Waste and Health

Quantifying health costs associated with particulate matter and nitrogen dioxide emissions from waste incinerators in the Netherlands.

Author: Ruben Greven (2545850)
Supervisor: dr. Mark Koetse
External supervisor: Erwin van Woudenberg
Date: 15-7-2019

Preface

This document holds the Master's thesis 'Waste and Health' and attempts to quantify the relation between waste and the adverse health effects it may cause. It has been written in order to complete the Master's programme Environment and Resource Management at the Vrije Universiteit.

The project was undertaken at the request of Sterkur, where I worked as an intern for the duration of the project. The original aim of the project was to assess the impacts that Sterkur's products has on nature and people. In particular those parts of the products that were derived from petrochemicals. This focus has been let go of due to feasibility considerations. Instead, the scope of this thesis is limited to the adverse health effects that occur as a result of incinerating waste. As such, this thesis only discusses a small part of the total impact that products have on people and the environment.

Ruben Greven

July 17, 2019

Abstract

Air pollution is major environmental risk to human health worldwide. The European Environment Agency (EEA, 2019) estimates that exposure to fine particulate matter (PM) and nitrogen dioxide (NO_2) resulted in approximately 467,000 premature deaths in 2015 in the EU-28. An increasing contributor to PM and NO_2 emissions is the incineration of waste, which has become the predominate waste management strategy for waste disposal in the Netherlands. In this thesis, an attempt is made to quantify the health impacts that can be attributed to the emissions from waste incinerators in the Netherlands. To achieve this, the impact pathway approach is used as methodology. This involves a step by step tracing of the emissions from waste incinerator to health effects. The spatial dispersion of the emissions is modelled using OPS. Using concentration-response functions recommended by the WHO (2013), various health effects of PM and NO_2 emissions from incinerators are quantified and monetized. It is estimated that the total health costs add up to €4.7 million (in 2017 prices), with a cost per ton of waste of € 0.61. Most of the health costs are attributed to the NO_2 emissions from waste incinerators.

Content

1 - Introduction

Waste is often considered a nuisance for any country. Disregarded products and materials that can no longer be profitably recovered require management of some sort. Traditionally, landfilling has been the main method of disposing of waste in many countries. Within the waste management hierarchy approach, landfilling is least preferred because of the demand for land and the risk of leaking pollutants to air, water or soil (Dijkgraaf and Vollebergh, 2004). Energy recovery from waste is one step better than landfilling, followed by recycling, reusing and finally avoiding waste generation as most preferred option. Nonetheless, waste incinerators emit air pollutants that have an impact on the health of people (Costa et al., 2014). Common health risks are cardiovascular and respiratory diseases.

The negative effects of air pollution are well studied and considered to be the biggest environmental risk for human health (WHO, 2016). According to the 2016 World Health Organizations assessment of the global burden of disease, air pollution-related conditions were responsible for one out of every nine deaths in 2012, approximately 3 million of which were attributable solely to ambient air pollution. For the Netherlands it was estimated that over four thousand people died as a result of ambient air pollution. The most relevant pollutants in terms of human health impact are NO_2, PM and O_3. The EEA (2019) estimates that exposure to $PM_{2.5}$ concentrations was responsible for approximately 391,000 premature deaths in the EU-28 in 2015. Estimates of premature deaths due to NO_2 and O_3 concentration exposure are around 76,000 and 16,400, respectively.

The contribution of waste to overall air quality is relatively small compared to other sectors such as transport and agriculture (EEA, 2018). Nonetheless, incineration is the primary method for waste disposal in the Netherlands, warranting an enquiry into the consequences of current waste management practices. In 2017, at least 32 percent of all municipal solid waste in the Netherlands was incinerated (CBS, 2018). Remaining waste was recycled or composted (only 1 percent was landfilled).

At the individual level, the health effects may appear to be insignificant. However, because an entire population is exposed to ambient air concentrations of pollutants, overall effects are considerable. Any improvement with regard to air pollution will therefore be beneficial to a society as a whole, both in terms of wellbeing and health care expenditures.

This thesis sets out to quantify and assign a monetary value to health impacts resulting from changes in waste incinerator emissions in the Netherlands, particulate matter (PM) and nitrogen dioxide (NO_2) emissions in particular. The results provide valuable information for policy and decision makers in the field of waste management and air pollution. The approach used in this thesis, the Impact Pathway Approach, has been extensively used for policy evaluations and research (Silveira et al., 2016; COWI, 2000). While this thesis does not attempt to perform a cost-benefit analysis for a particular policy, the methods and results can easily be incorporated into to cost-benefit analyses for policies regarding waste incinerators and air pollution.

1.1 - Research question

This thesis will attempt to answer the following question:

What are the monetized health impacts associated with particulate matter and NO_2 emissions from waste incinerators in the Netherlands?

Sub questions:

- What are the PM and NO_2 emissions from the 12 incinerators in the Netherlands?
- What part of the total PM and NO_2 concentration levels can be attributed to waste incinerators?

- How will the health of individuals be affected by a given concentration level of PM and NO_2?
- What are appropriate monetary values for per unit health impacts (e.g. value of a statistical life) for the Dutch context?

The next section will discuss some background information on the waste management practices in the Netherlands, exposure to nitrogen dioxide and particulate matter pollution and the potential health effects. Section 3 discusses theoretical concepts related to externalities. Section 4 reviews the literature on incinerators and the associated health effects. Section 5 discusses the methodology, where the impact pathway approach will be introduced as well all the required data. Next, the results will be presented in section 6 and discussed in section 7. Finally, section 8 concludes this thesis as well as provide some recommendations.

2 - Background

2.1 - Waste management

Waste management in the Netherlands is based on the National Waste Management Plan ('Landelijk afvalbeheerplan'). In general, the basis for regulations and policies is the waste hierarchy, which orders waste management options in terms of preferability (see figure 1). The most preferred option is to prevent waste from occurring, followed by re-using discarded products. It is difficult to determine exactly how waste is prevented. However, looking at the trend in municipal waste generation may give an indication. The trend in the past ten or so years has been one of significant decline. In 2018, an average of 533 kg municipal waste per inhabitant was generated in the Netherlands (CBS, 2019). Compared to a peak in 2000, this is a decrease of 17.6 percent. However, comparing to a few years before that, say 1993, reveals that a decline of only 8.9 percent has been achieved in the past 25 years.

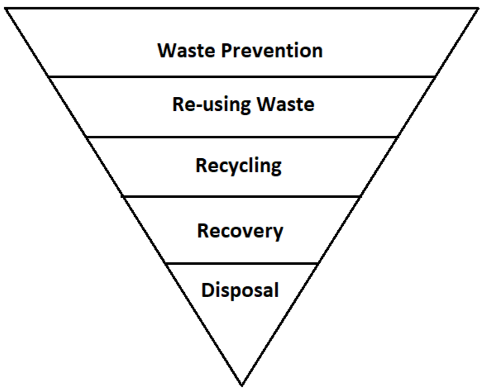

Figure 1. Waste hierarchy.

The next most preferred option is the recycling of materials. In the Netherlands, further distinction is made between recycling materials for use in the same or similar products, recycling materials for use in different products and, finally, the chemical recycling of materials. Recycling has been a much debated topic and plays a major role in the goal of achieving a circular economy. Similar to preventing waste from occurring, progress has been made in this category as well. In 2014, 51 percent of municipal waste was recycled or composted, an increase of 4 percent points, compared to 2004 (EEA, 2016). For packaging waste the recycling rate has increased even more, from 58.5 percent in 2004 to 68.5 percent in 2014.

The second least preferred option is the recovery of heat and/or energy from waste products. This can be achieved by incinerating waste in so-called 'Waste-to-Energy' (WtE) plants. Similar to coal and gas fired power plants, the heat generated from the combustion is used to turn water into steam, which powers steam generators and produces electricity. The remaining steam can be transported by pipes to be used for nearby district heating or industrial processes. All incinerators in the Netherlands are WtE plants. In 2017, 7.6 Mton waste was incinerated, 22.3 PJ of heat was delivered and 3,688 GWh of energy was generated (Rijkswaterstaat, 2018). Approximately 80% of the energy was delivered to the net or nearby industrial complexes. Most of the remaining energy was used on-site for flue gas cleaning.

Finally, the least preferred option is the disposal of waste, which entails discharging into waterbodies or landfilling. Because of a ban on the landfilling of household waste, the main categories of waste that are landfilled are non-combustible materials such as soil and construction debris. A small portion of the bottom ash that are produced in waste incinerators are also landfilled, but most of it is re-used in new infrastructure and construction projects.

2.2 - Exposure

As waste is incinerated and converted from bulk material to ashes, gases and airborne particles, pollutants are emitted resulting in an inevitable exposure of the population to these pollutants. Figure 2 shows the location of 12 incinerators and the population density in Dutch districts. Except for a few, most incinerators are located in or near densely populated areas. This is to be expected, as more waste is produced in more densely populated areas, making the collection and transport of waste more efficient.

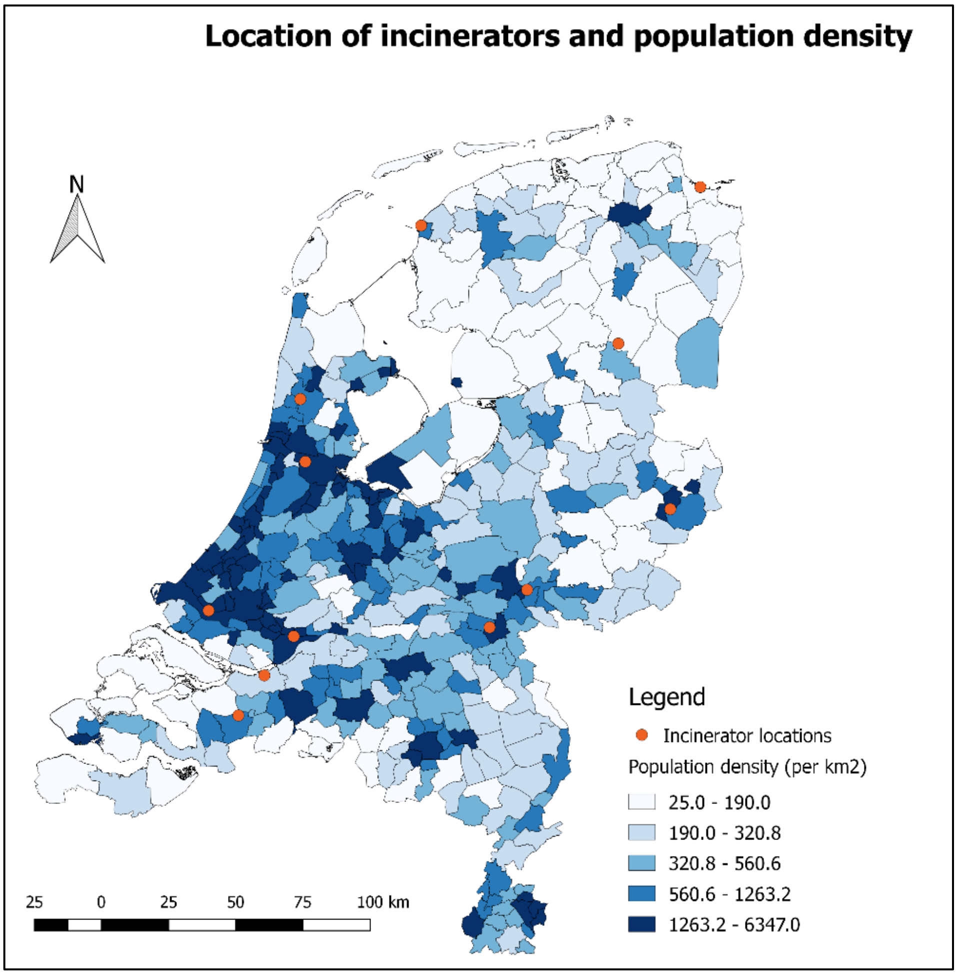

Figure 2. Location of incinerators and population density in Dutch municipalities in 2017.

