was 60 % and B_2 decreased to 21 % in 2014. The overall changes in WTE and landfill based on per kt of MSW generated raised by 32 %-points and decreased by 39 %-points respectively. By determination of *SWC*, the speed of WTE changes based on per kt of MSW generated during the transition period per year is 5.3 %-points/a. Also, *SLC* was -6,6 %-points/a which means annually 6,6 % of MSW treated by landfill based on per kt of MSW generation decreased.

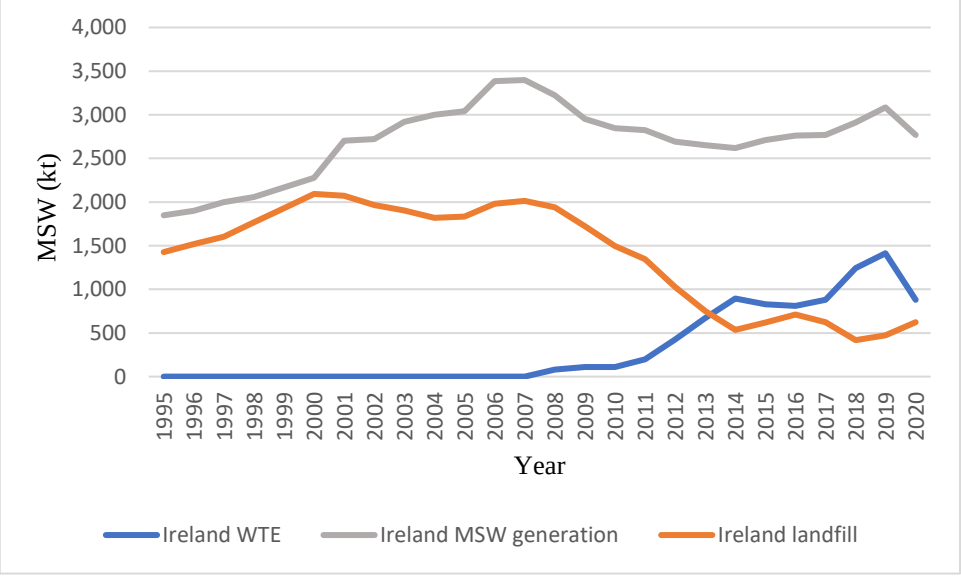

Figure 6. Comparison of MSW generation and WTE and Landfill treatment in Ireland (Eurostat - Data Explorer, 2022)

4.1.2. United Kingdom

As the UK approaches its recycling goals, there is less "residual" waste that can be burned. The development of additional incinerator capacity may lead to overcapacity since there is more incineration capacity than there is residual waste available for burning. This can negatively impact recycling efforts. From 2000 to 2001, the amount of waste collected by local authorities in England increased by 3 %. The Prime Minister's Strategy Unit predicted in 2002 that "household waste would double by 2020" if growth rates remained unchanged. However, this led to long-term contracts for waste. In actuality, total waste levels have decreased by about 8 % instead of rising to over 50 Mt (Facts & Figures – United Kingdom Without Incineration Network, 2021).

Based on Figure 7, the transition from landfill to WTE occurred rapidly from 2005 to 2018. It is observed, even since 2014, that the amount of MSW treated by the WTE method exceeds the amount of landfill. The speed of WTE has progressed with a significant trend, therefore, the overall speed of WTE transition (*SWT*) was estimated to increase by 25 %/a, and the percentage of development in WTE increased by 329 %.

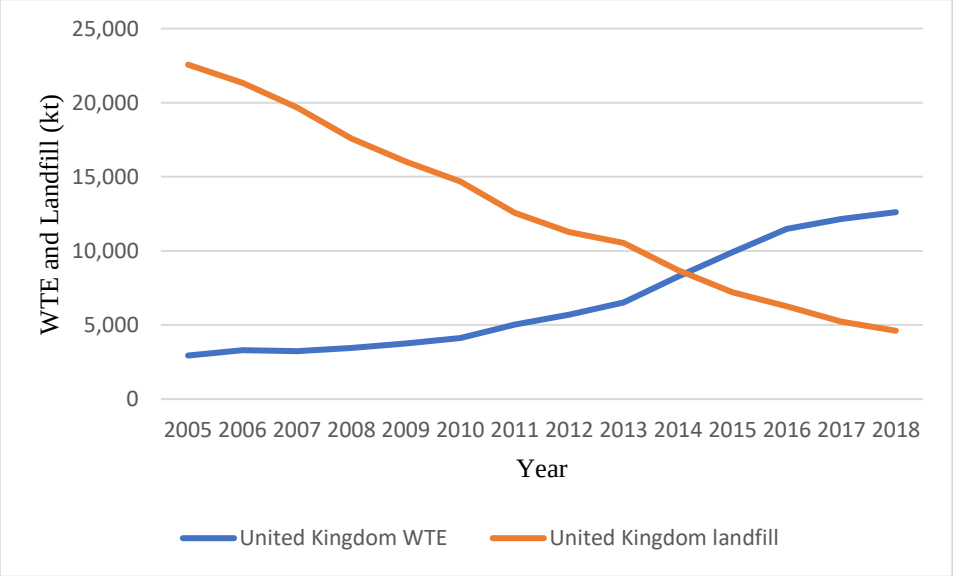

Figure 7. Amount of WTE and landfill in UK during the transition period (Eurostat - Data Explorer, 2022)

On the other hand, as can be seen from Figure 7, the UK has been able to significantly reduce landfills during this transition period. Also, by calculations for the speed of landfill transition, it can be showed that the overall speed of landfill transition (*SLT*) was -6 %/a, and the percentage of landfill reduction was 80 % during the transition period. Overall, the UK has been able to increase the use of WTE by 329 % over 13 years and reduce landfill by 80 % over the same period.

Figure 8 below illustrates the trends of MSW generation, WTE changes and landfill changes during the specified period. According to the calculations that have been done for the UK, the percentage of WTE based on MSW generated at the beginning of the determined period (A_1) in 2005 was 8 %) and at the end point in 2018 it (A_2) was 41 %. Furthermore, *SWC* (see Eq. (11111)) stood at 2.5 %-points/a. Whereas, the percentage of landfill based on the MSW generated at the beginning of the determined period (B_1) in 2005 was 64 % and it (B_2) was reduced to 15 % in 2018 ate the end of the transition period. Also, *SLC* (see Eq. (10)) was -3,7 %-points/a.

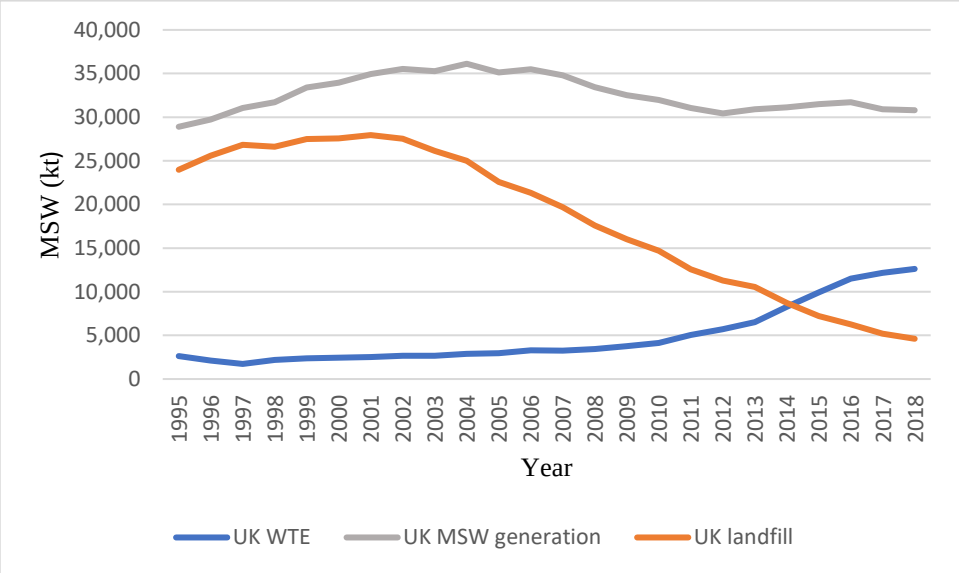

Figure 8. Comparison of MSW generation and WTE and Landfill treatment in UK (Eurostat - Data Explorer, 2022)

4.1.3. Norway

There were 17 WTE plants in Norway as of 2015, processing about 1.70 Mt of MSW and producing approximately 4 TWh of heat for district heating systems. Additionally, there was some steam and electricity available for industrial purposes (Becidan, M., et al., 2015). The treatment of MSW in Norway consists of two main methods: incineration and recycling. A total of 50 % of waste was incinerated in 2010, while 42 % was recycled (Norway - Municipal Waste Management — European Environment Agency, 2010).

According to Figure 9, there has been a steady increase in WTE until 2005 where the amount of WTE increased rapidly from 2006 to 2013. For this reason, this period has been chosen to calculate the speed of transition in WTE, and *SWT* (see Eq. (2)) was 16 %/a, and the percentage of development in WTE was calculated to be 114 %. In the same period, the amount of landfill decreased significantly, so that in 2006, the amount of landfill in this country was 390 kt/a, while in 2013, it reached 52 kt/a. Therefore, *SLT* (see Eq. (10)) was -12 %/a, and the percentage of landfill reduction was 87 % during the transition period.

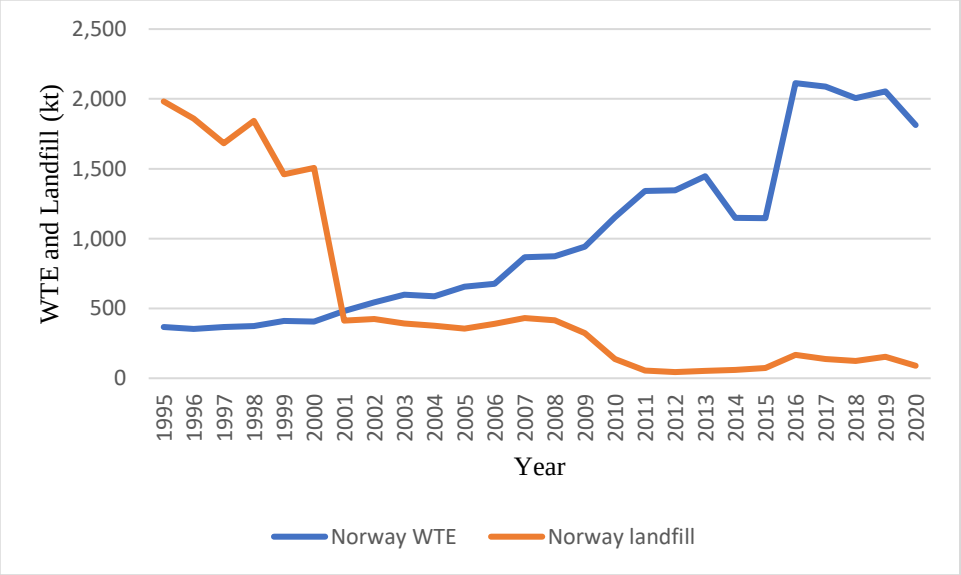

Figure 9. Amount of WTE and Landfill in Norway (Eurostat - Data Explorer, 2022)

The generation of MSW per capita increased slightly from 2004 to 2007 and remained at around 490 kg per capita until 2013. However, various factors such as population and household expenditure influence MSW generation. The waste prevention program (WPP) from 2013 resulted in a decrease in MSW generation from 496 kg per capita in 2013 to 422 kg per capita in 2015. However, from 2016 onwards, MSW generation surpassed 700 kg per capita, due to a break in the data collection, and significantly exceeded the EU average of 495 kg per capita (Waste prevention in Europe, 2021).