2.2.1 - Particulate matter

Particulate matter (PM) is a mixture of solid and liquid particles suspended in the air varying in size, shape, chemical composition, solubility and origin. PM is classified according to their aerodynamic diameter as differently sized particles can have different health effects. PM_{10} refers to the subset of inhalable particles with an aerodynamic diameter under 10 µm and are thought to be able to penetrate the thoracic region of the respiratory tract (Brown et al., 2013). For smaller particles, that are capable of penetrating the gas-exchange region of the respiratory tract, a cut of point of particles with an aerodynamic diameter of under 2.5 µm ($PM_{2.5}$) is often used. Furthermore, the 2.5 µm boundary can also be used to distinguish between fine and coarse particles. Fine particles usually originate from combustion processes or are formed from gases as secondary particulate matter. Coarse particles often consists of dust or other particles that become airborne due to agriculture, traffic, mining, volcanic activities etc. Another important source of coarse particulate matter, especially in the Netherlands, are sea salts. It is important to note that $PM_{2.5}$ is a subset of PM_{10}. Larger particles are removed from the atmosphere by dry deposition under the influence of gravity, with the largest particles being removed the quickest. Particles that are not deposited due to gravity can remain airborne for weeks and travel large distances. Such particles are most effectively removed from the atmosphere through precipitation (wet deposition).

2.2.2 - Nitrogen dioxide

Nitrogen oxides (NO_x) emissions result from the conversion of nitrogen that is embedded in the waste or from atmospheric nitrogen that is introduced as combustion air, alongside oxygen. Usually, the proportion NO/NO_2 in stack emissions is around 95% NO and 5% NO_2. NO_2 and NO are often considered together as 'NO_x' due to the rapid cycling between the two in the atmosphere. The interconversion between the two is facilitated by the presence of tropospheric O_3. The interaction between NO, NO_2 and O_3 can be represented by the following three simplified reactions (Monks et al., 2015):

$$NO_2 + \text{light} \rightarrow NO + O \quad (1)$$

$$O + O_2 + M \rightarrow O_3 + M \quad (2)$$

$$NO + O_3 \rightarrow NO_2 + O_2 \quad (3)$$

Where M is a co-reactant (e.g. N_2). These reactions create a closed loop that essentially recycles an oxygen atom. Finally, the ratio NO/NO_2 finds an equilibrium that is dependent on the local concentration of O_3 and the frequency at which light converts NO_2 to NO. More O_3 will result in a lower NO/NO_2 ratio as can be derived from reaction 3. At night, reaction 1 is slowed down so that less NO is formed from NO_2. In the presence of volatile organic compounds (VOCs), the formation of O_3 is further complicated as more oxygen is made available to convert NO to NO_2.

NO_2 can be further oxidised to form NO_3, which can react with hydrocarbons to form HNO_3. During daytime, NO_2 can also react with OH to form HNO_3 directly. HNO_3 can react with ammonium to from ammonium nitrate, which itself is part of secondary particulate matter. NO_x itself is not water-soluble and is therefore not efficiently deposited. The main sink of NO_x from the atmosphere is through HNO_3, which is soluble in water and can thus be deposited. It can also be deposited when HNO_3 is embedded in secondary particulate matter.

2.2.3 - Concentration levels

The contribution of waste incinerators to the overall exposure to PM and NO_2 is only limited (less than 1 percent). Incinerators are equipped with modern air pollution control (APC) installations that prevent most of the emissions. For example, the guidebook on emissions from waste incinerators published by the EEA (2016) reports abatement efficiencies of around 99% for PM. The overall concentration levels are being tracked in two ways: modelling emissions and monitoring stations. Each year, based on the modelling of emissions, concentration and deposition maps of several pollutants are published by the RIVM (Velders et al., 2018). Mean concentration levels for nitrogen dioxide and particulate matter are modelled and calibrated using data from monitoring stations. Mean concentration levels for the Netherlands in its entirety in 2017 for $PM_{2.5}$, PM_{10} and NO_2 are 9.6, 16.5 and 14.8 µg/m^3, respectively. Concentration levels of both NO_2 and PM have been declining in the past and are expected to continue to decrease. Figure 3 shows the concentration levels of PM and NO_2 across the Netherlands in 2015. For NO_2 it can be clearly seen how traffic contributes to increased exposure. Furthermore, for both particulate matter maps the concentration levels and patterns are clearly different in the north compared to the south of the Netherlands.

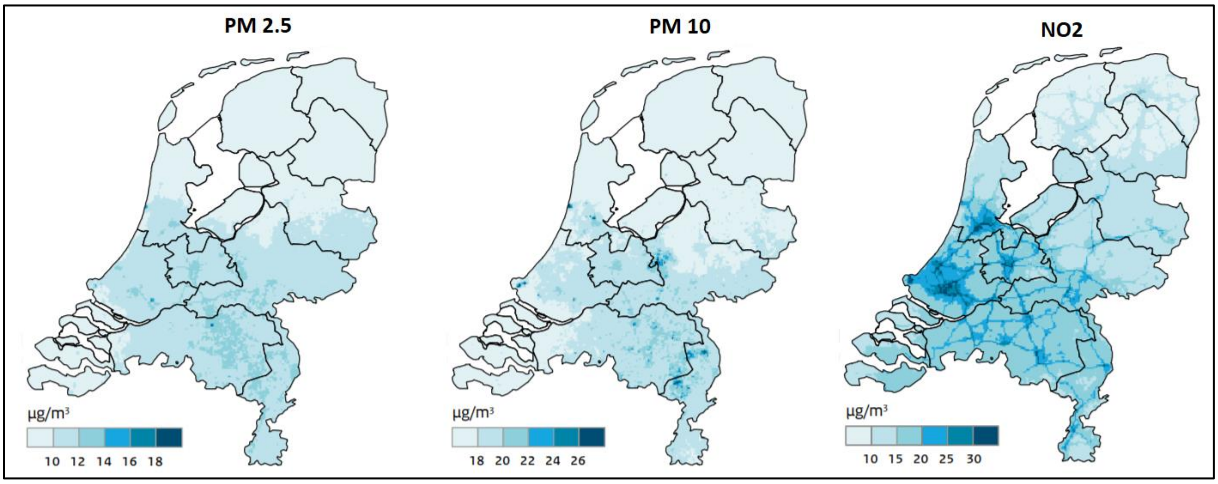

Figure 3: Large scale concentration maps for particulate matter and nitrogen dioxide in 2015 (Velders et al., 2016).

Modelling has the benefit that is estimates the concentration levels everywhere in the Netherlands. A major disadvantage, however, is that it estimates concentration levels based on anthropogenic emissions, ignoring the contribution of natural processes. As a result, a calibration factor is used to correct for this. The calibration factor is determined based upon measurements from measuring stations across the Netherlands. An obvious advantage of using measuring stations to determine the concentration level is that the results are much more accurate and reliable. However, the results can only be used locally and due to the limited amount of measuring stations no maps such as in figure 3 can be created.

Concentration levels are measured at around 80 monitoring stations across the Netherlands for NO_2 and PM_{10}. There are less, around 50, stations monitoring $PM_{2.5}$ concentrations. The concentration levels are monitored at three types of locations: regional, urban and traffic-heavy locations. The results from the monitoring stations can be viewed and accessed from 'Luchtmeetnet.nl'. In general, the regional monitoring stations measure the lowest concentrations levels and stations in traffic-heavy locations measure the highest concentrations levels. In 2017, the annual mean $PM_{2.5}$ concentrations levels ranged from 9 to 15 µg/m^3. PM_{10} concentrations ranged from 13 to 28 µg/m^3 and NO_2 concentrations ranged from 8 to 48 µg/m^3.

2.2.4 - Air quality regulations

Air quality legislation is laid down in EU Directive 2008/50/EC and is transposed into Dutch national law in the 'Wet Milieubeheer'. The directive contains measures aimed at reducing the negative effects of ambient air pollution on human health and the environment (EC, 2008). Member States are required to monitor and assess ambient air quality and ensure that such information is made available to the public. The annual mean concentration limit values set within the directive are as follows: 40 µg/m^3

for nitrogen dioxide, 40 µg/m^3 for PM_{10} and 25 µg/m^3 for $PM_{2.5}$. These limit values have to be met at present times. For 2020, an indicative limit value of 20 µg/m^3 for $PM_{2.5}$ is also included. In the Netherlands, the modelled estimates for $PM_{2.5}$ did not exceed the limit anywhere in 2017 as the maximum value was 16.4 µg/m^3. However, both the limits for PM_{10} and NO_2 were exceeded at a few points places. Maximum values of 52.9 and 48.2 µg/m^3 were modelled for PM_{10} and NO_2, respectively. EU limits for PM were not exceeded at any of the monitoring stations. However, five stations monitoring NO_2 measured mean concentrations levels exceeding 40 µg/m^3.

However, the limits in the directive should not be regarded as safe limits below which no negative health effects occur. Instead, they are best viewed as intermediate objectives. Based on a review of the scientific literature, the WHO (2005) provides guidelines for limit values for, among others, particulate matter and nitrogen dioxide. For NO_2 the guideline is equal to the EU directive, that is, 40 µg/m^3 as an annual mean. For particulate matter, the guidelines are lower than the EU directive: 20 µg/m^3 for PM_{10} and 10 µg/m^3 for $PM_{2.5}$ as an annual mean.

Table 1. Modelled mean concentration levels compared to EU and WHO limit values. All values are in µg/m^3.

Pollutant	Mean concentration estimate[1]	Uncertainty range[1]	EU Directive	WHO guidelines
$PM_{2.5}$	9.6	7.1-12.1	25 (20[2])	10
PM_{10}	16.5	14.0-19.0	40	20
NO_2	14.8	12.6-17.0	40	40

1. Velders et al. (2018).
2. Indicative limit for 2020.

While the central estimates for 2017 are all below EU regulations and WHO guidelines, considering only the mean concentration levels in major cities and the upper range of the estimates will give concentration levels exceeding the WHO guidelines for both categories of PM (Velders et al., 2018). Based on the modelled concentration levels, the limits of the EU directive are not at risk of being exceeded, even in major cities and considering the upper value of the uncertainty range.

Only about a third of the monitoring stations measured $PM_{2.5}$ concentration levels on or below the WHO guideline. For PM_{10}, about two thirds of the stations measured concentration levels on or below the WHO guideline. In summary, further progress towards reducing PM and NO_2 concentrations levels is still warranted.

Besides regulations for overall air quality, regulation for emissions from waste incinerators also exist. Emission limits for waste incinerators as well as other requirements are laid down in the EU's Waste Incineration Directive 2000/76/EC. General limits on the total annual load of emissions per incinerator do not exist, because all installations differ in terms of capacity. Instead, the limit values are defined as daily averages and in terms of volume of exhaust. For PM[1] the limit is 10 mg/m^3 exhaust and for nitrogen oxides, expressed as NO_2, the limit is 200 mg/m^3. For both limits, stricter national limits are in place, laid down in the Activities Decree ('Activiteitenbesluit'). The daily average limit values for PM and NO_2 are 5 mg/m^3 and 180 mg/m^3, respectively. For NO_2, a monthly average limit of 70 mg/m^3 also exists. The regulations are summarized in table 2.

Table 2. EU and national regulation regarding emissions to air from waste incinerators. All values are in mg/m^3.

Pollutant	EU directive (daily average)	Activities Decree (daily average)	Activities Decree (monthly average)
PM	10	5	
NO_2	200	180	70

2.3 - Health impacts and other consequences

The adverse health impacts of air pollution have been studied extensively in the past. Once inhaled, both PM and NO_2 can cause a wide range of health impacts of varying severity (Costa et al., 2016). For particulate matter the health impacts depend partially on the size of the particles, as the smaller particles are able to penetrate further in the respiratory system and are, therefore, expected to be able to do more damage. The damage occurs as a result of both chemical and physical interaction between the PM and the lung tissue. While it can be expected that inhaling air pollution can cause damage to the respiratory system, inhalation of PM is also associated with cardiovascular diseases, which may pose even a more significant health risk.

At the lower end of the severity spectrum are symptoms such as coughing and irritations in the airways. More severe symptoms and impacts include bronchitis, asthma attacks as well as other respiratory and cardiovascular diseases. The most severe consequences of PM exposure are lung cancer and premature mortality. A review of European epidemiological studies by Pelucchi et al. (2009) concludes that long term exposure to PM can be directly related to mortality, in particular from cardiopulmonary diseases, which affect the heart and lungs. In general, $PM_{2.5}$ is considered to be more dangerous than PM_{10} as it can penetrate deeper into the lungs and remain airborne for longer (Pope and Dockery, 2006)

Exposure to NO_2 can have similar health effects that are manifested in the lungs and heart (Costa et al., 2016). For example, increased NO_2 concentration level have been linked to airway inflammation and a decrease in the immune defence, resulting in increased susceptibility to respiratory diseases. Like PM, NO_2 has also been associated with mortality, in particular due to short-term exposure. The evidence for the health impacts due to NO_2 is somewhat more controversial, because NO_2 is a an inevitable combustion product and therefore often correlated with many other pollutants, making it more difficult to study the effect of only NO_2.

Because exposure to air pollution is almost impossible to avoid, the risks apply to the population in its entirety. Consequently, the total health impact of air pollution can be quite considerable, even when the risk per individual may appear to be relatively low. As a result of exposure to PM_{10} in the Netherlands, it is estimated that in 2013 around 1,628 people died prematurely (CBS, PBL, RIVM, WUR, 2015). Furthermore, it was estimated that in 2012 around 1 percent of all respiratory and cardiovascular hospital admissions were the result of short term exposure to PM_{10}. In 2017, 1 percent of all respiratory and cardiovascular hospital admissions corresponds to around 1,300 and 2,300 hospital admissions.

While not the focus of this thesis, air pollution also effects other categories besides human health. Other impacts related to NO_2 include eutrophication and acidification of aquatic and terrestrial ecosystems. As a precursor of ozone, which damages plants, NO_2 pollution can also be related to reduced agricultural productivity. Furthermore, tropospheric ozone also contributes directly to climate change. PM is mostly relevant for human health, but is also associated with soiling of buildings, resulting in reduced amenity value. Using the same concentration-response functions recommended by the WHO, the OECD (2016) also estimated the economic costs of air pollution. This study includes the effects of ozone on human health and agricultural yield, but ignores the effects of NO_2. For OECD countries in Europe it is estimated that welfare costs resulting from premature deaths add up to 730 billion USD (2010 prices). This estimate is expected to increase significantly in the future.

3 - Theoretical framework

A society faces certain trade-offs where it concerns waste management. All approaches will incur costs and require investments. Several options are considered to be too expensive, technologically unfeasible or otherwise undesirable. For example, ignoring the problem entirely will quickly result in protest from people as waste will accumulate in households and businesses. The other extreme is currently unattainable. Progress is made towards a world where all products are made such that all materials can and are recovered and reused, effectively reducing the amount of waste to zero, but we are not there yet. The immediate constraints appear to be financial and technological as well as consumer preferences.

At present times, any waste management strategy will involve multiple approaches, including recycling, incineration and landfilling, which inevitably results in some pollution of the environment. All approaches have advantages and disadvantages. Not managing waste is practically impossible as waste will stack up at people's houses and businesses, landfilling is no longer deemed an option as it requires increasingly larger amounts of space and may result in hazardous leakage of pollutants into the air, water or soil. Incineration plants with expensive air pollution control (APC) will reduce the volume of waste significantly as it is reduces to mostly gaseous molecules that are emitted into the atmosphere. Better APC is expensive, but will also reduce the negative health impacts associated with air pollution.

Economic theory can place these trade-offs in a theoretical framework in which they can be understood and analysed. The atmosphere is a medium that absorbs emissions from industrial activity and contains the air we breathe and are exposed to. Emissions into the atmosphere can be understood as an externality and air quality can be understood as a public good, no one can realistically be excluded from consuming air quality and consumption by one does not reduce the overall availability of air quality. State intervention in the form of air quality standards and emission limits regulate the quality of the air. These regulations represent a society's desire for clean air, while also allowing emissions associated with general economic activity.

3.1 - Externalities

The full costs, or social costs, of waste incinerators include private as well as external costs. While private costs are considerable in the case of incinerators, they are not the subject of study in this thesis. External costs are the costs associated with the externalities resulting from waste incineration. In Economic theory, externalities refer to the effects of a market transaction on someone that is not involved in that transaction (Harris and Roach, 2015). There are several distinctions that can be made regarding externalities. For example, externalities can be both positive and negative. Distinctions can also made regarding the impact category or the physical source. In this thesis, the source of externalities are the emissions associated with waste incineration and are limited to health impacts only. In the context of waste incinerators, the disamenity from the incinerator in the landscape can also be considered an externality (COWI, 2000).

In a basic economic analysis of markets, economic efficiency is achieved when marginal costs equal marginal benefits, the costs associated with a marginal increase in provision of a good will not outweigh the additional benefit. Disregarding externalities, private decision making can result in socially optimal outcomes, the market equilibrium and price (Q_m and P_m, in figure 2). Including externalities in this analysis gives rise to the concept of social marginal costs, which includes both the private marginal costs and the external costs. From a social perspective and the presence of externalities, optimality is achieved when social marginal costs equal marginal benefits. Compared to the market equilibrium, this implies lower quantity and a higher price (Q* and P*, in figure 2). Regarding waste incinerators, ignoring externalities results in an oversupply of waste incineration and emissions into the atmosphere.

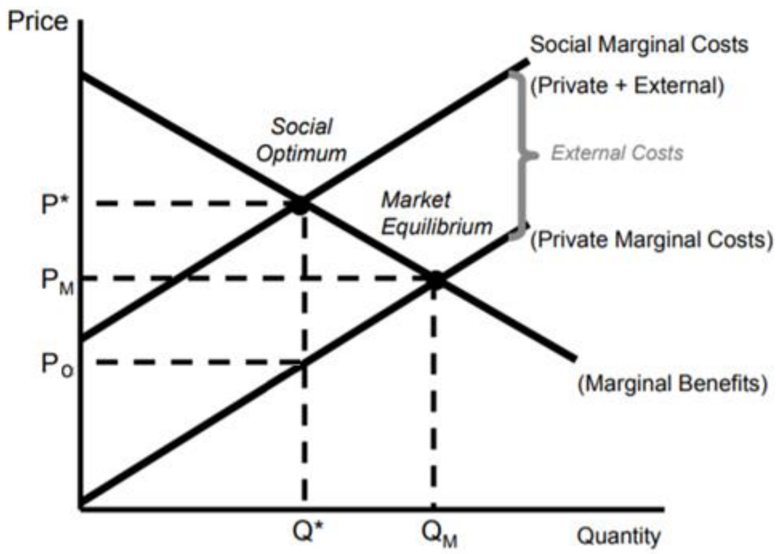

Figure 4. Negative externalities in a market. Adapted from Harris and Roach (2015).

Economic optimality can be achieved by internalizing the external costs. The most common approach is to impose a tax on pollution. The tax introduces an additional cost per unit of emission. A tax per unit of pollution equal to the external costs for that same amount of pollution increases the private marginal costs to the social marginal costs. The producer will reduce the output from Q_m to Q*, which represent the output for the social optimum. The socially optimal tax, the Pigouvian tax, is equal to the external costs in the optimum, i.e. the difference between P* and P_o in figure 2. Finding out the right amount of tax is generally difficult and sensitive to errors. Although the aim of this thesis is not to design a Pigouvian tax for waste incineration, the results presented here may still be useful for any effort towards such an objective.

4 - Literature review

Numerous studies have reviewed the epidemiological evidence relating waste incinerators to health effects (Hu and Shy, 2001). Porta et al. (2009) reviewed epidemiological studies examining the relation between landfills and incinerators and the health of employees and people living in the nearby. Limited evidence was found for an increased risk for several types of cancer, reproductive disorders as well as respiratory diseases, especially in children. However, they found that most of the assessed studies suffer from several issues. Because of this, there are large uncertainties and interpretation should be done with care. They conclude that more detailed research is needed. Also, no mentioning was made of which pollutant was responsible for the health effects.

Another review of waste management options focussed solely on the health effects of dioxins, arguing that is has been the most concerning pollutant in terms of human health (Giusti, 2009). Dioxins are a group of persistent organic compounds that are also emitted from waste incinerators. They are relatively resistant to biodegradation and therefore bioaccumulate. As a result, the main pathway for exposure to dioxins is not through inhalation, but through digestion of contaminated food. Many dioxins are considered toxic and can cause different types of cancer. While there are indications that waste incinerators can be linked to specific diseases, it has proven to be extremely difficult to prove this unequivocally, due to the limitations of environmental epidemiological studies. Because environmental epidemiological studies are based on observations, rather than experiments, controlling for other factors is very difficult. Giusti (2009) also notes that many of the epidemiological studies were based on the emissions from old waste incinerators. Newer abatement technologies have made modern waste incinerator much cleaner than the old ones. While better designed epidemiological studies may be conducted in the future, the continuing decrease in harmful emissions may still make it hard to detect any excessive health effects. In conclusion, the current epidemiological research has not been able to prove convincingly and without controversy that proximity to waste incinerators is associated with a detectable increase in risk of specific health effects.

Waste incineration is often compared to landfilling, as they are the two main final waste disposal methods. Dutch policy towards final waste disposal is focused entirely on incineration, even prohibiting landfilling of municipal solid waste. The high population density and land scarcity in the Netherlands are undoubtably factors that have contributed to this focus. While landfilling is often considered the worst option in terms of environmental externalities, several studies have questioned this assumption. Dijkgraaf and Vollebergh (2004) estimated private and social (external) costs for both landfilling and incineration. Their results indicate that incinerators have much larger private costs, partially due to expensive air pollution control, but also higher gross social costs, 45.95 euro per ton of waste compared to 26.36 euro per tonne waste. Only once energy and material recovery from incineration is considered will net social costs be lower incinerators compared to landfilling.