As can be seen in Figure 10 below, a comparison of the trends in MSW generation, WTE, and landfill treatment shows that from 2006 to 2013, WTE treatment has increased parallel to the increase in MSW generation, whereas landfill treatment decreased. Therefore, the percentage of WTE based on the MSW generation in 2006 at the starting point of the selected period (A_1), was 32 %, and it (A_2) reached to 57 % at the end of the period in 2013. Analysing this data indicate that Norway has experienced 26 %-points growth in WTE based on MSW generated during the transition period. Also, *SWC* (see Eq. (7)) stood at 3.7 %-points/a. On the other hand, the percentage of landfill based on the MSW generated in 2006 at the starting point of the selected period, was 18 % (B_1), and it decreased to 2 % (B_2) at the end of the period. These decreasing changes demonstrate that *SLC* (see Eq. (10)) was -2,3 %-points/a.

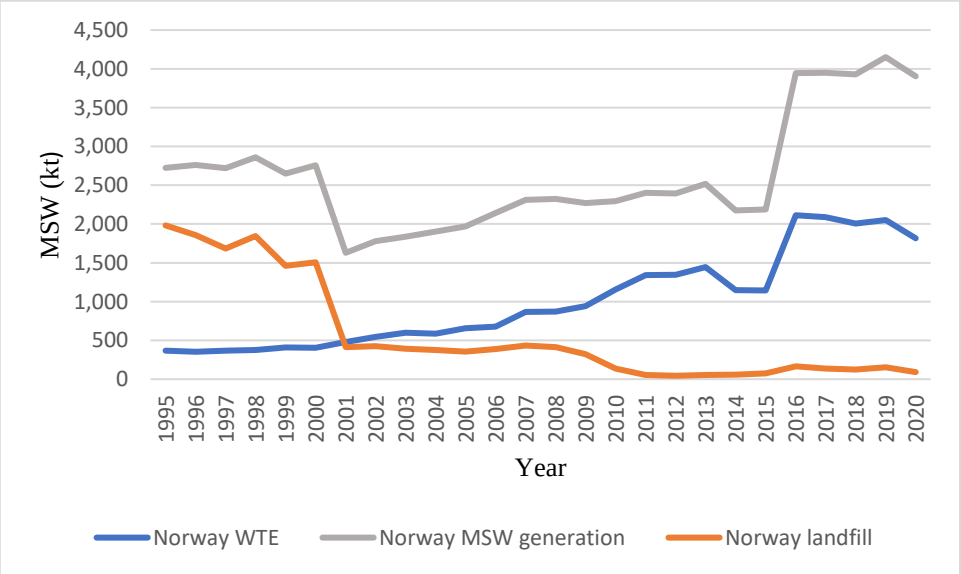

Figure 10. Comparison of MSW generation and WTE and Landfill treatment in Norway (Eurostat - Data Explorer, 2022)

4.1.4. Finland

Since 1994, waste incineration with energy recovery has been rapidly increasing due to the revision of the Waste Act, the revision of the Environmental Protection Act and the ban on organic waste being disposed of in landfills since 2016 in addition to the escalating price of traditional fuels (Finnish Environment Institute Centre for Sustainable Consumption and Production, 2022) As part of the MSW management process in Finland, WTE plays an extremely critical role, so that As a result, WTE plants produce about 8-10 % of the district heating supply, which varies based on the yearly heating demand of buildings and seasonal weather changes (Waste-to-Energy Solutions - EastCham Finland Ry, 2022.

It is evident that there has been a significant increase in the mass MSW treated by WTE in Finland over the last 9 years based on the data obtained from Eurostat - Data Explorer (2022). In 2010, the mass of MSW treated by WTE in Finland was 556 kt/a compared to 1736 kt/a in 2020, illustrating a substantial increase over the last 9 years. While in the same period, Finland was able to reduce the amount of landfill from 1136 kt/a to 30 kt/a (Figure 11). Hence, the SWT (see Eq. (2)) is estimated in this period to be 24 %/a, and the percentage of development in WTE percentage is estimated to be 212 %. Also, *SLT* (see Eq. (4)) and percentage of landfill reduction are determined -11 %/a, and 97 % respectively during the transition period. The overall results represent that Finland has been able to reduce landfilling by 97 % and increase WTE treatment by 212 % during the transition period.

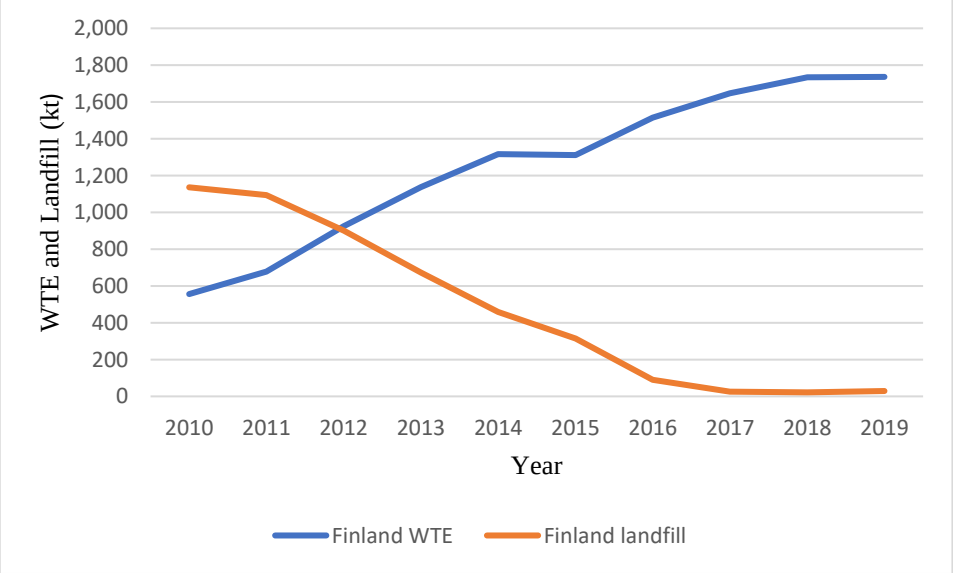

Figure 11. Amount of WTE and Landfill in Finland (Eurostat - Data Explorer, 2022)

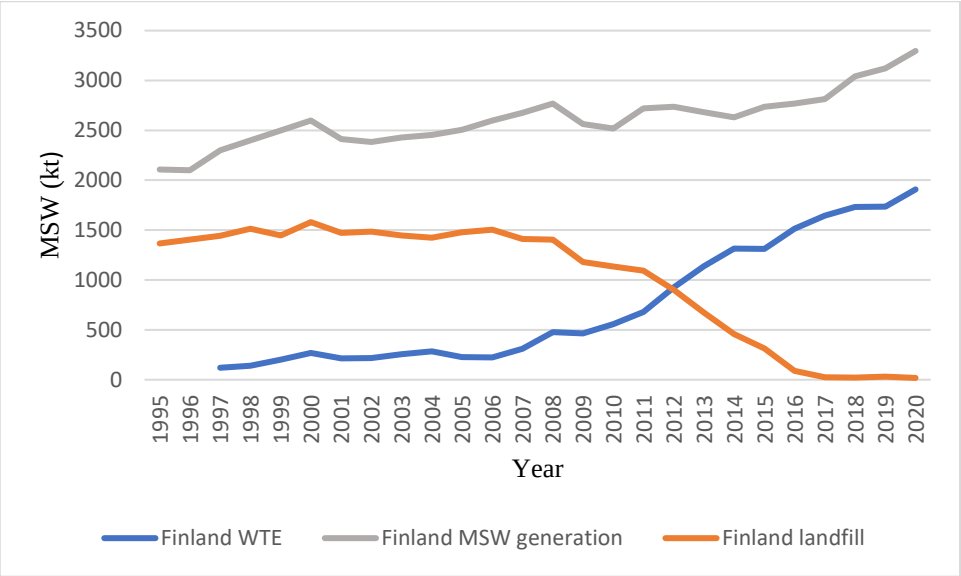

Figure 12. Comparison of MSW generation and WTE and landfill treatment in Finland (Eurostat - Data Explorer, 2022)

Figure 12 clearly shows a dramatic increase and decrease in the amount of WTE and landfill treatment during the determined period that in contrast, the amount of MSW generated has increased slightly. Based on the calculations to demonstrate the percentage of WTE based on the MSW generated, this value in 2010 at the starting point of the selected period (A_1) stood at 22 %, and it (A_2) reached a peak 56 % at the end of the period in 2019 that shows 34 %-points growth over the 9 years. On the other side, the percentage of landfill based on the MSW generated in 2010 at the starting point (B_1), stood at 45 %, and it (B_2) decreased dramatically to 1 % at the end of the transition period. Therefore, the speed of WTE and landfill changes based on MSW generated (*SWC*, *SLC*) were 3.6 %-point/a, and -4,9 %-point/a respectively.

4.1.5. Poland

Currently, the WTE sector in Poland is experiencing rapid growth. Several projects are currently being carried out, including the construction of several modern MSW incineration facilities. These undertakings remain in progress. (Warsaw Waste To Energy Project, Poland, 2021; Malinauskaite et al., 2017). According to calculations made to represent the

progress of WTE method as well as the potential reduction of landfill, before 2012, there was almost no WTE treatment, however, from 2012 to 2020, WTE treatment increased by a 5400 % and the overall speed of WTE transition (*SWT*) was estimated as 680 %/a. Furthermore, the overall speed of landfill transition (*SLT*) is estimated to be -4 %/a, and the percentage of reduction was 35 % over the transition period.

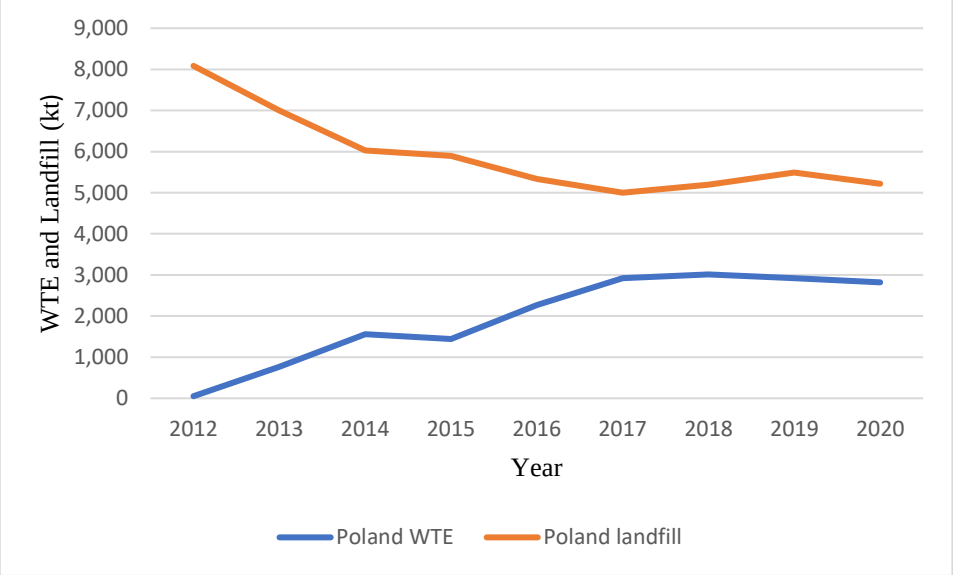

Figure 13. Amount of WTE and landfill in Poland (Eurostat - Data Explorer, 2022)

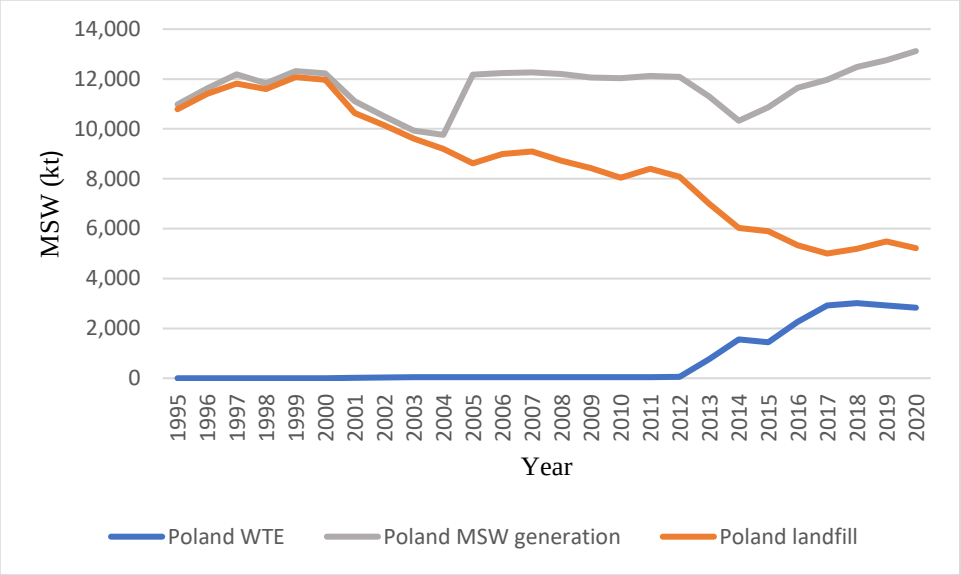

Figure 14. Comparison of MSW generation and WTE and landfill treatment in Poland (Eurostat - Data Explorer, 2022)

As can be seen in Figure 14, the amount of WTE treatment is significantly lesser than amount of MSW generation. Hence, it can be concluded that Poland has a high capacity to utilize WTE technology to produce energy from MSW. Furthermore, Figure 14 indicates that roughly all the MSW generated in Poland between 1995 and 2003 was disposed of in landfills. However, after that, especially during the transition period, the amount of landfill decreased compared to the MSW generated. Between 2012 and 2020, as a result, the percentage of MSW treated by WTE based on the MSW generated (A_1) in 2012 was 0,42 % which (A_2) increased to 22 % in 2020. These values demonstrate roughly 21 %-points growth of WTE treatment based on MSW generated over the 8 years. Also, *SWC* (see Eq. (7)) was 2,6 %-points/a over the transition period. On the other side, the percentage of landfill based on the MSW generated in 2012 was 67 % (B_1) which declined to 40 % (B_2) in 2020. *SLC* (see Eq. (10)) was 3,39 %-points/a during the transition period.

4.1.6. Austria

In Austria, there are 11 MSW incineration facilities with a combined annual capacity of 2.5 Mt (Warrings & Fellner, 2018). It is expected that the Austria WTE market will grow at a compound annual growth rate (CAGR) of more than 3.3 % during the period 2020-2025.

According to the Austrian government, 34 % of the country's energy must be derived from renewable sources by 2020, and this is driving the adoption of WTE market (Austria waste to energy market, 2022). To estimate the speed of transition of WTE in Austria, a fifteen-year period between 2005 and 2020 was evaluated, and the overall speed of WTE transition (*SWT*) was 7 %/a, and the percentage of development in WTE was 102 %. Also, the speed of landfill transition (*SLT*) was -5 %/a, and the total landfill reduction during the transition period was 74 %. Figure 15 illustrates the trend of WTE and landfill treatment over the transition period.

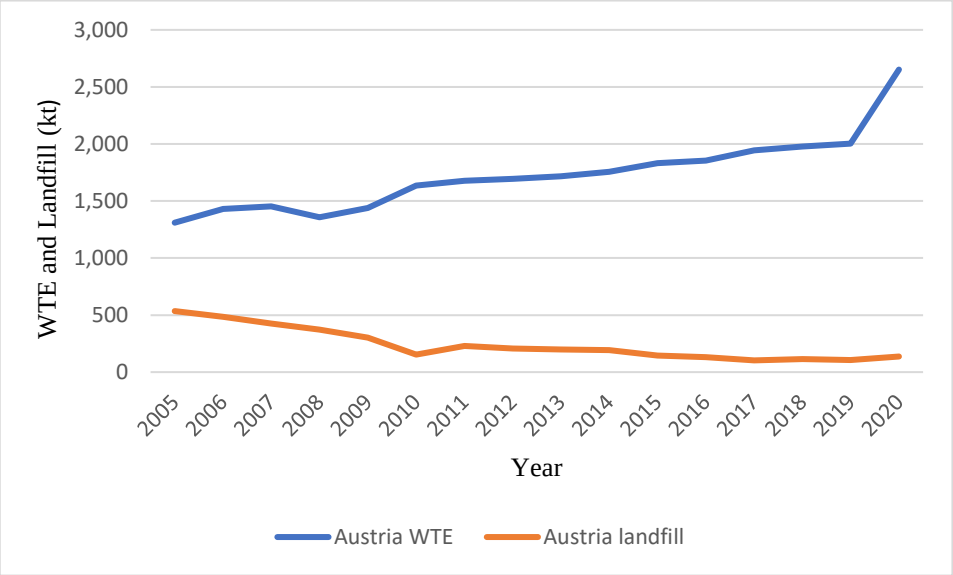

Figure 15. Amount of WTE and landfill in Austria (Eurostat - Data Explorer, 2022)

The trends of changes in WTE treatment and MSW generation have been rising slowly from 2005 to 2020. This country has a relatively low number of landfills, but it has decreased gradually in comparison with MSW generation during the same period. Hence, the percentage of WTE based on the MSW generated (A_1) in 2005 was 28 % and it (A_2) grew to 36 % in 2020. It should be mentioned that over the 15 years the changes of WTE based on MSW generated was 8 %-points. Hereupon, the *SWC* (see Eq. (7)) in each year was 0,5 %-points/a during the transition period. Furthermore, the percentage of landfill based on the MSW generated in 2005 was 11 % (B_1) and fell to 2 % (B_2) in 2020. Therefore, *SLC* (see EQ. (10)) was a negligible value of -0,6 %-points/a. This trend can be seen in Figure 16.

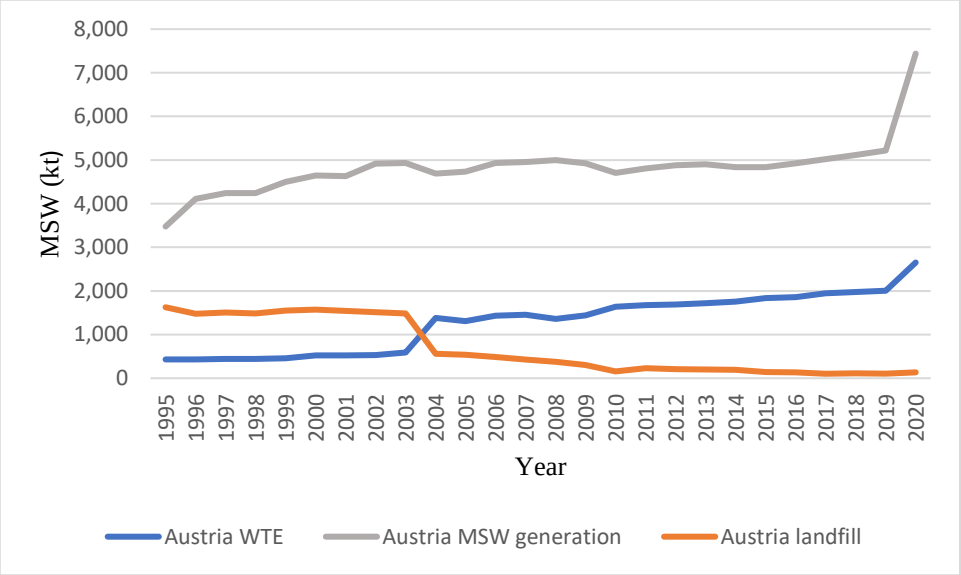

Figure 16. Comparison of MSW generation and WTE and landfill treatment in Austria (Eurostat - Data Explorer, 2022)

4.1.7. Lithuania

In the near future the cogeneration power plant is anticipated to utilize almost 160 kt of non-recyclable MSW that would have otherwise gone to landfills in Lithuania. This utilization of MSW will provide heat to 20 % of Vilnius and electricity to 90,000 households. It will save the city's residents approximately 10 million EUR annually in waste disposal fees. In addition, the WTE facility will help to increase environmental sustainability by reducing the amount of landfill space needed by about 95 %. It is estimated that the plant will decrease national GHG emissions from the waste sector by 10 %, or about 130 kt of carbon dioxide each year (Ec.europa.eu, 2019).

The amount of WTE treatment in Lithuania was almost zero until 2012. In 2013, the mass of MSW treated by WTE method was 92 kt/a, Lithuania was able to increase this amount to 349 kt/a in 2020. This was not the only achievement in MSW management in this country, also it was able to reduce the amount of landfill in the same period from 971 kt/a to 220 kt/a in 2020. The trend of WTE and landfill treatment during the transition period is shown in Figure 17. Lithuania, like Ireland and Poland, has made a big leap in a short period of time, so that in the selected seven-year period, its overall speed of WTE transition (*SWT*) was 40 %/a, and its percentage of development was 279 %, which are significant values. Also, to

calculate the speed of transition in landfill, the overall speed of landfill transition (*SLT*) was -10 %/a and percentage of landfill reduction was 72 % during the transition period.