All waste incinerators in the Netherlands are so-called Waste-to-Energy (WtE) plants, where electricity and heat are generated from the combustion of waste. Depending on the alternative energy sources, often fossil fuel based, WtE prevents externalities from alternative energy sources. This has been a main selling point for waste incinerators with energy recovery in favour of landfilling. A complete transition towards renewable energy will undermine this argument as benefits from avoided emissions become zero.

Using the impact pathway approach, Rabl et al. (2008) also compared external costs from incinerators and landfilling. They find external costs of emissions from incinerators of 22.9 euro per tonne of waste, mostly the result of PM, NO_x, SO_2 and CO_2 emissions. They also find that external costs from landfilling are lower, roughly 13 euro per tonne of waste. These costs are almost exclusively the result of CO_2 and CH_4 emissions. In terms of health impacts, landfilling seems to be the preferred option. However, it should be noted that the primary health impacts from landfilling occur through leachate, which is far more difficult to quantify than health impacts from air pollution.

In the past, values were often calculated per unit of pollution. For example, Eshet et al. (2005) reviewed a range of studies that estimated values per kg of emissions. Averages for PM and NO_2 are 36.16 US dollar (2003) and 6.81 US dollar (2003), respectively.

Using a similar methodology as the one proposed in this document, Brandt et al. (2013) estimated health related external costs from air pollution from economic sectors in the EU and Denmark. For the EU they estimated external health costs of 7.8 billion euros resulting from waste treatment in the EU. While being far from insignificant, this only represents 1% of total external health costs resulting from air pollution from all economic activity in the EU.

5 - Methodology and data

5.1 - Impact Pathway Approach

A popular approach towards quantifying damages from activities that result in air pollution is the Impact Pathway Approach (IPA) (Silveira et al., 2016). The IPA systematically identifies and traces the effects of airborne pollutants, from changes in emissions at point or diffuse sources, to changes in (social) welfare (DEFRA, 2019) (see figure 2). The step by step tracing of airborne pollutants along its impact pathway allows for a quantification of the impacts in terms of damages, which can be assigned a monetary value. The fact that the IPA facilitates the monetization of air pollution makes it a valuable method for the appraisal of air quality policies. The IPA was developed in the context of the ExternE project series funded by the European Commission for the purpose of monetizing transport externalities (Bickel and Friedrich, 2001).

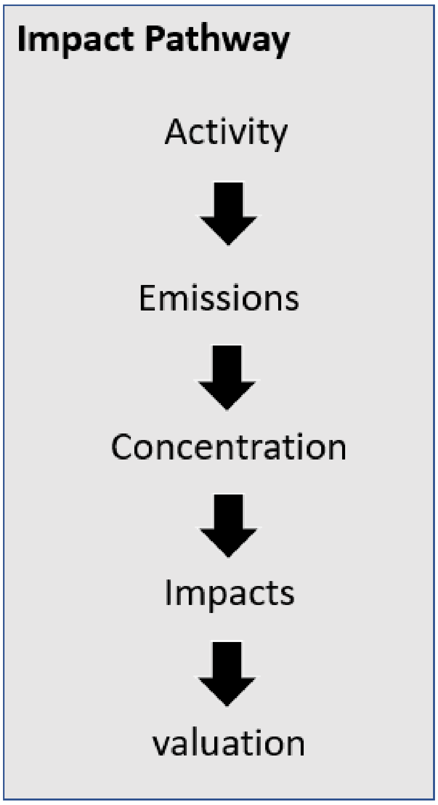

Figure 5. Impact pathway.

The assessment of the impact chain starts with identifying an activity that generates emissions, such as incinerating municipal waste. Emitted air pollutants are transported and transformed in the atmosphere and change the concentration of the pollutant. A group of receptors (humans, crops, buildings, etc.) is exposed to increased concentrations of the pollutant. The interaction between the increased concentration of the pollutant and the receptor while induce a response from the receptor. Functions that describe this relation are referred to as exposure-response functions. Information on the total group of exposed receptors, change in exposure and the exposure-response functions can be combined in order to derive the impacts. Finally, the impacts can be valued and aggregated at different levels. For a specific pollutant and health impact for example, but also for an entire scenario that involves multiple pollutants and impact categories. Another result can be the damage costs per tonne of emitted pollutant, which in turn can be applied to other scenarios, under certain conditions.
The next sections will discuss the individual stages of the method in more detail as well as how they can be operationalized within the context of waste incineration. A detailed overview of the impact pathway approach applied to the health impact from emission originating from waste incineration is given in figure 6. It presents both the individual steps as well as the method for conversion of each step to the next.

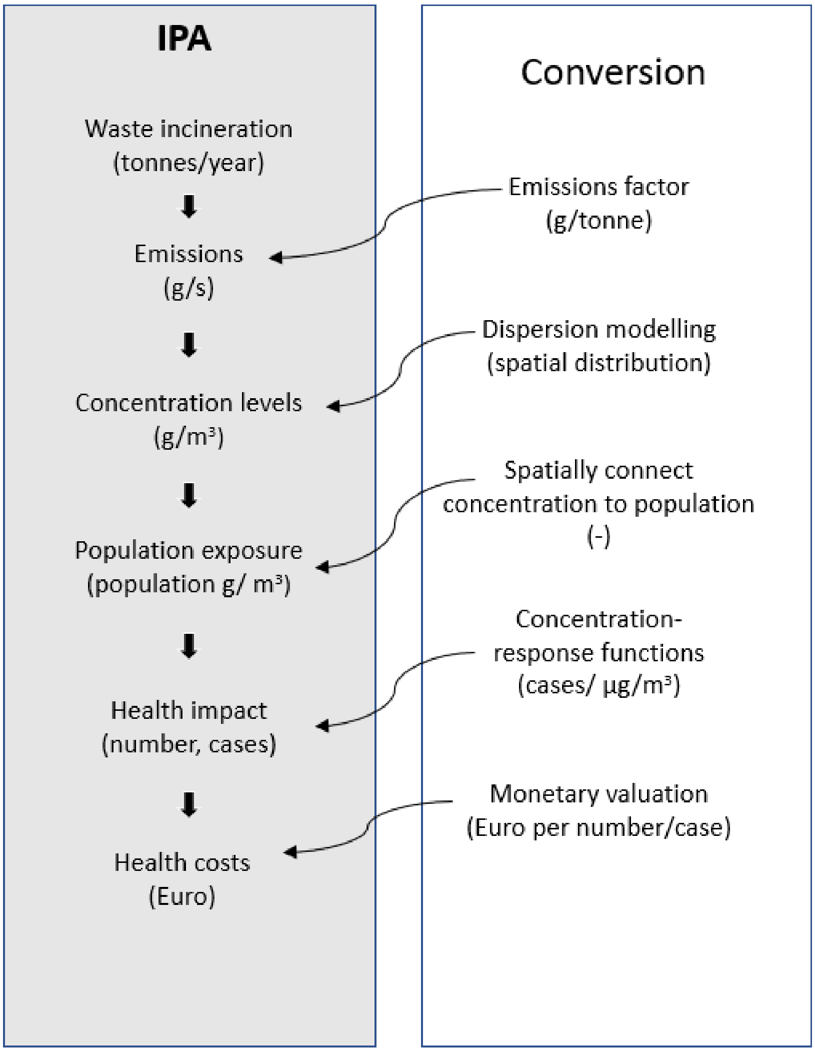

Figure 6. Detailed overview of the IPA applied to waste incinerators

5.2 - Activity

Since 2011 there have been 13 waste incinerators in operation with a current capacity of around 8 Mt waste. One incinerator only processes dangerous medical waste and has a relatively low capacity. Because of this, the incinerator in question is not considered for the purpose of this thesis. The remaining 12 incinerators process all Dutch residual waste as well as a fraction of waste from the United Kingdom. For the amount of waste that is incinerated as well as data per installation, data from Rijkswaterstaat (2018) can be used. The trend for total waste incinerated in Dutch incinerators for the period 2011-2017 is presented in figure 7.

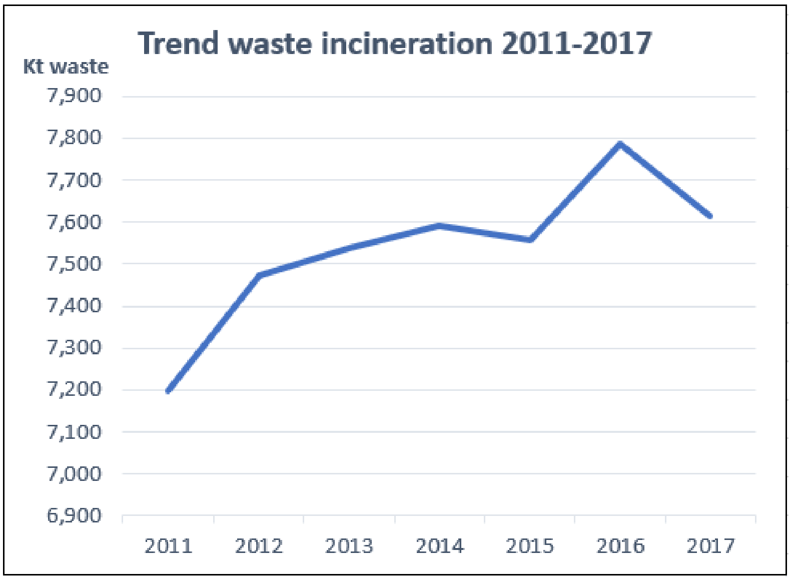

Figure 7. Total waste incineration trend 2011-2017.

The figure shows that total waste incineration has steadily increased for this period, with a maximum of 7,785 Kt in 2016, which decreased to 7,614 Kt in 2017. The total amount of waste incinerated has increased by 5.8% over the period 2011-2017. Data for individual installations in 2017 is presented in table 3 and is used for the remainder of this thesis. Data for all installations and total for the period 2011-2017 is provided in appendix A (table A1).

Table 3. Total amount of waste incinerated per location in 2017.

Location	Waste (t)
Harlingen	260,993
Delfzijl	343,789
Wijster	658,000
Hengelo	622,198
Weurt	272,384
Alkmaar	677,940
Dordrecht	290,801
Botlek Rotterdam	1,282,970
Duiven	390,340
Amsterdam	1,477,126
Moerdijk	1,017,110
Roosendaal	321,127
Total	**7,614,778**

5.3 - Emissions

Combustion under aerobe circumstances results in the emission of several substances and gases. The immediate gases that are produced in incinerators are referred to as flue gas. The flue gas consists of, among others, CO2, NOx, SO2, particulate matter and dioxins. A large portion of the gases are removed using air pollution control (APC) technologies. A wide range of flue-gas cleaning technologies exist. For dust (some documents, including the regulation refer to PM as dust), bag filters and electrostatic precipitators (ESP) are commonly used to clean the flue-gas. For NO_2, the most common techniques to reduce NOx formation are combustion control as well as selective catalytic reduction (SCR). Modern flue gas cleaning equipment greatly reduces particulate matter emissions, but is most effective against larger particles. Consequently, particulate matter emissions are mostly made up of the smaller sized particles. The remaining gases are emitted from the smoke stack. Incinerators operate and emit continuously and monitor their emissions in weight per volume of gas/smoke (g/Nm^3) or per unit of time (e.g. g/s). As a consequence of data and modelling limitations, only the PM and NO2 are considered in this thesis. Alternative scenarios related to the air pollution control (APC) can be considered at this stage. However, APC in Dutch incinerators is among the most effective in the industry and capable of complying with the regulations.