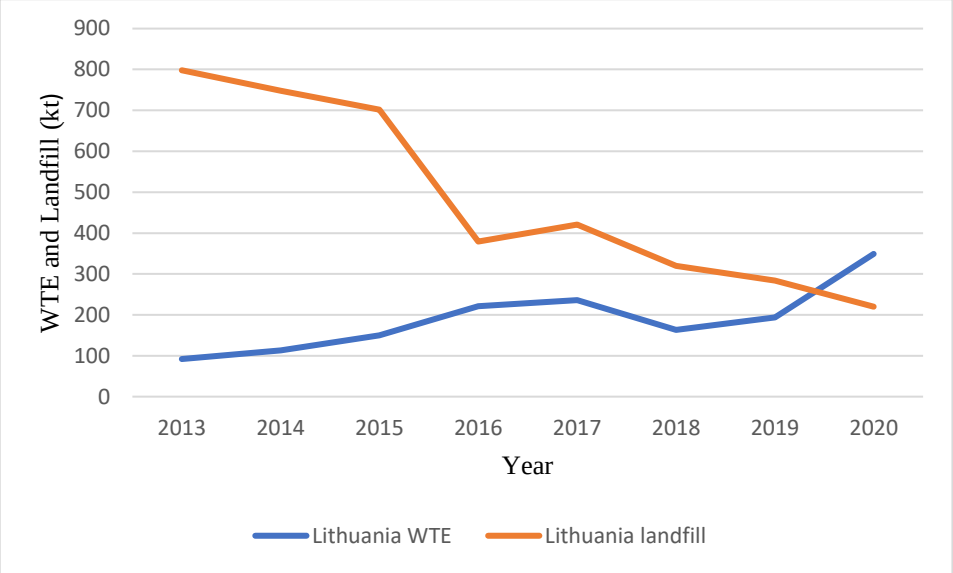

Figure 17. Amount of WTE and landfill in Lithuania (Eurostat - Data Explorer, 2022)

Figure 18 below illustrates that the trend of MSW generation in Lithuania has been almost constant during the determined period, while the trend of WTE treatment has been increasing since 2013. Also, landfill had significant decrease in the same period. Based on the calculation was done to demonstrate the changes of landfill and WTE based on the MSW generated, the percentage of WTE based on the MSW generated (A_1) in 2013 was 7 %, and it (A_2) has been reached to 27 % in 2020. It shows that the percentage of WTE has been increased by 19 %-points in seven years. Hence, *SWC* (see Eq. (7)) was 2,7 %-points/a. On the other hand, the percentage of landfill based on the MSW generated at the beginning of the determined period in 2013 was 62 % (B_1) and it was reduced to 16 % (B_2) in 2020 at the end of transition period. Also, *SLC* (see Eq. (10)) was -6,6 %-points/a.

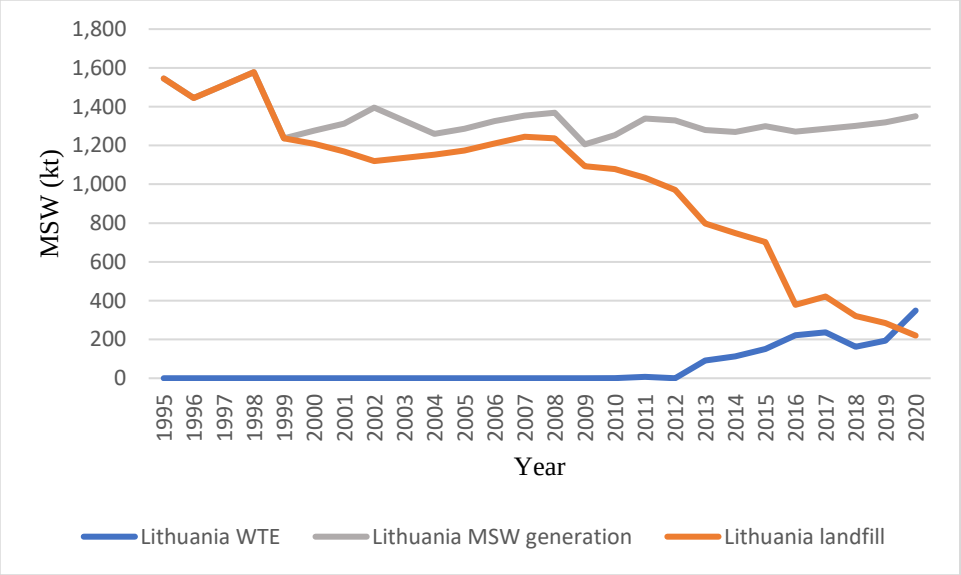

Figure 18. Comparison of MSW generation and WTE and landfill treatment in Lithuania (Eurostat - Data Explorer, 2022)

In the near future the cogeneration power plant is anticipated to utilize almost 160 kt of non-recyclable MSW that would have otherwise gone to landfills in Lithuania. This utilization of MSW will provide heat to 20 % of Vilnius and electricity to 90,000 households. It will save the city's residents approximately 10 million EUR annually in waste disposal fees. In addition, the waste-to-energy facility will help to increase environmental sustainability by reducing the amount of landfill space needed by about 95 %. It is estimated that the plant will decrease national GHG emissions from the waste sector by 10 %, or about 130 kt of carbon dioxide each year (Ec.europa.eu, 2019).

4.1.8. China

In terms of MSW generation and incineration, China is the world's first-largest generator and incinerator (Wei, Li, & Liu, 2022). WTE treatment has been adopted by the central and local governments of China as a core technology for achieving the goal of renewable and sustainable MSW disposal. A series of policies and regulations have been promulgated to facilitate the sustainable implementation of this technology (Cui et al., 2020). The major transition from landfill to WTE in China was between 2010 and 2017, and in this period, the

utilization of WTE technology was focused, so that the number of WTE plants was 109 in 2008 and was reaching 331 in 2018 (National Bureau of Statistics of China, 2021).

This rapid increase in the number of WTE plants led to *SWT* (see Eq. (2)) and the percentage of development in WTE estimated as 39 %/a, and 270 % respectively. It was expected that the transition from landfill to WTE in China would experience a decrease in the amount of landfill. While this country has increased the amount of landfill at the same time as the use of the WTE method has increased, but with a lower slope. Hence, *SLT* (see Eq. (4)) was 4 %/a, and the percentage of reduction in landfill treatment was -25 %. Figure 19 represents the trend of WTE and landfill during the transition period.

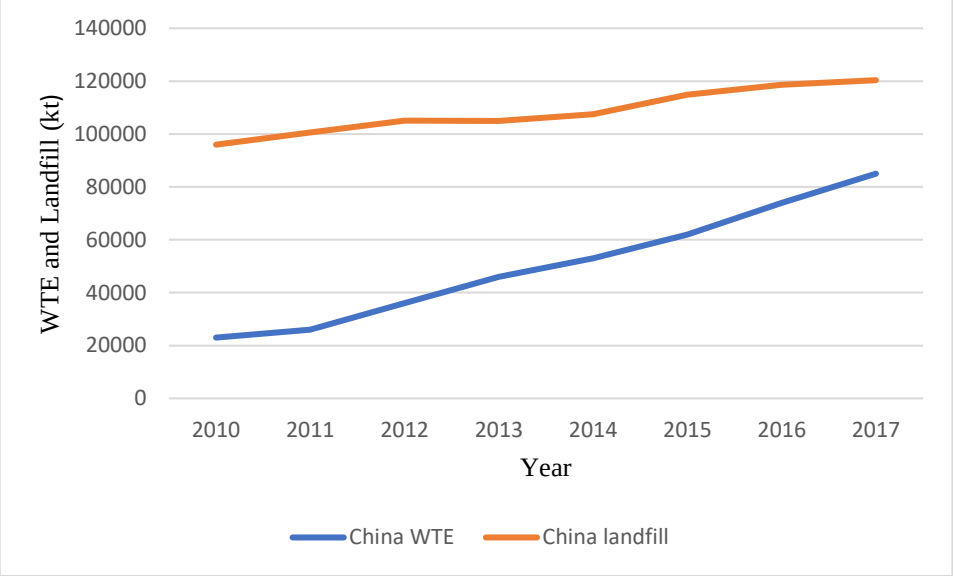

Figure 19. Amount of WTE and landfill in China data from annual statistical yearbooks (MoHURD, 2020)

Considering the increase in MSW treated by WTE and landfill compared to the trend of MSW generation, the calculations indicate that between 2010 and 2017 in China, the percentage of WTE based on MSW generated (A_1) was 15 % in 2011 and it (A_2) has increased to 39 % in 2018. Accordingly, the changes of WTE based on MSW generated was 24 %-points over the seven years and *SWC* (see Eq. (7)) to be 3,8 %-points/a during the transition period. China has increased the overall speed of transition of landfill, however the changes in landfill treatment based on MSW generation have been decreasing. In 2010, at the

beginning of the transition period, the percentage of landfills based on MSW generated (B_1) was 61 %, it (B_2) decreased to 56 % in 2017 at the end of the period. Generally, the calculations indicate that *SLC* (see Eq. (10)) was -0,69 %-points/a during the transition period.

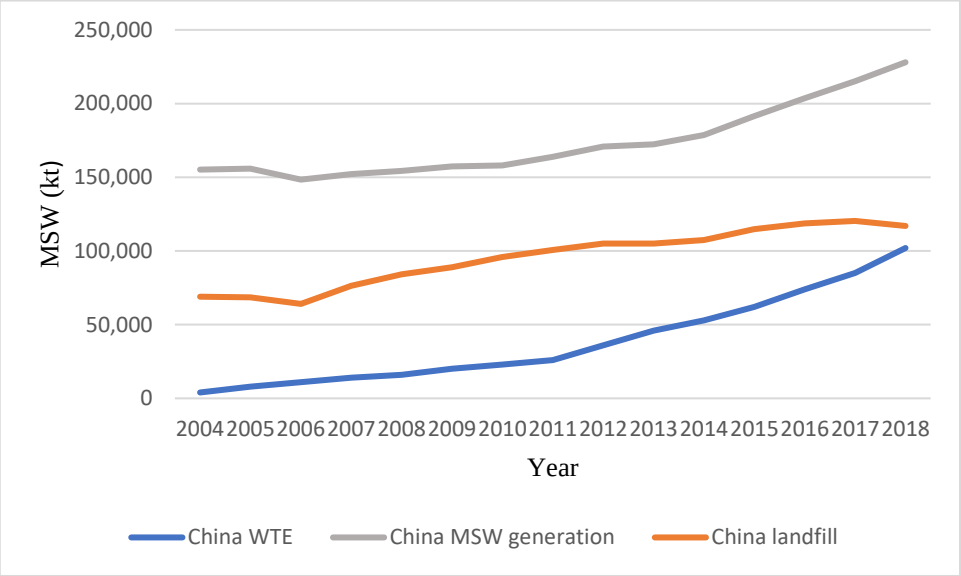

Figure 20. Comparison of MSW generation and WTE and landfill treatment in China (MoHURD, 2020)

4.1.9. Comparing the case countries by speed of transition

Since one of the aims of this study is to evaluate the speed of transition from landfill to WTE, the transition period with the most significant change for each country was determined. Then, the overall speed of WTE transition (*SWT*), the percentage of development in WTE, the percentage of WTE based on MSW generation at the start point and end point of the period (A_1 and A_2) for each country has calculated and finally the indicator of *SWC* which shows the speed of WTE changes based on MSW generation in each year during the transition period has been determined. On the other hand, during the transition from landfill to WTE, the amount of landfill has significantly reduced in most of the selected countries. Therefore, calculating the speed of landfill transition is necessary to evaluate in this study. Also, the overall speed of landfill transition (*SLT*), the percentage of landfill reduction, the

percentage of landfill based on MSW generated at the start point and end point of the period (B_1 and B_2) for each country have calculated. Finally, as a last step in this section, *SLC* which demonstrates the speed of landfill changes based on MSW generated in each year during the transition period has concluded. A summary of the results obtained from the calculation of the speed of transition from landfill to WTE is shown in Table 10 Table 11.

Table 10. Results of speed of WTE transition

Countries	Time period	Number of years	Percentage of development in WTE (%)	SWT (%/a)	A_1 (%)	A_2 (%)	$A_2 - A_1$ (%-points)	SWC (%-points/a)
Ireland	2008 to 2014	6	990	165	3	34	31	5,26
United Kingdom	2005 to 2018	13	330	25	8	41	33	2,51
Norway	2006 to 2013	7	115	16	32	57	25	3,70
Finland	2010 to 2019	9	212	24	22	56	34	3,72
Poland	2012 to 2020	8	5400	680	0,42	22	21	2,64
Austria	2005 to 2020	15	102	7	28	36	8	0,53
Lithuania	2013 to 2020	7	279	40	7	26	19	2,67
China	2010 to 2017	7	270	39	15	39	24	3,56

- *SWT*: The overall speed of WTE transition,
- A_1: The percentage of WTE based on MSW generation at the start point of the period,
- A_2: The percentage of WTE based on MSW generation at and end point of the period,
- *SWC*: The speed of WTE changes.

Table 11. Results of speed of landfill transition

Countries	Time period	Number of years	Percentage of landfill reduction (%)	SLT (%/a)	B_1 (%)	B_2 (%)	$B_2 - B_1$ (%-points)	SLC (%-points/a)

Ireland	2008 to 2014	6	72	-12	60	21	-40	-6,63
United Kingdom	2005 to 2018	13	80	-6	64	15	-49	-3,79
Norway	2006 to 2013	7	87	-12	18	2	-16	-2,31
Finland	2010 to 2019	9	97	-11	45	1	-44	-4,90
Poland	2012 to 2020	8	35	-4	67	40	-27	-3,39
Austria	2005 to 2020	15	74	-5	11	2	-9	-0,63
Lithuania	2013 to 2020	7	72	-10	62	16	-46	-6,58
China	2010 to 2017	7	-25	4	61	56	-5	-0,69

- *SLT*: The overall speed of landfill transition,
- B_1: The percentage of landfill based on MSW generation at the start point of the period,
- B_2: The percentage of landfill based on MSW generation at and end point of the period,
- *SLC*: The speed of landfill changes.

Since the selected countries have experienced significant changes in the transition from landfill to WTE over varying periods of time, one of the purposes of this study is to compare the speed of the transition in the case countries. Therefore, to be able to compare the countries with each other, *SWC* and *SLC* (see Eq. (710 and 10)) are considered indicators to compare the selected countries with each other. With these indicators, it is possible to demonstrate the annual speed of increase or decrease of WTE and landfill based on MSW generation during the transition period. In other word, these indicators demonstrate how many percentages of MSW treated by WTE and landfill have decreased or increased annually based on per kt of MSW generated during the transition period.

Upon comparing the results of the speed of waste-to-energy (WTE) changes (SWC) for the selected countries (as presented in **Table *10***), it is evident that Ireland had the highest SWC during its transition period. This finding suggests that Ireland was able to make significant progress in transitioning from traditional landfill practices to the more sustainable WTE

method, at a relatively fast pace, compared to the other selected countries. Also, results indicate that *SWC* is 5,26 %-points/a In Ireland. After that, Finland, Norway, and China had the highest *SWC* during their transition period, which are 3,72 %-points/a, 3,70 %-points/a, and 3,56 %-points/a respectively. It should be noted that in the Finland the percentage of WTE treatment based on MSW generated has increased by 34 %-points during its specified period. This amount for China and Norway was 24 %-points and 26%-points respectively during their transition period. The three countries of Lithuania, Poland and the UK have experienced moderate WTE changes during their transition period compared to other countries. *SWC* for these countries is 2,67 %-points/a, 2,64 %-points/a, and 2,5 %-points/a respectively. Finally, compared to other countries, Austria had the lowest annual speed of change in WTE based on MSW generated, which is 0,53 %-points/a during its transition period. Figure 21 illustrates the comparison of *SWC* in all case countries.

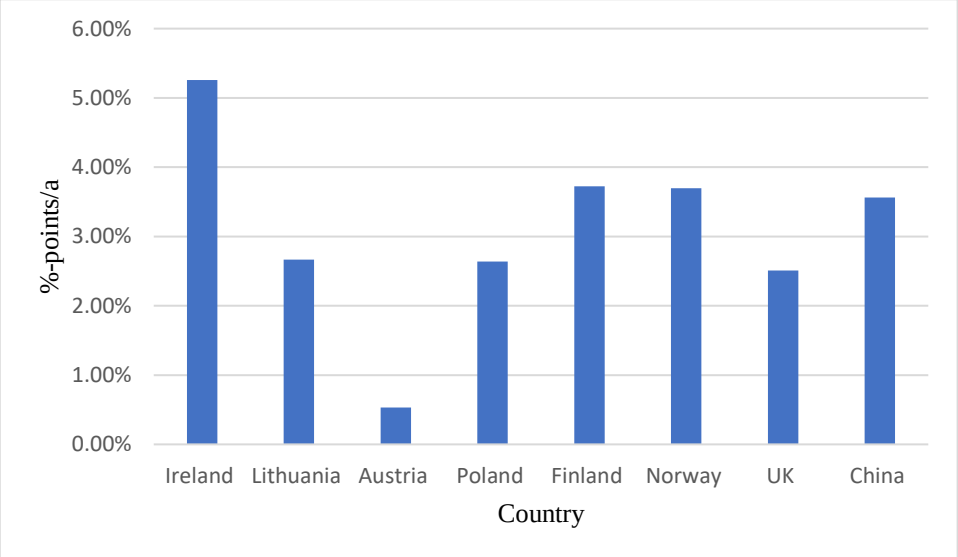

Figure 21. Comparison of SWC in each country

SLC is defined as an indicator of the speed of landfill changes in a country (see Eq. (10)). In other words, this indicator shows the annual reduction percentage of the landfill based on MSW generated. Hence, according to the results represent in Table 11, two countries, Ireland and Lithuania, had the highest percentage of annual reduction of MSW treated by landfill method based on MSW generated (*SLC*), which were -6,63 %-points/a and -6,54 %-points/a respectively during their transition period. After that, Finland had the most *SLC* during its

transition period which was -4,90 %-points/a. *SLC* values for the UK, Poland, and Norway were in an average range, corresponding to -3,79 %-points/a, -3,39 %-points/a, and -2,31 %-points/a, respectively. China and Austria had the lowest *SLC* values during their transition periods, with values of -0,69 %-points/a, and -0,63 %-points/a respectively. Figure 22 demonstrates the comparison of *SLC* in all case countries

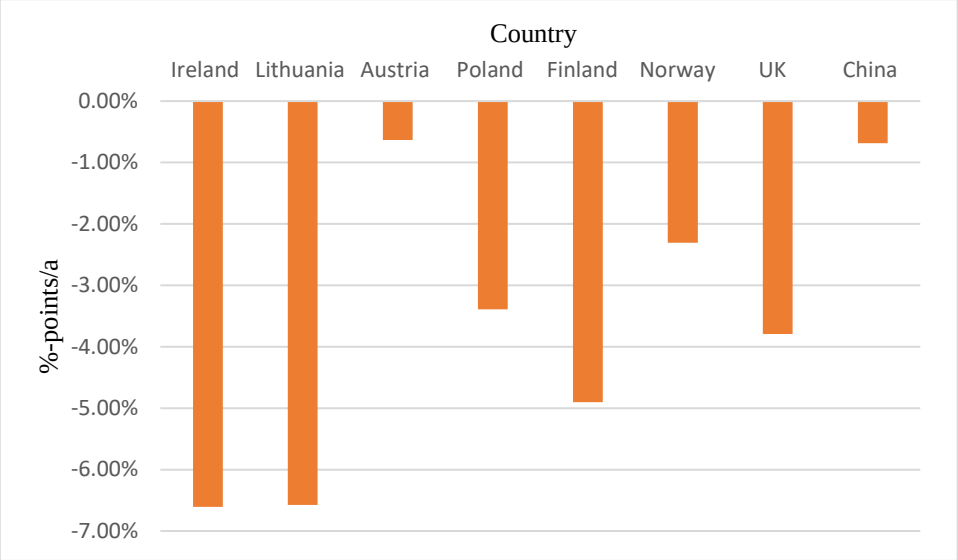

Figure 22. Comparison of SLC in all case countries

By comparing Figures 21 and 22 as well as the obtained results, it can be concluded that the highest speed of transition from landfill to WTE in a certain period was in Ireland. After that, Finland showed satisfactory results in this transition. This country was able to significantly reduce the amount of landfill treatment and on the other hand, increase the treatment of WTE. In this regard, the countries of the UK, Poland, and Norway have made important achievements in the transition from landfill to WTE. These countries were able to reduce landfills at an acceptable rate and increase the treatment of WTE in their industry at a high rate. China was also able to increase WTE treatment very quickly. This is even though China does not have a significant decrease in the landfill. Even in the transition period an increase in the amount of landfilling was observed. But the amount of landfill was decreasing based on each kt of MSW generated. Among all the countries studied, Austria had the slowest transition from landfill to WTE. This is due to the fact that MSW treatment by landfill and WTE methods has the least contribution to MSW management in Austria. This is due to the increased recycling capacity in this country, and Austria has the highest

recycling rate among European countries with 65 % (Highest recycling rates in Austria and Germany - EEA, 2020).