5.3.1 - Emissions data

There are several methods to obtain emission estimates from waste incinerators. A guidebook by the European Environmental Agency (EEA) describes three different tiers of methods for municipal waste incinerators (EEA, 2016). Tier 1 is using a default emission factor in terms of emissions per unit of waste. For NOx and PM emissions the default factors are 1,071 and 3 gram per ton of waste, respectively. PM_{10} and $PM_{2.5}$ are given the same emission factor, because it is assumed that (almost) all PM emissions are of the $PM_{2,5}$ kind. Tier 2 is similar, but it takes into consideration the use of different technologies at different incinerators. This requires technology specific emission factors. Finally, tier 3 is the use of data on emissions of individual incinerators and is considered to be the most accurate. A combination of method 1 and 3 will used for this thesis. Because emissions of NO_2 and PM can vary between years while the amount if incinerated waste remains more or less equal, emissions in one year may not necessarily be representative. Instead, emissions and the amount of incinerated waste in multiple years is used to calculate installations specific emission factors. These factors are multiplied with the amount of incinerated waste in 2017 to obtain total emissions for NO_2 and PM, which serves as an input for the atmospheric dispersion modelling.

Companies are required to report their annual emissions to the government in a so-called Pollutant Release and Transfer Register (PRTR). Emissions data for over 350 polluting substances is collected in the Emissions Register, this data is freely and publicly accessible (http://www.emissieregistratie.nl/erpubliek/bumper.en.aspx). The data in the register is used to produce reports for national lawmakers as well as international organization such as the European Union and the United Nations. Most incinerators only report their PM_{10} emissions and not their $PM_{2.5}$ emissions. Instead, $PM_{2.5}$ emissions are estimated using an emissions factor of roughly 0.99, because most PM_{10} is in fact $PM_{2.5}$. The register does not report NO and NO_2 separately. Instead, the value for NO_x is expressed as NO_2 in the register. For the period 2011-2017, only data on emissions for 2015 and 2016 is available. NO_2 and PM emissions for 2015 and 2016 for all incinerators are presented in table 4.

Table 4. PM and NO_2 emissions of incinerators in 2015 and 2016.

	2015			2016		
Location	NO2 (kg)	PM10 (kg)	PM2.5 (kg)	NO2 (kg)	PM10 (kg)	PM2.5 (kg)
Harlingen	84,970	432	428	95,740	758	751
Delfzijl	115,200	1,563	1,550	122,600	1,539	1,525
Wijster	210,100	5,678	5,628	177,500	3,328	3,299
Hengelo	271,300	3,216	3,193	283,800	3,360	3,336
Weurt	134,400	1,300	1,289	135,400	2,200	2,181
Alkmaar	316,800	1,475	1,159	319,500	1,267	996
Dordrecht	118,000	1,259	1,248	114,000	1,260	1,248
Botlek Rotterdam	523,600	2,321	2,303	523,700	2,758	2,738
Duiven	164,500	1,928	1,911	143,800	1,740	1,725
Amsterdam	653,400	13,970	13,970	691,600	14,240	14,240
Moerdijk	318,800	1,340	1,328	353,300	1,272	1,261
Roosendaal	137,400	716	709	141,000	704	698
Total	**3,048,470**	**35,198**	**34,716**	**3,101,940**	**34,426**	**33,998**

The emission factors are calculated for each installation by dividing the emissions by the incinerated waste for both 2015 and 2016 (see appendix table A2-3). The average emission factor is calculated for each installation and multiplied by the amount of waste incinerated in each incinerator to obtain emissions for the year 2017. The results are presented in table 5. Regarding the emission factors, the spread around the mean is much smaller for NO_2 compared to PM. The minimum and maximum emission factors for NO_2 are 0.27425 kg/t and 0.47476 kg/t, respectively. In contrast, the minimum and maximum emission factors for PM_{10} are 0.00140 kg/t and 0.00997 kg/t, respectively. In total, around 3 million kg of NO_2 and 35 thousand kg PM are emitted from all incinerators collectively.

Table 5. Waste, emissions and emission factors for incinerators in 2017.

Location	Waste (t)	e_{fno2} (kg/t)	e_{fpm10} (kg/t)	$e_{fpm2.5}$ (kg/t)	NO2 (kg)	PM10 (kg)	PM2.5 (kg)
Harlingen	260,993	0.36991	0.00241	0.00239	96,544	629	623
Delfzijl	343,789	0.31712	0.00414	0.00410	109,022	1,422	1,401
Wijster	658,000	0.27425	0.00638	0.00632	180,458	4,198	4,161
Hengelo	622,198	0.43341	0.00513	0.00510	269,668	3,195	3,171
Weurt	272,384	0.46663	0.00613	0.00607	127,101	1,669	1,654
Alkmaar	677,940	0.47539	0.00205	0.00161	322,288	1,389	1,091
Dordrecht	290,801	0.37556	0.00408	0.00404	109,213	1,186	1,175
Botlek Rotterdam	1,282,970	0.38728	0.00188	0.00186	496,873	2,407	2,389
Duiven	390,340	0.40206	0.00478	0.00474	156,939	1,867	1,851
Amsterdam	1,477,126	0.47476	0.00997	0.00997	701,286	14,721	14,722
Moerdijk	1,017,110	0.35881	0.00140	0.00138	364,953	1,421	1,408
Roosendaal	321,127	0.40206	0.00205	0.00203	129,111	658	652
Total [Average]	**7,614,778**	**[0.39477]**	**[0.00420]**	**[0.00414]**	**3,063,457**	**34,762**	**34,309**

5.4 - Dispersion modelling

In order to quantify the exposure of people to harmful emissions, the transport of pollutants in the atmosphere needs to be modelled. For this study the Operational Priority Substances (OPS) model is utilized. OPS calculates concentration and deposition levels and performs well on local scales. Meteorological information required by the model is provided by the Royal Netherland Meteorological Institute (KNMI) on yearly basis. The primary use of the model is the creation of annual large-scale concentration maps in the Netherlands (Velders et al., 2018). The OPS model cannot reliably model the chemical reactions regarding ozone (Van der Swaluw et al, 2017). Effects related to ozone are therefore excluded in this thesis. Recently, Sauter et al. (2018) documented a detailed description and validation of the latest versions of the model (OPS 4.5.x). Spatially combining concentration levels with population data from the CBS will give an estimation of the population that is exposed and to what increases in concentration they are exposed to. In our daily lives, we move around and are therefore exposed to different levels of concentrations depending on where we are at any given time. Correctly measuring the exposure would thus require that we follow individuals throughout the day. Because this is practically impossible, using residence location as an indicator for population distribution is an accepted practice. However, an argument could also be made for using the location of employment as a large portion of our day is spend there. Because data on residence location is readily available in a format that allows for combination with OPS output, residence location is used as an indicator for population density across the Netherlands. Data on population and its density across the Netherlands is made available in district and neighbourhood maps ('Wijk- en Buurtkaarten')(CBS and Kadaster, 2018). OPS output and the discussed population data are combined to calculate a population weighted mean concentration level for all three pollutants. A detailed description of all the technical steps in OPS-pro and QGIS that are required to calculate the population weighted mean concentration level is given in appendix B.

5.5 - Derivation of attributable number of cases

The health impacts resulting from an increase in concentration levels of air pollutants can be obtained using concentration-response functions (CRFs), which are typically expressed as a relative risk (RR) per 10 $\mu g/m^3$ increase in concentration. The relative risk is a ratio between the incidence rate of an exposed population and an unexposed population. For example, compare two populations, one of which is exposed to an increase in concentration of a pollutant of 10 $\mu g/m^3$ more than the other population. For both populations a certain fraction will experience an impact (e.g. premature death). Say that 5 percent of the unexposed population experiences premature death and 6 percent of the exposed population experiences premature death, then the RR is calculated as the ratio of the two, 5/6 = 1.20. The correct interpretation of an RR of 1.20 is that the exposed population had 1.20 times the risk dying prematurely compared to the unexposed population. The RR is an empirically derived statistic obtained from epidemiological studies and is often reported with a 95% confidence interval (CI). However, studies that apply the RR for quantifying health impacts do not always report the CI.

To obtain damage indicators that can be assigned a monetary value (RADs, deaths, hospital admissions, etc.), the following expression can be used:

$$\Delta D = I * N * CRF * \Delta C$$

Where ΔD is the change in health impact (cases, hospital admissions, deaths etc.) due to increased exposure, I is the annual baseline incidence rate for morbidity and mortality, N is the exposed population, CRF is a coefficient that describes the expected increase in incidence rate per change in concentration (RR-1)* and ΔC is the change in concentration ($\mu g/m^3$). Note that the change in concentration has to be expressed in the same unit as the CRF. The CRF is often based on an increment of 10 $\mu g/m^3$, meaning that a change in concentration of 5 $\mu g/m^3$ needs to be included in the calculation as 0.5 and not 5. (simply dividing the change in concentration by the incremental change used for the CRF will suffice.)

Note that the CRF coefficient is equal to the RR-1, the RR is relative to a baseline incidence, in order to calculate the change in health impacts, the baseline health impact needs to be subtracted from the new total. However, calculating with RR (per $\mu g/m^3$) and a change in concentration gives some complications. When the change in concentration is twice that of unit of the RR, the product of the two gives incorrect results: say that RR is 1.10 per 10 µg/m^3 and the increase in concentration is 20 µg/m^3, multiplying would give a coefficient (CRF) of 2.20, which is incorrect. The correct CRF is 1.20 ((0.10 *2)+1).

The RR is sometimes reported as a percentage instead of a ratio (e.g. DEFRA, 2019). In this case, converting to a ratio is required before calculating damages. In some cases, the baseline incidence rate (I) is incorporated in the impact coefficient (CRF) (e.g. Brandt et al., 2013). The CRF coefficient is then not expressed as a relative risk (ratio per µg/m^3), but rather as health impact (cases) per µg/m^3 increase in concentration. The calculation simplifies to:

$\Delta D = N * ERF * \Delta C$

Where the ERF coefficient is expressed as number of cases per µg/m^3. The benefit of this approach is that the ERF coefficient is much more intuitive than the RR. Of course, the ERF is still derived from the RR. For example, say we have a population of 1000 that experiences a mortality rate of 0.005 (5 deaths in total) in a given year and a CRF, expressed as a RR of 1.10 per 10 µg/m^3 for a certain pollutant (see table 6). The impact of an increase in concentration of 20 µg/m^3 is calculated as follows:

$\Delta D = 0.005 * 1000 * (1.10-1) * 2 = 1$

One additional death can be attributed to an increase in exposure of 20 µg/m^3.

Table 6. Example data for baseline and increased exposure.

	Baseline exposure	Increased exposure (+20 µg/m^3)
Mortality rate (I)	0.005	(5 + ΔD)/1000
Population (N)	1000	1000
Exposure	X	X + 20 µg/m^3
Cases (ΔD)	5	5 + ΔD

5.5.1 - Health effects data

The most comprehensive recommendations for concentration-response functions for PM, NO2 and O_3 are provided by the World Health Organization’s regional office for Europe in the “health risks of air pollution in Europe – HRAPIE” project (2013). Concentration-response functions are presented for 20 combinations of the three pollutants and health effects. These effects include mortality and morbidity indicators such as hospital admissions and restricted activity days (RADs). The relevant health effects and their corresponding recommended CRFs, as relative risk, are summarized in table 7. In total, thirteen health effects are considered, ten of which are related to particulate matter and three of which are related to nitrogen dioxide.