4.2. Rate of changes in GHG emissions

In this section, the overall changes in GHG emissions caused by WTE and landfill treatment examined during a transition period in each country. Also, this changes in GHG emissions for each treatment methods (landfill and WTE) analysed separately. Hence, it can be determined how effective each treatment method (landfill and WTE) has been in the overall changes in GHG emissions caused by landfill and WTE treatment. Two general concepts have expressed in this section, which are the overall changes in GHG emissions caused by landfills and WTE treatments, and the annual rate of change in GHG emissions caused by landfills and WTE treatments. The overall changes in GHG emissions generated by landfill and WTE treatments (*OCG*) was calculated by using Eq. (14) and the annual rate of change in GHG emissions generated by landfill and WTE treatments (*RCG*) was calculated by using Eq. (17). Then, *OCG* and *RCG* were calculated separately for landfill and WTE treatment using Eq. (15), (16), (18), and (19). A more detailed analysis of GHG emissions from landfill and WTE treatment has done to compare the case countries, which will be discussed further.

4.2.1. Analysing the results of Rate of changes in GHG emissions

First, it was necessary to determine how much GHG emissions were generated by WTE and landfill treatment to evaluate the changes in GHG emissions. Therefore, the amount of GHG emissions caused by WTE was calculated using the IPCC method at the beginning and end of the transition period for each country (see Eq. (11) and (12)). Also, by relying on Eurostat-statistics (2022) and Bian et al. (2022), the amount of GHG emissions caused by landfill treatment was adopted in each country. It should be noted that the amount of GHG emissions from landfill treatments was determined by the accumulated mass of the landfill in the past years. Table 3 and Table ***12*** show the amount of GHG emissions from landfill and WTE.

Table 12. Amount of GHG emissions from WTE treatments

Countries	Time period	WTE GHG emissions (kt CO_2.eq)	
		start time	end time
Ireland	2008 to 2014	27	273
Lithuania	2013 to 2020	27	135
Austria	2005 to 2020	564	1142
Poland	2012 to 2020	19	1073
Finland	2010 to 2019	285	891
Norway	2006 to 2013	157	337
United Kingdom	2005 to 2018	962	3859
China	2010 to 2017	9360	41500

The calculation of annual rate of GHG emissions (*RCG*) and overall changes in GHG emissions (*OCG*) can be both positive and negative. Positive reports represent the increase in GHG emissions during the transition period, and on the contrary, negative reports demonstrate the decrease in GHG emissions.

According to Table 13 below, among the European countries and China that have made fundamental changes in the MSW management sector and have experienced a significant increase in the incineration sector during the specified period, compared to other countries, the UK had the biggest overall GHG emissions changes (*OCG*) from WTE and landfill treatment, which was -60 %. Also, the annual rate of GHG emissions change (*RGE*) in this country was -4,58 %/a during its transition period. Also, the changes in GHG emissions caused by landfill treatment (OCG_L and RGE_L) in this country were -68 % and -5,19 %/a respectively. While OCG_W and RGE_W have increased to 301 %, 23,16 %/a during its transition period. These results show that the reduction of GHG emissions caused by landfill treatment has a greater contribution to the overall changes in GHG emissions because landfill GHG emissions are much higher than WTE. The percentage of overall GHG emissions changes (*OCG*) from WTE and landfill treatment and the percentage of the annual rate of

GHG emissions changes (*RGE*) for the three countries of Austria, Lithuania, and Poland are in the same range. Consequently, the *OCG* for Austria, Poland, and Lithuania was -34 %, -22 %, and -19 % respectively, while the RGE for these three countries was -2,29 %/a, -3,21 %/a, and -2,41 %/a. In addition, by observing the OCG_W, OCG_L, RGE_W and RGE_L parameters in these three countries, it is found that the reduction of GHG emissions caused by landfill treatment was the main factor to reduce the overall changes of GHG emissions. However, GHG emissions by WTE treatments was increased.

Table 13. Comparing the results of GHG emissions rate

Contries	Time period	Number of years	*OCG* (%)	*RGE* (%/a)	OCG_L (%)	RGE_L (%/a)	OCG_W (%)	RGE_W (%/a)
Ireland	2008 to 2014	6	88	14,63	40	6,62	919	153,13
United Kingdom	2005 to 2018	13	-60	-4,58	-68	-5,19	301	23,16
Norway	2006 to 2013	7	-1	-0,14	-14	-2,0	114	16,32
Finland	2010 to 2019	9	-6	-0,62	-34	-3,81	212	23,58
Poland	2012 to 2020	8	-19	-2,41	-25	-3,16	90	11,29
Austria	2005 to 2020	15	-34	-2,29	-66	-4,39	102	6,83
Lithuania	2013 to 2020	7	-22	-3,21	-36	-5,08	404	57,77
China	2010 to 2017	7	87	12,38	58	8,35	344	49,10

OCG: overall changes in GHG emissions from landfill and WTE during the transition period

OCG_L: overall changes in GHG emissions from landfill during the transition period

OCG_W: overall changes in GHG emissions from WTE during the transition period

RGE: annual rate of change in GHG emissions in WTE and landfill

RGE_L: annual rate of change in GHG emissions in landfill

RGE_W: annual rate of change in GHG emissions in WTE

As shown in Table 13, Finland and Norway have the lowest percentage reduction of overall GHG emissions changes caused by WTE and landfill treatment. In Finland and Norway, the percentage of overall GHG emissions changes (*OCG*) was -6 % and -1 % respectively and the annual rate of GHG emission changes (*RGE*) was -0,62 %/a, and -0,14 %/a respectively. In contrast, two countries, China and Ireland, showed in their results, the percentage of overall GHG emissions changes (*OCG*) from WTE and landfill treatment, as well as the percentage of annual GHG emission changes (*RGE*) increased during their transition period. Therefore, *OCG* in Ireland was 88 % and *RGE* was 14,63 %/a, similarly, in China, *OCG* was 87 % and *RGE* was 12,38 %/a. In addition, by observing the results of Table 13 all parameters such as OCG_W, OCG_L, RGE_W and RGE_L were incremental. It will be discussed in more detail how WTE and landfills increase GHG emissions in these two countries. In summary, these results demonstrated only a trend in the percentage change in GHG emissions during the transition period as well as throughout each year. Furthermore, these parameters cannot be a suitable criterion for comparing countries. The analysis of more countries and their comparison will be discussed further.

4.2.2. Comparing case countries in GHG emissions

In order to compare countries among themselves, this study aimed to develop indicators that can be used to examine and compare the changes in GHG emissions caused by landfilling and WTE treatment in different countries. Then, using Eq. (20) the sum of GHG emissions from landfilling and WTE is divided by the sum of landfilled and incinerated masses of MSW. Therefore, Eq. (20) demonstrates how much GHG emissions were emitted per kt of MSW treated by landfill and the WTE method in each year. Also, Eq. (21) represents the changes in GHG emissions during a transition period, which can be used to compare countries with each other.

Table 14. Amount of GHG emissions based on the per kt of landfilled and incinerated MSW

Contries	Time period	Number of years	G_1 ($kt_{CO2eq}/kt_{(landfill+WTE)}$)	G_2 ($kt_{CO2eq}/kt_{(landfill+WTE)}$)	ΔG ($kt_{CO2eq}/kt_{(landfill+WTE)}$)

Ireland	2008 to 2014	6	0,243	0,644	0,401
United Kingdom	2005 to 2018	13	1,752	1,049	-0,704
Norway	2006 to 2013	7	1,455	0,531	-0,924
Finland	2009 to 2020	11	1,451	0,719	-0,731
Poland	2012 to 2020	8	1,243	1,072	-0,171
Austria	2005 to 2020	15	1,627	0,707	-0,920
Lithuania	2013 to 2020	7	1,012	1,227	0,215
China	2010 to 2017	7	0,795	0,794	-0,001

G_1: The amount of GHG emissions from WTE and landfill divided by total mass of landfill and WTE at the start point of transition period (kt CO_2eq/kt (landfill +WTE))

G_2: The amount of GHG emissions from WTE and landfill divided by total mass of landfill and WTE at the end point of transition period (kt CO_2eq/kt (landfill +WTE))

ΔG: Changes in the amount of GHG emissions (kt CO_2eq/kt (landfill +WTE)) divided by total mass of landfill and WTE during transition period

In this study, changes in GHG emissions based on per kt of MSW treated by landfill and WTE (ΔG) were investigated. Logically ΔG (see Eq. (21)) should be decreasing (negative), because that is one of the arguments for WTE, that it reduces GHG emissions compared to landfilling. However, the two countries of Ireland and Lithuania had unexpected results. Table 14 shows the amount of GHG emissions based on per kt of MSW treated by landfill and WTE (G_1 and G_2) during transition a period of each country.

The changes of GHG emissions based on per kt of MSW treated by landfills and WTE (ΔG) increased in Ireland and Lithuania during their transition period. These results are unexpected, at least for Ireland, because it was observed in section 4.1.9 56that Ireland has the fastest speed of transition from landfill to WTE. Therefore, a more detailed analysis will be done in these two countries. In Ireland G_1 in 2008 was 0,243 kt $_{CO2eq}$/kt $_{(landfill +WTE)}$ and G_2 increased to 0,644 kt $_{CO2eq}$/kt $_{(landfill +WTE)}$ at the end of the transition period. It means that the changes of sum of GHG emissions from landfilling and WTE divided by the sum of landfilled and incinerated masses of MSW (ΔG) increased by 0,442 kt $_{CO2eq}$/kt $_{(landfill +WTE)}$ during the transition period. For further analysing, the total GHG emissions from WTE was

divided by the total mass of incinerated MSW in each year (*GI* see Eq. (22)), so that the amount of GHG emissions caused by each kt of incinerated MSW can be calculated. Table 15 below shows the results of the mentioned calculations, with the analysis of the results It is found that GHG caused by WTE was decreased during the transition period. In fact, GI_1 was 0,327 kt $_{CO2eq}$/kt $_{(WTE)}$ in 2008 and GI_2 decreased to 0,306 kt $_{CO2eq}$/kt $_{(WTE)}$ at the end of the period. Therefore, it can be concluded that the reason for increase in ΔG was not the emission changes in WTE treatment. Also, by evaluation of this country, it was found that the reason for the decrease in *GI* was the change in the composition of MSW and the reduction of non-biogenic carbon in the incineration process. In order to find the reason for increase in ΔG, extensive studies were conducted. Finally, the European Environmental Protection Agency (EPA) has made a report for Ireland in 2022, which states that the increase in methane emission trends from landfills from 2009 to 2016 is due to a decrease in landfill gas flaring as overall landfill gas volumes decline. Since the amount of GHG emissions caused by landfilling was taken from Eurostat- statistic (2022), it is also considered that GHG emissions reported are based on the accumulated mass of landfills in the past years. Therefore, this study decided to calculate the amount of GHG emissions caused by landfill at the beginning and end of the transition period. Using Eq. (13), the potential of GHG emissions caused by the amount of landfilled MSW per year was calculated. It should be noted that this method was introduced by IPCC (2006) and it produces fairly good estimates of the yearly emissions. This means that it estimates the potential of the total landfilled MSW that can produce GHG emissions. Therefore, the results demonstrate that, according to the amount of landfilled MSW in 2008, the potential of producing GHG emissions was 4060 kt/a, which decrease to 1125 kt/a at the end of the transition period. This number is the total production potential of GHG emissions caused by the landfill, which is released gradually over the years. In this case, it can be concluded that the amount of GHG emissions caused by the landfill treatment has also decreased, while the cumulative GHG emissions, which was affected by the landfill in the past years, has increased during the transition period. In other words, the increase in changes of GHG emissions in per kt of MSW treated by landfill and WTE methods (ΔG) is due to the increase in landfill emissions, which is affected by landfill emissions in previous years. The reason for this can be seen as the lack of management and control of GHG emissions caused by landfills in the past years.