Table 7. Relative risk coefficients for health effects.

Pollutant	Health effect	Group	Relative risk	Lower	Upper
PM2.5	Respiratory hospital admissions	A*	1.0190	1	1.038
PM2.5	CVD hospital admissions	A*	1.0091	1.0017	1.0166
PM2.5	All cause mortality	A*	1.0620	1.04	1.083
PM2.5	Restricted Activity Days (RADs)	B*	1.0470	1.0094	1.0846
PM2.5	Work days lost	B*	1.0460	1.0092	1.0828
PM2.5	Mortality	A	1.0123	1.0045	1.0201
PM10	Incidence of asthma symptoms in asthmatic children	B*	1.0280	1.0056	1.054
PM10	Incidence of chronic bronchitis	B*	1.1170	1.0234	1.2106
PM10	Prevalance of bronchitis	B*	1.0800	1	1.16
PM10	Post -neonatal infant mortality	B*	1.0400	1.008	1.072
NO2	Respiratory hospital admissions.	A*	1.0180	1.0115	1.0245
NO2	All cause mortality	B*	1.0550	1.011	1.099
NO2	Bronchitic symptoms in asthmatic children	B*	1.0210	1	1.042

While experts working on the HRAPIE-project agreed that there is sufficient evidence to recommend all CRFs, some CRFs are more reliable than others. To represents the discrepancy in reliability, they categorized the CRFs in two groups: A and B. Group A are CRFs for which enough data is available to reliable quantify the effects of increased concentration levels. Group B are CRFs for which there is more uncertainty about the precision of the data used. The practical consequence of this distinction is how the lower and upper bound are established. All CRFs are statistically determined and thus come with 95% confidence intervals (see table A2 in appendix A). For most of the group A CRFs, the 95% confidence interval can be directly used for the lower and upper bound. For group B CRFs, it was concluded that a range of +/-80% around the central estimate should be adopted. Finally, for three CRFs used in this thesis the lower range of the 95% confidence interval is below 1. Another way of phrasing this is that the 95% confidence interval exceeds 100% of the central estimate. A relative risk below 1 implies that an increase in exposure would have a beneficial health impact. However, none of those involved with the HRAPIE-project considered this to be real possibility. Instead, a range of +/-100% around the central estimate is recommended to establish the lower and upper bound. A consequence of adopting this range is that the lower boundary defaults to 1. In order not to skew the uncertainty range, the upper boundary is also limited to +100% of the central estimate. Employing these conditions results in the lower and upper boundaries as they are presented in Table 7.

The asterisks refer to CRFs that contribute to the quantification of the total health impact. Only one health effect (i.e. mortality from short term $PM_{2.5}$ exposure) should be excluded from the quantification of the total health impact. The reason for this is that the effects are already accounted for in the 'all-cause mortality' category. Therefore, mortality from short term $PM_{2.5}$ is included for information only. Additionally, the health effects 'work days lost' and 'incidence of asthma symptoms in asthmatic children' should be subtracted from 'restricted activity days' for the same reason.

Some other sources for CRFS includes those reported by the Department for Environment, Food and Rural Affairs (DEFRA) in the UK, which report relative risks for 14 combinations of pollutants and effect (DEFRA, 2019). For the Netherlands specifically, Fischer et al. (2015) provide hazard ratios for four impact categories for PM and NO2.

5.5.2 - Incidence rates and population data

Besides the CRFs, data on the population at risk and the rate at which health effects currently occur are required to estimate the health impacts that would result from a given increase in concentration of a pollutant. All health effects have a specified range of the population that is at risk. These ranges partially depend on the health effect (e.g. by definition, post-neonatal infant mortality only affects infants aged 1 month to 1 year) as well as the underlying epidemiological studies that estimated that CRFs. Some health effects only apply to a narrow range of the population while other health effects, such as hospital admissions, apply to the entire population that is exposed. A summary of these ranges as well as the amount of individuals that fall within these ranges in the Netherlands is provided in appendix A (Table A3).

The HRAPIE-project itself does not report incidence rates for the health effects, but in a supporting document aimed at demonstrating how the HRAPIE-project recommendations can be implemented in cost-benefit analyses for the European Commission, Holland (2014) provides incidence rates that can be used within Europe. Some of these incidence rates have been adopted in this thesis, in particular the incidence rates for the more specific health effects. The reason for this is that finding incidence rates specifically for the Netherlands proved to be impossible. For the remaining incidence rates, mortality and hospital admission rates, data from Statistics Netherlands was used. In 2017, the total amount of deaths from natural causes (ICD-10 codes A00-R99) was 142,243 across all ages, this is equivalent to around 83 deaths per 10,000 people. Using the same approach for cardiovascular and respiratory hospital admissions gives incidence rates of around 136 and 78 per 10,000 people, respectively. Both the mortality and hospital admission rates are consistent with the ones reported by Holland (2014).

5.6 - Valuation

The final step in the impact pathway approach is the valuation of the health impacts. Monetary valuation is based on individual preferences, typically revealed through the choices they make in terms of consumption and the market price they pay (Bickel and Friedrich, 2001). Impacts such as days spent not working can be estimated by the average wage. For mortality, different methods have been explored. For example, differences in wage for risky and non-risky jobs can be used to estimate the value of avoiding mortality. However, for most aspects of health impacts, market prices are not available and instead monetary values for avoiding mortality and morbidity are derived from contingent valuation surveys. A willingness to pay or avoid health risks can be implied from such surveys. While values measured from individual studies have limitations, collectively they provide an accepted range of values that can be used for monetizing health benefits (Hall et al, 2010). The next sections describe values for the relevant health impacts in more detail.

5.6.1 - Mortality

With respect to mortality, two metrics are relevant: the value of a statistical life (VSL) and the value of a statistical life-year (VOLY). The VSL is appropriate for acute mortality, whereas the VOLY is more appropriate for chronic mortality (Brandt, 2013). The rationale being that acute mortality includes deaths that can occur at any time during someone’s life. Within transport economics, it is common to use the VSL to value transport victims. On the other hand, the primary victims from air pollution are the elderly as a result of chronic exposure and latency time lag. Valuing a reduction in life expectancy is more appropriately done by accounting for the years of life lost, rather than accounting for an entire statistical life. According to Bickel and Friedrich (2001), the VOLY can be derived by dividing the VSL by the life expectancy. In contrast, Brandt et al (2013) derived a factor of 27 between VSL and VOLY. The matter of valuing a human life, even statistically and ex ante (before damage takes place), is not without controversy. The willingness to pay or avoid health risks is presumable related to income levels and other factors, implying that a different VSL applies to different groups of people, which can be considered immoral. It has been the position of EU to use single base values for the VSL and VOLY EU wide.

Establishing base values for the VSL for a single country is difficult. OECD guidelines (OECD, 2012) recommend the use of USD (2005) 3.6 million for the EU-27. This value is based on a meta-analysis performed by the OECD. Converting to euros using the PPP for the Netherlands and adjusting to current prices using the Consumer Price Index results in a value of € 4 million, in 2018 prices. Real GDP per capita increased by 22.7% percent over the period 2005-2018, further increasing the VSL to € 4.9 million, in 2018 prices. In a different report on air pollution from road transport, the OECD (2014) provides country specific VSL's. For the Netherlands, a value of USD (2010) 3.761 million is given. Exchanging to euros and adjusting for inflation gives a value of around € 3.3 million (2017 prices). No values are given for the VOLY. Another source for the VSL and other unit values for health outcomes is Holland (2014). These unit values are particularly useful as they are specifically determined for the health effects recommended by the HRAPIE-project. These values are used for a cost-benefit analysis of the Clean Air Policy Package adopted by the European Commission. A value between 1.09 and 2.22 million euros is given for the VSL (2005 prices). Values for VOLYs of between € 57,700 and € 133,000 (2005 prices) are also given. Based on these values, the ratio between VSL and VOLY is around 18. Finally, Brandt et al. (2013) used values for VSL and VOLY of € 2.1 million and € 77,000 (2006 prices), respectively.

The unit value for the VSL provided by the OECD (2014) will be the main value used in this thesis, as it is based on an extensive study on the VSL (OECD, 2012), but other values will be used as well to give a range of the estimated health costs. Because the OECD provides no guidelines for the conversion from VSL to VOLY, a factor of 27 is used to determine a VOLY of € 122,222 (2017 prices). While this unit value is high compared to other VOLY discussed here, it still falls within the range provided by Holland (2014).

5.6.2 - Morbidity

For morbidity there are no recommended values available from the OECD, or really anywhere. In fact, there is not much consensus regarding valuing morbidity. The primary reason for this is the large variety in morbidity health endpoints (OECD, 2014). Some clarity can be obtained by distinguishing between three cost categories, the sum of which represent total economic costs of health impacts from air pollution: resources costs, opportunity costs and disutility costs. Resource costs include the treatment costs as well as non-medical costs that are incurred because of the illness. Opportunity costs refer to foregone wage or utility from leisure. Finally, disutility costs refer to the pain, suffering and other forms of discomfort linked to the illness. Morbidity costs are thus calculated as a sum of separate elements. Which elements are included in this thesis is mostly dependent on the availability of data on CRFs and monetary values.

5.6.3 - Hospital admissions

To give an indication of the range of values in use, several values are presented. On the low end, Holland (2014) gives a value of € 2,200 (2005 prices) for both respiratory as well as cardiovascular hospital admissions. Brandt et al. (2013) provide a unit cost only for respiratory hospital admissions. Their estimate is based on a cost-of illness approach and gives a value of € 7,931 (2006 prices). Based on WTP research by Chilton et al. (2004), DEFRA (2019) applies values of around € 9,750 (2017 prices) for respiratory and cardiovascular hospital admissions. Finally, on the upper end, a comprehensive study on the cost of hospitalisation conducted by Chestnut et al. (2006) in California estimates that the total average cost of illness ranges between $ 22,000 and $ 39,000 (2002 prices). While these values cannot be directly compared without accounting for exchange rates and inflation, it does indicate that the range of values is considerable.

5.6.4 - Other health effects

The value for chronic bronchitis in adults is based on two values There seems to be a relatively large degree of agreement on what the unit value for chronic bronchitis in adults is. RADs can be categorized into two groups: minor restricted activity days (MRADs) and work loss days. Brandt et al. (2013) assume that 40% of all RADs are work loss days. The WHO recommendations instruct to subtract work loss days from RADS, indicating that remaining RADs only refer to MRADs. To properly add all values, days spent in the hospital should also be subtracted, per admission around 6.3 days should be subtracted from RADs.

5.6.5 - Final unit values

For choosing the central unit value for all health effects, the following considerations are applied. First, the values provided by Holland (2014) are determined specifically for the health effects for which the HRAPIE-project recommends CRFs. Therefore, these unit values are considered reliable and consistent. Second, the VSL and the VOLY are based on the VSL provided by the OECD as this value is the result of extensive research into the topic. Third and finally, the unit value for hospital admissions provided by Holland (2014) appear to be based on the 'hotel cost' of a hospital admission, which does not include treatment cost and the opportunity and disutility cost of the patient. As a result, they are lowest values discussed here. Instead of using these values as the central value, they will be used as the lower boundary. The other two values (DEFRA, 2019; Brandt et al.,2013) are both approximately four times larger. The lower unit value of the two will be used as the central value and the higher value will be used as the upper boundary. Based on these considerations a central unit value per health effect was determined as well as an lower and upper boundary. The final unit values are presented in table 8. A complete overview of all the values used to determine the central, lower and upper values, except for the VSL and VOLY, are presented in the appendix (table A6).

Table 8. Unit values for all health effects.

Health effect	Central unit value (2017)	Lower	Upper
Respiratory hospital admissions	9,404	2,637	9,750
CVD hospital admissions	9,404	2,637	9,750
All cause mortality	122,222	49,655	159,440
Restricted Activity Days (RADs)	110	110	155
Work days lost	156	156	156
Mortality	122,222	25,663	159,440
Incidence of asthma symptoms in asthmatic children	50	50	128
Incidence of chronic bronchitis	64,256	62,797	64,256
Prevalance of bronchitis	588	588	588
Post -neonatal infant mortality	3,300,000	1,600,000	3,956,040
Respiratory hospital admissions.	9404	2,637	9,750
All cause mortality	122,222	49,655	159,440
Bronchitic symptoms in asthmatic children	588	588	588

6 - Results

The results will be discussed in three sections. First the results from the dispersion modelling will be presented. They will give insight into how much waste incinerators contribute to overall concentration levels of PM and NO_2. Next, the additional health effects that can be attributed to waste incinerator emissions are presented. Finally, the monetized costs associated with the health impacts are presented. This will include a breakdown of overall costs as well as costs per pollutant and costs per ton of waste.

6.1 - Exposure

Based on the emission data and location of waste incinerators, the dispersion of PM and NO_2 could be modelled across all the land area in the Netherlands. The map in figure 8 shows the twelve incinerator locations and the concentration level for $PM_{2.5}$ in the Netherlands. The concentration levels range from around 0.00022 μg/m^3 in the south-west, far away from any waste incinerator to around 0.00121 μg/m^3 in the immediate vicinity of the incinerators. As a result of the prevailing wind direction, emissions appear to disperse in a mostly north-east direction from the incinerators. Keeping this in mind, the contribution of the two northern most incinerators (Harlingen and Delfzijl) seems negligible as emissions are transported and deposited over seas or across borders. Furthermore, most of the emissions appear to deposit or disperse relatively soon after emissions, as the dark red areas often do not exceed an approximately 10 km radius around the incinerator (except for the Amsterdam region). The population weighted mean concentration level is 0.00074915 μg/m^3 and is used to calculate the health impacts.

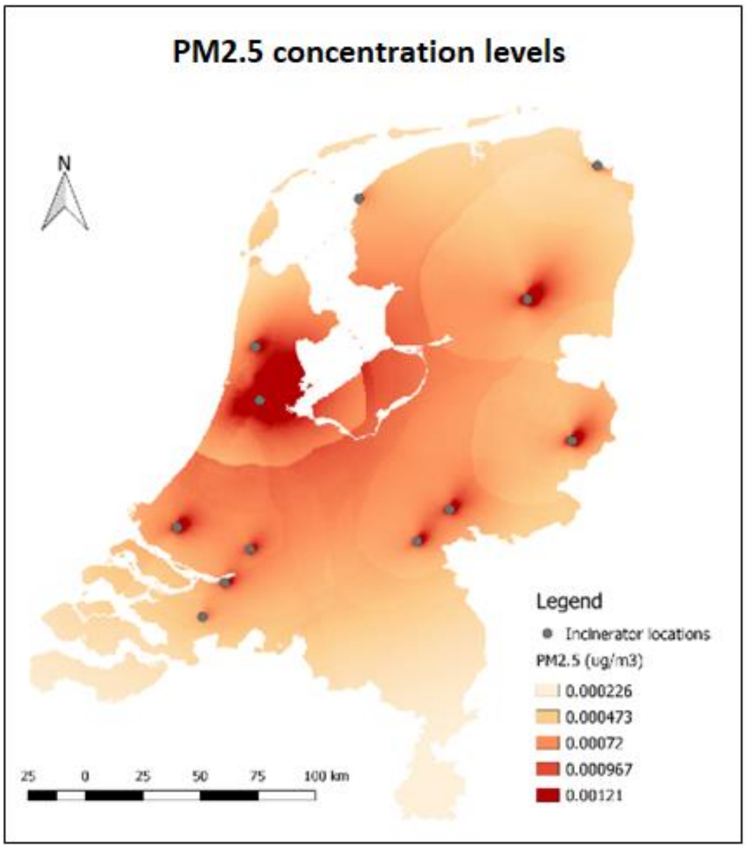

Figure 8. $PM_{2.5}$ concentration levels.

As discussed, $PM_{2.5}$ is a subcategory of PM_{10} and in the case of incinerator emissions, it even makes up the vast majority of the PM_{10}. This is also visible in the two PM maps, patterns of high and low concentration levels are nearly identical. Although the overall concentration levels are slightly higher for PM_{10}, the maps for $PM_{2.5}$ and PM_{10} are essentially the same, as is evident from all the patterns around the waste incinerators in both figures (see figures 8 and 9). Due to a disproportional amount of PM emissions from the Amsterdam incinerator, both maps show a large area around Amsterdam for which concentration levels are particularly high. The resulting concentration levels are in the range of 0.00024 to 0.00132 µg/m^3. The population weighted mean concentration level used for the calculation of the health impacts is 0.000813695 µg/m^3.

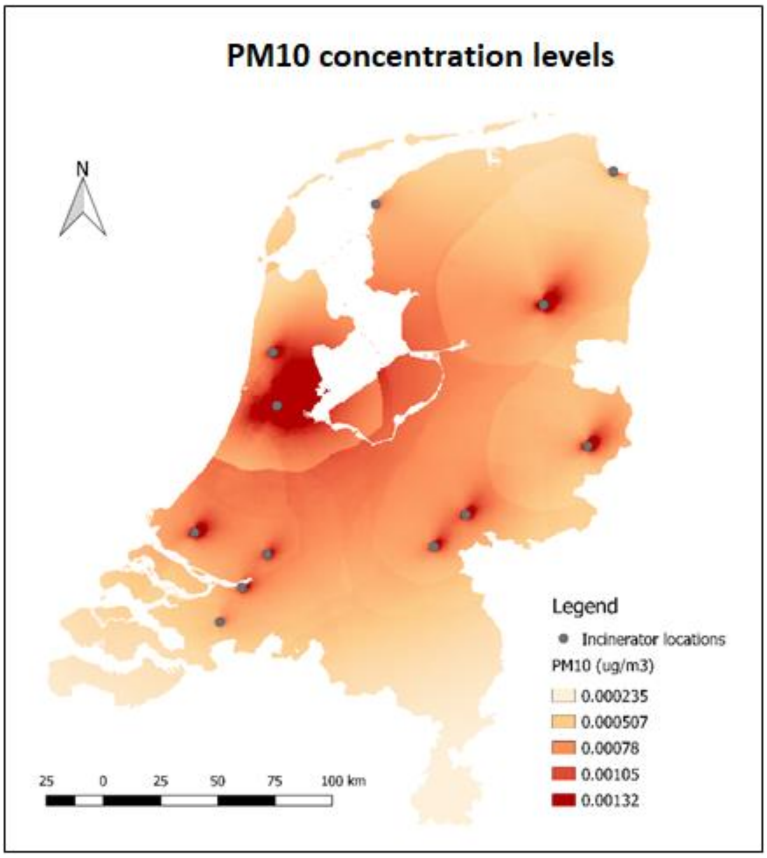

Figure 9. PM_{10} concentration levels

The resulting map for NO_2 shows similar features due to the underlying meteorological data (see figure 10), but also shows some distinct features. Most notably, the large radiuses around two incinerators in the south-west. Similar to the PM maps, a large dark red area with high concentration levels can also be identified in the Amsterdam region. The incinerator in Amsterdam processed the most amount of waste and emitted the most amount of PM and NO_2 (see tables 4 and 5). The concentration levels range from around 0.0212 to around 0.107 µg/m^3, with a populated weighted mean concentration level of 0.06795 µg/m^3.

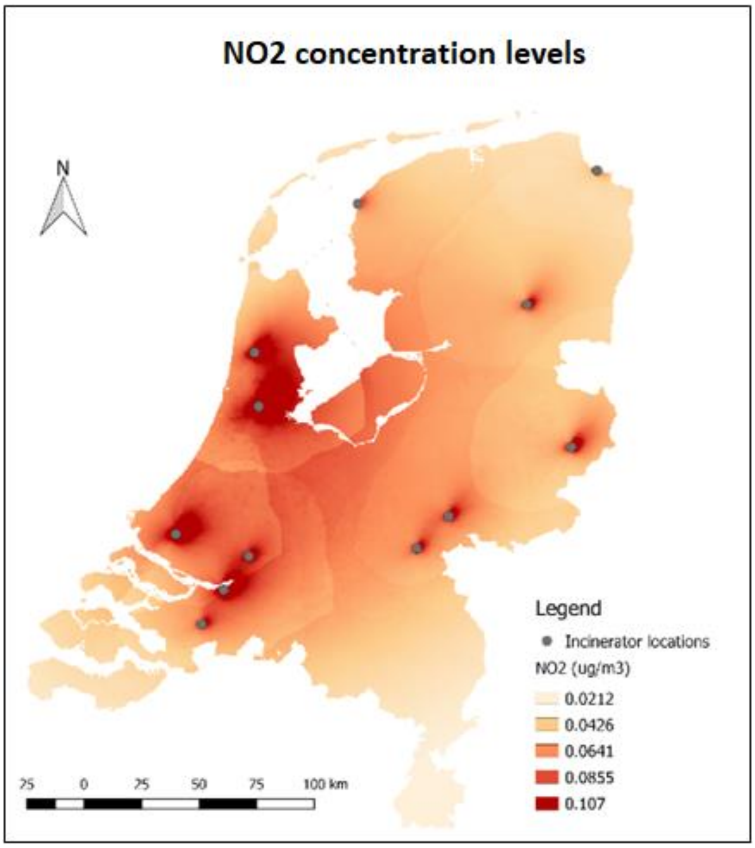

Figure 10.

6.2 - Health impacts

Having calculated the population weighted mean concentration levels for all pollutants means that the health effects can be estimated using the equation discussed in section 5. The results of the health impacts from waste incinerator emissions are summarized in table 9. Included are a central estimate 'additional cases' as well as a lower and upper boundaries, which are based on the uncertainty of the relative risk coefficients. Several health effects are only increasing by one unit, indicating a low impact. For example, $PM_{2.5}$ emissions from waste incineration result in less than 0.5 hospital admissions. For mortality, the numbers are similar. For PM_{10} the impacts appears to be similarly low. It should be noted that while the impact for post- neonatal infant mortality is listed as 0.00, the actual number is not zero, but 0.0019 instead. For NO_2, the impacts 'respiratory hospital admissions' and 'all cause mortality' are roughly a factor 100 higher compared to $PM_{2.5}$. The results indicate that due to the NO_2 emissions around 36 life years are lost. In all cases, the central estimate is in the middle of the lower and upper boundaries. For three health effects the lower boundary suggest that there could be no impact. The combined loss of life years due to $PM_{2.5}$ and NO_2 is 7.39 at the lower boundary to 64.55 at the upper boundary.

Table 9. Health effects as a result of emissions from waste incinerators in 2017.