Lithuania was also subjected to a detailed analysis as a result of incremental changes in GHG emissions in per kt of MSW treated by landfill and WTE methods (ΔG). ΔG in this

country was 0,215 kt $_{CO2eq}$/kt $_{(landfill +WTE)}$ during its transition period, while G_1 was 1,012 kt $_{CO2eq}$/kt $_{(landfill +WTE)}$ in 2013 and G_2 increased to 1,227 kt $_{CO2eq}$/kt $_{(landfill +WTE)}$ at the end of the transition period (**Table *14***).

Also, for a more detailed analysis, the amount of GHG emissions generated by incinerated MSW was divided by the total mass of incinerated MSW in each year (Eq. (23)) to determine the trend of changes in GHG emissions caused by WTE (*GI*) (**Table *15***). The results of this analysis show that GI_1 was 0,291 kt $_{CO2eq}$/$k_{t\ (WTE)}$ in 2013 and GI_2 increased to 0,387 kt $_{CO2eq}$/kt $_{(WTE)}$ in 2020. Therefore, *GI* increased in this country during the transition period. Following the reason for this increase in *GI*, this study found that during the transition period in Lithuania, the amount of non-biogenic carbon in the fuel of incinerators increased. So that the percentage of plastic in the composition of waste has increased from 9 % to 12 %. It should be noted that this increase in GHG emissions changes in WTE was not so great that it could be regarded as the primary cause of the increase in overall GHG emission changes (ΔG). Eventually, this study decided to calculate the potential of producing GHG emissions from landfill in each year of the transition period (Eq. (13)). The results showed that the generation potential of GHG emissions caused by landfill mass in 2013 was 1750 kt/a, and in 2020, due to the mass of landfilled MSW, this amount of GHG emissions will decrease to 480 kt/a. Since the GHG emissions reported on the Eurostat-statistic (2022) is based on the cumulative mass of the landfill in the past years also the GHG emissions from landfill treatments in the past years was also considered in this data. Therefore, it can draw a conclusion that the reason for the increase in the overall changes in GHG emissions (ΔG) can be the lack of proper LFG collection in this country. Also, in this regard, EPA published a report in 2013, and stated the amount of LFG collection in Lithuania is not transparent.

On the other hand, the rest of the case studies have had a decreasing trend in changes of GHG emissions in per kt of MSW treated by landfill and WTE methods (ΔG). This decreasing trend indicates that the approach of using WTE and recycling technologies over time has replaced the landfill, which has the highest amount of GHG emissions. The countries of Norway, Austria, Finland, and UK have achieved appropriate decrease in GHG emission changes (ΔG) during their transition period, and this decrease was between the range of -0,704 kt $_{CO2eq}$/kt $_{(landfill +WTE)}$ and -0,924 kt $_{CO2eq}$/kt $_{(landfill +WTE)}$. Meanwhile, two countries, Poland and China, demonstrated minor changes in their transition period. In Poland, the amount of GHG emissions from each kt of MSW treated by WTE and landfill

(G_1) was 1,243 kt $_{CO2eq}$/kt $_{(landfill +WTE)}$ in 2012 and (G_2) decreased to 1,072 kt $_{CO2eq}$/kt $_{(landfill +WTE)}$ in 2020. Accordingly, GHG emission changes (ΔG) for this country during its transition period was -0,171 kt $_{CO2eq}$/kt $_{(landfill +WTE)}$. While the changes in GHG emissions (ΔG) in China were negligible, which was -0,001 kt $_{CO2eq}$/kt $_{(landfill +WTE)}$ between 2010 and 2017. Upon reviewing the results obtained from these countries, it is concluded that the increasing use of WTE technology and reducing the landfilling process has led to a significant reduction in the levels of GHG emissions in these countries.

Table 15. Amount of GHG emissions based on the per kt of incinerated MSW (WTE GHG emissions factor)

Countries	Time period	Number of years	GI_1 (kt CO2eq/kt $_{WTE}$)	GI_2 (kt CO2eq/kt $_{WTE}$)
Ireland	2008 to 2014	6	0,327	0,306
United Kingdom	2005 to 2018	13	0,327	0,306
Norway	2006 to 2013	7	0,233	0,233
Finland	2010 to 2019	9	0,513	0,513
Poland	2012 to 2020	8	0,380	0,380
Austria	2005 to 2020	15	0,431	0,431
Lithuania	2013 to 2020	7	0,291	0,387
China	2010 to 2017	7	0,407	0,407

GI_1: The amount of GHG emissions of WTE (kt CO_2eq/kt $_{WTE}$) divided by total mass of WTE at the start point of transition period

GI_2: The amount of GHG emissions of WTE (kt CO_2eq/kt $_{WTE}$) divided by total mass of WTE at the end point of transition period

Table 15 presents data on the amount of GHG emissions generated by each kt of incinerated MSW in the selected countries. It is important to note that in the calculations of GHG emissions caused by WTE in the countries of the UK, Ireland, and Lithuania, the composition of waste for the beginning and end of the period is different. In contrast, for the remaining countries, the composition of waste is assumed to be the same for the entire period since such data was not available. In addition to the data presented in Table 15, the present

study also obtained results on the GHG emissions factor for WTE treatment in each of the selected countries. The GHG emissions factor represents the amount of GHG emissions generated per unit of waste incinerated by the WTE process, and it provides valuable insights into the environmental impact of this waste management approach. According to Table 15, Finland, Austria, and China have the highest WTE GHG emission factors. Based on an analysis of MSW composition in these countries, it was discovered that the amount of non-biogenic carbon is higher than in other countries. There is a possibility that this is the main reason for the high *GI* in these countries.

4.3. Compiling dioxin emissions from landfill fire and WTE plants

According to investigations of Liamsanguan & Gheewala (2007), human exposure to dioxin emissions can result in health issues such blood disorders, skin cancer, heart, liver, and kidney diseases. Considering these harmful impacts of dioxin emissions and the fact that WTE technology and landfill fire emit a significant amount of dioxin, this study aimed to use Eq. (23) to calculate the amount of dioxin emission caused by WTE for each country during their transition period.

In this study, official data were not available to calculate the amount of dioxin emissions caused by the landfill fire. Therefore, part of the initial data was assumed based on the articles that conducted dioxin calculations. Dioxin emissions caused by landfill fires were estimated using Eq. (24) to (26) for all countries during their transition period.

The present study aims to examine the observations during the transition period of waste management practices, focusing on the estimation of dioxin emissions from landfill fire and WTE methods. In Table 16, the total dioxin emissions from landfill fires and WTE at the beginning and end of the transition period are presented. Additionally, using Eq. (27), the amount of dioxin emissions based on per kt of MSW treated by WTE and landfill methods was estimated.

Table 16. Total amount of dioxin emissions from WTE and landfill fire

Countries	Time period	*DEI+DEL* (g-TEQ)	*DEI+DEL* (g-TEQ)
		Start point	End point
Ireland	2008 to 2014	25,40	7,08
United Kingdom	2005 to 2018	295,56	60,94
Norway	2006 to 2013	5,13	0,74
Finland	2010 to 2019	15,05	0,95
Poland	2012 to 2020	105,84	68,45
Austria	2005 to 2020	7,06	1,91
Lithuania	2013 to 2020	10,45	2,90
China	2010 to 2017	1260,33	1590,17

DEI: Total amount of dioxin emissions from WTE

DEL: Total amount of dioxin emissions from landfill fire

The data presented in Table 17 provides valuable insights into the changes in dioxin emissions based on the amount of MSW treated by WTE and landfill. Notably, since all the selected countries experienced a significant increase in WTE and a decrease in landfill, the reduction in dioxin emissions in all countries is attributed to the increase in WTE. The findings highlight the potential for WTE to contribute to reducing dioxin emissions and enhancing environmental sustainability by providing a more sustainable and innovative approach to waste management practices.

Table 17. Changes in dioxin emission in each country

Contries	Time period	Number of years	*D1* (mg-TEQ/kt landfill+WTE)	*D2* (mg-TEQ/kt landfill+WTE)	*ΔD* (mg-TEQ/kt landfill+WTE)
Ireland	2008 to 2014	6	12,56	4,95	-7,61
United Kingdom	2005 to 2018	13	11,59	3,54	-8,05
Norway	2006 to 2013	7	4,82	0,50	-4,32
Finland	2010 to 2019	11	8,89	0,54	-8,36
Poland	2012 to 2020	8	13,01	8,51	-4,50
Austria	2005 to 2020	15	3,83	0,68	-3,14
Lithuania	2013 to 2020	7	11,74	5,09	-6,65
China	2010 to 2017	7	10,59	7,74	-2,85

D_1: Total amount of dioxin emissions (mg-TEQ/kt $_{landfill+WTE}$) from landfill fire and WTE at the start point of transition period.

D_2: Total amount of dioxin emissions (mg-TEQ/kt $_{landfill+WTE}$) from landfill fire and WTE at the end point of transition period.

ΔD: Changes in the amount of dioxin emissions (mg-TEQ/kt $_{landfill+WTE}$) from landfill fire and WTE during transition period

The emissions reported by international organizations are often aggregated without even the most basic segmentation by sector. Furthermore, in the past, there was no requirement to report background data on relevant activities, which makes it challenging to obtain information on dioxin emissions. As a result, there is a scarcity of data that would help in gaining insight into a nation's specific circumstances regarding dioxin emissions and how to reduce them. Hence, Comparing the development of countries in reducing dioxin emissions is a difficult task (Paradiž & Dilara, 2003).

Nevertheless, as shown in **Error! Reference source not found.** and **Error! Reference source not found.**, Finland has had the highest changes in the amount of dioxin emissions, so that it has reached from 8,89 mg-TEQ/$kt_{landfill+WTE}$ in 2013 to 0,54 mg-TEQ/$kt_{landfill+WTE}$ in 2020 at the end of its transition period. As a result, changes in dioxin emission (*ΔD*) for Finland is -8,36 mg-TEQ/$kt_{landfill+WTE}$ over the 11 years, followed by Ireland and the UK achieved *ΔD* of -7,61mg-TEQ/$kt_{landfill+WTE}$ and -8,05 mg-TEQ/$kt_{landfill+WTE}$, respectively during their transition period.

Furthermore, the remaining *ΔD* values for the other countries fall between -6.65 mg-TEQ/$kt_{landfill+WTE}$ and -2.85 mg-TEQ/$kt_{landfill+WTE}$. Of note, China and Austria have experienced the lowest changes in dioxin emissions. In Austria, as demonstrated in **Error! Reference source not found.**, dioxin emissions were 3.83 mg-TEQ/$kt_{landfill+WTE}$ in 2005 and declined to 0.68 mg-TEQ/$kt_{landfill+WTE}$ by the end of the transition period. The analysis indicates that dioxin emissions remained at low levels during the transition period in Austria. In contrast, according to Cheng & Hu (2010b) study, the emission rates of dioxins from MSW incineration in China are considerably higher compared to those in developed countries, partially due to the disparity in incineration and flue gas cleaning technologies. Additionally, the existing management policies and practices in China contribute significantly to this issue. Therefore, the changes in dioxin emissions (*ΔD*) in China during its transition period were the lowest, with a value of -2.85 mg-TEQ/$kt_{landfill+WTE}$.

4.4. Overall discussion

In the study, it was observed that all countries were undergoing a rapid transition from landfills to WTE methods. On the other hand, the world is facing energy and environmental impacts crises these days. In the energy sector, WTE technology can play an important role, while the world is trying to benefit from renewable energies. But with the upcoming new crisis (war between Ukraine and Russia), the world, especially Europe, is on another brink of an energy crisis after the 1970s. The invasion of Ukraine by Russia in 2022 has greatly affected the worldwide energy market, causing price instability, supply disruptions, security concerns, and economic instability. This crisis, according to the International Energy

Agency (IEA), is the first one to have a worldwide impact and its effects will be felt for years to come. One potentially positive aspect of higher fossil fuel prices is that they provide strong reasons to accelerate toward sustainable alternatives. However, the need for energy security may drive more investment in renewable fuel projects. Therefore, WTE technology can be effective in the energy sector. Since huge amount of waste is generated by people daily. By managing it in the WTE factory, not only energy is generated, but also a big step is taken in the waste management sector.

On the other hand, from environmental protection points of view, WTE treatment can be a sustainable and appropriate option for waste treatment. The high rate of GHG emissions from landfills, the lack of land for landfilling, and its adverse environmental impacts for many years, all these factors make landfills to be significantly reduced. What makes WTE treatment one of the best alternatives to landfill is its ability to reduce GHG emissions levels in the shortest amount of time. On the other hand, recycling requires high investment cost and after the separation stage, some waste remains to be disposed. It should be noted that one of the potentially positive aspects of WTE is the quick rate of reduction of GHG emissions. Therefore, all selected countries, especially European countries, are increasing the use of the WTE industry to meet the EU target (climate-neutral) by 2050.

Based on the study's objectives in showing how WTE treatment has developed and the factors affecting its development in selected countries, Ireland had the highest speed of transition from landfill to WTE. Several factors contribute to the development of WTE technology in Ireland. One of the important factors in the transition from landfill to WTE in Ireland is Landfill restrictions. To reduce the environmental impact of landfills and comply with EU waste management targets, Ireland has implemented strict regulations on the disposal of waste in landfills, leading to a decrease in the amount of waste going to landfills and an increase in alternative waste management options, including WTE. Also, another important factor can be financial incentives. The Irish government has implemented various financial incentives, such as tax credits, to encourage the development of WTE facilities. The Irish government's proactiveness in providing financial incentives aimed at bolstering the WTE sector stems from its recognition of the significant environmental and economic benefits that such facilities offer. These incentives have provided a boost to the WTE industry in Ireland and made it a more attractive investment opportunity (Howley et al., 2005; Kopetz, 2007; Rajendran et al., 2019).

After Ireland, Finland has an impressive speed of transition from landfill to WTE. One of the most important factors that caused this speed of transition is that Finland has set ambitious sustainability and climate goals, including being the first climate-neutral country by 2035, also reducing GHG emissions and increasing the use of renewable energy sources (Central Government Debt Management – Climate Neutral Finland 2035, 2022). Hence, WTE is one of the important factors to meet its target soon. Also, advances in WTE technology have made it more efficient and environmentally friendly, making it a viable option for managing waste in Finland. The primary impetus behind investments in WTE facilities can be attributed to the 2016 landfill ban on organic waste. This regulatory measure had been anticipated well in advance, as evidenced by the commencement of investments in WTE facilities as early as 2010. Concurrently, the substantial increase in landfill taxes further incentivized these investments by rendering them financially viable (Hupponen et al., 2015; Virtanen et al., 2019).

It is worth noting that the rest of the selected European countries, such as the UK, Norway, and Lithuania, had a similar rate of development of the WTE industry. The reason for this progress in WTE treatment and landfill reduction in these countries is the implementation of EU targets. In this regard, some of the specific provisions of the EU directive that have led to an increase in WTE in these countries include landfill reduction targets, promotion of energy recovery in which the EU directive provides a harmonized regulatory framework for waste management across member states, making it easier for WTE facilities to be developed and operated. Overall, the EU directive on waste management has helped to increase the use of WTE in these European countries as a way to manage waste, generate energy, and meet sustainability and climate goals.

Compared to other countries, Poland had the highest percentage of development in WTE treatment during its transition period. However, the annual speed of the WTE transition was not the highest. This shows that Poland has greatly increased its capacity to use WTE treatment in the last years of the transition period. Also, the recent activities of this country in waste management show that Poland has invested in the WTE industry to provide energy. Undoubtedly, in the coming years, Poland will be one of the leading countries in the use of the WTE industry.

On the other hand, Austria, experienced the slowest transition from landfill to WTE of all European countries. WTE has not been as widely adopted in Austria as in other countries for

several reasons. One of the main reasons is strong emphasis on recycling. Austria has a strong tradition of waste recycling, with a focus on separating recyclable materials and renew them. This has led to a lower emphasis on WTE compared to other waste management methods. Also, the Austrian government has not provided the same level of financial and policy support for WTE as some other countries, making it more challenging for WTE facilities to be developed and operated (Schwarzböck et al., 2017). As a last point, Austria has abundant hydropower and other renewable energy sources that is caused fading out the need for WTE as a source of energy (Austria Electricity Statistics - Worldometer, 2016).

Finally, China also showed that it had experienced significant development in the use of WTE treatment. Seeking to find the reasons for this development, this study found that In China, where the demand for land is increasing due to the expanding population and industries, WTE has emerged as a particularly effective solution in reducing the need for additional landfill areas. The cost of land has become increasingly expensive in the country, making WTE an economically viable alternative for waste management (Zheng et al., 2014). Also, China has faced growing environmental concerns, including air pollution and water pollution. WTE has been seen to reduce emissions and generate energy in a more environmentally responsible manner (Xu et al., 2016).

5. Conclusion

The waste hierarchy and EU policies emphasize recycling over WTE as the preferred method for managing waste. In many countries, politics and regulations favour recycling, with some policies aimed at punishing WTE through taxation. However, given the urgent need to reduce GHG emissions in the EU, WTE can be a viable solution. While recycling is the optimal choice for achieving a circular economy, WTE can be leveraged to quickly reduce GHG emissions. In fact, the need for the development and advancement of MSW treatment with the WTE method is becoming increasingly important, due to the growing amount of MSW generation and the increasing concern for its management to meet the need for a sustainable urban life, as well as the increasing emphasis on non-fossil fuel sources of energy. In this context, WTE technologies have attracted the attention of many countries as a method of MSW treatment that not only burns MSW and extracts energy but also has fewer environmental impacts than landfilling.

Many countries are looking to replace landfilling in their waste management systems due to the inevitable drawbacks associated with it. This study evaluated eight countries that underwent the most fundamental changes in transitioning from landfill to WTE during a specific period. The approach of this study is to examine the speed of the transition from landfill to WTE and the factors affecting it, as well as the results of this transition on the environmental impact. WTE can play a crucial role in reducing GHG emissions and managing MSW sustainably. Although recycling is the preferred method, WTE can provide a complementary solution that can be leveraged to achieve the desired goals. The transition from landfill to WTE can be a challenging process, but it is a necessary step toward a more sustainable future.

For the purpose of evaluating the speed of transition from landfill to WTE, it was calculated the annual percentage of WTE treatment progress based on each kt of MSW generated (*SWC*), as well as the annual percentage of landfill regression based on each kt of MSW generated (*SLC*). Based on the results obtained in eight developed countries in the WTE industry, the *SWC* ranges between 0,53 %-points/a, and 5,26 %-points/a. On the other hand, the results obtained from the speed of landfill transition (*SLC*) indicated a regress speed between 0,63 %-points/a, and 6,63 %-points/a.

During this study, the environmental impact was examined in terms of two aspects: GHG emissions and dioxin emissions. In the GHG emissions study, the most critical aspect was to determine the changes in GHG emissions from each kt of MSW treated by WTE and landfill over the transition period (ΔG). The results varied in the range of -0,924 kt $_{CO2eq}$/kt $_{(landfill\ +WTE)}$ and 0,421 kt $_{CO2eq}$/kt $_{(landfill\ +WTE)}$ among countries. Since during the transition period all case countries experienced a decrease in landfills and an increase in WTE treatment, it was expected that the changes in GHG emissions would be negative and decrease for all countries. However, the results in Ireland and Lithuania were unexpected. A detailed examination of these countries has revealed that inadequate management of LFG collection, coupled with significant past landfilling and poor waste management practices, has resulted in incremental increases in GHG emissions in these countries.

Another achievement of this study is the calculation of the GHG emissions factor for WTE plants in each country. In this regard, it was found that for each kt of MSW burned by the WTE method, how much kt of GHG is released. Also, this GHG emissions factor was in the range of 0,291 kt $_{CO2eq}$/kt $_{(WTE)}$ to 0,513 kt $_{CO2eq}$/kt $_{(WTE)}$.

Landfill fire occurs accidentally and depending on location and weather conditions in the landfill plants, which releases dioxin as a result. Furthermore, dioxin is also released in the burning of MSW in incinerators. Since dioxin emissions is very dangerous for human health, this study aimed to examine the changes in dioxin emission approximately during the transition period for each country case. The results indicate that with the increase in the use of WTE technology and the decrease in landfilling, which subsequently reduces the accidental incidents of landfill fires in landfill plants, the changes in dioxin emissions between countries were decreasing and between 2,85 mg-TEQ/kt $_{landfill+WTE}$ and 8,36 mg-TEQ/kt $_{landfill+WTE}$.

In general, with the overall evaluation of conducted study, it can be concluded that WTE is a quick and short-term response to reduce environmental impact. By using WTE technology, not only GHG emissions are reduced in a short period of time, but also energy is generated impressively. WTE treatment also reduces landfill, which is the worst scenario for MSW treatment. Since a massive amount of MSW is generated daily in the world, the WTE industry can be a golden opportunity for other countries to reduce the environmental impact and help the energy sector.

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