Pollutant	Health impact	Unit	Additional cases	Lower	Upper
PM2.5	Respiratory hospital admissions	cases	0.19	0.00	0.38
PM2.5	CVD hospital admissions	cases	0.16	0.03	0.29
PM2.5	All cause mortality	life years lost	0.44	0.28	0.59
PM2.5	Restricted Activity Days (RADs)	days	1149.39	229.88	2068.91
PM2.5	Work days lost	days	356.36	71.27	641.44
PM2.5	Mortality	life years lost	0.13	0.05	0.22
PM10	Incidence of asthma symptoms in asthmatic children	days	1.06	0.21	2.04
PM10	Incidence of chronic bronchitis	cases	0.44	0.09	0.79
PM10	Prevalance of bronchitis	cases	1.36	0.00	2.72
PM10	Post -neonatal infant mortality	deaths	0.00	0.00	0.00
NO2	Respiratory hospital admissions.	cases	16.12	10.30	21.94
NO2	All cause mortality	life years lost	35.53	7.11	63.96
NO2	Bronchitic symptoms in asthmatic children	cases	11.39	0.00	22.78

6.3 - Health costs

Multiplying the health effects with the unit values as presented in table 8 gives total health costs as well as a lower and upper estimate (see table 10). Note that for the total costs, the lost days from other categories (e.g. hospital admissions) are subtracted from the RADs. The total health cost using the central estimate and unit value adds up to around € 4.7 million (2017 prices). The primary contributor to this total is the life years lost due to NO_2 emissions (€ 4.3 million). Respiratory hospital admissions due to NO_2 (€ 151,592) and RADs due to $PM_{2.5}$ (€ 126,433) are the two subsequent health impacts in terms of highest estimated health costs.

Table 10. Health costs associated with emissions from waste incinerators.

Pollutant	Health effect	Health costs (€ 2017)	Lower	Upper
PM2.5	Respiratory hospital admissions	1,787	0	3,705
PM2.5	CVD hospital admissions	1,505	79	2,828
PM2.5	All cause mortality	53,778	13,903	94,070
PM2.5	Restricted Activity Days (RADs)	126,433	25,287	320,681
PM2.5	Work days lost	55,592	11,118	100,065
PM2.5	Mortality	15,889	1,283	35,077
PM10	Incidence of asthma symptoms in asthmatic children	53	11	261
PM10	Incidence of chronic bronchitis	28,273	5,652	50,762
PM10	Prevalance of bronchitis	800	0	1,599
PM10	Post -neonatal infant mortality	6,270	640	13,451
NO2	Respiratory hospital admissions.	151,592	27,161	213,915
NO2	All cause mortality	4,342,548	353,047	10,197,782
NO2	Bronchitic symptoms in asthmatic children	6,697	0	13,395
	Total	**4,740,486**	**423,159**	**10,925,772**

The lower and upper boundaries are based on both the uncertainty in the relative risk coefficients as well as the range of the unit values (see table 8). The lower and upper boundaries represent a large range of € 423,159 - € 10,925,772.

Based on the previous results, several other informative numbers can be determined (see table 11). First, health costs per pollutant can be obtained. Once again, it reveals that NO_2 is that primary source of the estimated health costs and PM_{10} contributes the least (€ 35,396). Even after converting to costs per kg emissions, NO_2 is associated with higher health costs than PM_{10}. However, according to the cost per kg metric, $PM_{2.5}$ is by far the most impactful pollutant with a cost of € 5.96 per kg emission. Finally, The health costs per ton of waste can also be calculated and add up to € 0.61 per ton of waste. Due to the significantly higher emission factor of NO_2, the costs per ton of waste are also dominated by NO_2, making up around 95 percent of the costs per ton of waste.

Table 11. Costs for different categories in 2017 euros.

Pollutant	Cost per kg emissions	emissions	Cost per pollutant	Cost per ton waste
PM2.5	5.96	34,298	204,254	0.0246
PM10	1.02	34,762	35,396	0.0043
NO2	1.47	3,063,456	4,500,837	0.5843
Total			**4,740,487**	**0.6132**

7 - Discussion

This section will first discuss the results in more detail and in relation to other studies. Next, limitations and uncertainties will be discussed. Finally, the issue of double counting will be considered.

7.1 – interpretation of results

In terms of particulate matter emissions, Dutch waste incinerators are relatively, and perhaps, surprisingly clean, emitting around 35 tons of PM in a year. Although, it should be noted that the average emission factor for PM in the Netherlands is around 0.0041 kg per ton of incinerated waste, which is quite high compared to the recommended emission factor of 0.003 kg per ton of waste. While the average is somewhat skewed due to the high emission factor of the incinerator in Amsterdam (0.010 kg/ton), only five incinerators report an emission factor below 0.003. Either the recommended emission factor is rather optimistic or Dutch waste incinerators are relatively polluting compared to other waste incinerators. Once dispersed in the atmosphere across the entire Netherlands and beyond, the emissions contribute less than 0.008 percent to the total average $PM_{2.5}$ concentration level. It may not come as a surprise then that health impact in terms of attributable cases due to PM is low. For most health effects, around 1 or less than 1 case could be attributed, which in the worst case represents a loss in life years. For arguably the least severe health effect, RADs, the central estimate considerably higher than 1. A total of 1149 restricted activity days are estimated to be the result of $PM_{2.5}$ emissions. However, this represents roughly 6 seconds per person per year. While zero attributable cases are better than a small number of cases, waste management is a necessity and alternative waste management options are likely to also have nonzero health impacts. The total health costs associated with PM are € 239,650, which represents around 5 percent of the total health costs.

Instead, the majority of the health impacts and costs are attributable due to NO_2 emissions. The average emission factor for Dutch incinerators is 0.39 kg per ton of waste, which is low compared to the recommended 1.071. Contrary to the PM emission factors, all waste incinerator have an emission factor for NO_2 that is close to the average. Nevertheless, over 3 million kg of NO_2 is emitted from Dutch waste incinerators. Despite the fact that per kg of emissions, $PM_{2.5}$ are associated with much higher health costs, the much higher NO_2 emissions, make it the primary contributor. The combined NO_2 emissions have an estimated health cost of around €4.5 million, with an upper boundary approach € 10.2 million.

With total costs per ton of waste of € 0.61, investing in further emission abatement seems hardly worth the financial cost. Especially considering that current air pollution control technologies are already expensive. On the other hand, introducing a tax or fee per ton of waste or per kg of emission would generate a tax revenue equal to the total health costs, which could be used for further abatement as well as incentivize a reduction in the amount of waste that is produced.

An approach as the one in this thesis is often used estimation of health impacts of large (EU scale) projects or sectors (Holland, 2014; OECD 2016; Brandt et al., 2013). On the other hand, studies examining the direct impact of incinerators on human health have focussed mainly on dioxins (Giusti, 2009). Few studies have assessed one or multiple waste incinerators and modelled the emissions and resulting exposure. Ashworth et al. (2016) modelled the emissions of PM_{10} from two waste incinerators in the UK. While only looking at an area around the incinerators with a radius of 10 km, they found that the incinerators contributed at most less than 0.01 µg/m^3. The mean concentration level they found was 0.000117 µg/m^3, also noting that this value is extremely low. While this mean concentration level is lower than the ones reported in this thesis, both results are reasonably in line with each other, especially after considering differences in the total amount of waste incinerated.

In 2013, it was estimated that 1,628 people died prematurely due to PM_{10} exposure (CBS, PBL, RIVM, WUR, 2015). Prematurely in this case refers to several days or weeks. The results presented in this thesis estimate that 0.44 life years were lost due to premature mortality as a result of $PM_{2.5}$ emissions. While the life years lost due to incinerator emissions seem low, compared to the estimated premature deaths associated with PM_{10} exposure, they are certainly non-negligible and perhaps high.

No studies could be found that modelled the exposure and estimated and monetized the health effects. However, Eshet et al. (2005) reviewed valuation studies of incineration externalities and compiled a list of unit values (USD per kg of emission), which includes NO_x and PM_{10}. While the results presented in this thesis are in the lower end of the range provided by Eshet et al. (2005), they are still reasonably comparable.

7.2 - Limitations

Along the entire pathway from waste incinerators to monetized health effects, choices have been made that reduced the scope of this thesis, but likely also limited the total impact of waste incinerators on human health that would be considered. Only PM and NO_2 were considered, while other pollutants such as dioxins and various heavy metals have been ignored. Furthermore, only the effects through air pollution were considered, while exposure to food from contaminated soils or exposure to contaminated water was also ignored.

Because many air pollutants can travel large distance trough the atmosphere, it would be appropriate to approach this topic from a transnational perspective, ignoring national boundaries. The effect that emissions from Dutch waste incinerators has on the air quality and, subsequently, the health of the population of neighbouring countries would ideally be considered as well. Likewise, the impact from foreign waste incinerators on the Dutch population would also be valuable information. However, in this thesis no such considerations are made. Further research from a transnational perspective would be able to provide a more complete overview.

7.3 - Uncertainties

There are many steps involved in the entire process, all of which involve uncertainties and give rise to potential inaccuracies. First of all, the emissions may not be monitored correctly. However, they emission data is verified by official organizations and used for a wide range of official documents. Furthermore, the data is directly and continuously measured from incineration plants as way monitor the emissions and ensure compliance to regulations. While some inaccuracies may occur, it seems unlikely that the reported emissions data would differ substantially from the actual emissions. Similar to the emissions data, the data on the amount of waste incinerated at each incinerator is also expected to be reliable. Besides, inaccuracies in the waste numbers do not influence the results on total health costs.

In contrast to the emissions data, the remaining steps all involve considerable more uncertainty. The dispersion modelling software is likely to be a major source of uncertainty. There are many models in use that all work slightly different and should be used for different purposes (e.g. LOTOS-EUROS and ADMS 3). These models are complex and often require simplification, which introduces inaccuracies. The OPS model has been in use for a long time and has been validated (Sauter et al., 2018). Sensitivity and uncertainty analysis showed that on a national scale, the results can deviate around 15 percent compared to measured results. The OPS model is deemed suitable as it was developed for the Dutch context, which means that it was calibrated for the Netherlands and has pre-prepared meteorological and surface roughness data. The sensitivity of the modelling output has also been tested by simulating a reduction in emissions of 10 percent in all waste incinerators. For such small quantities at least, a reduction in emissions of 10 percent resulted in a population weighted mean concentration level of 10 percent lower. This was the case for all pollutants. Considering that the results are calculated through simple multiplication, all changes are linear. In conclusion, a percentage change in the emissions will result in a change in health impacts and costs equal to that percentage.

The next major source of uncertainties are the concentration-response functions. While the CRFs recommended by the WHOs HRAPIE-project are based on a large amount of studies, they are not without flaws or uncertainties. Most of the uncertainties have been accounted for by using a lower and upper range of the CRFs, which are based on the 95 percent confidence intervals. Furthermore, the CRFs have been deemed suitable for application within Europe, but Europe is not homogenous and differences within countries occur. However, quantifying these differences has not been possible. Another source of uncertainty involves the incidence rates used in this thesis. For several specific health effects (e.g. RADs), incidence rates from other studies in other countries have been used. Finally, the health effects considered in this thesis are included because sufficient evidence was available to quantify the relation between emissions and health impact, but PM and NO_2 may very well have other health effects which are not included due to insufficient evidence or unawareness. The total health effects may then be underestimated.

The final source of uncertainty relates to the valuation of the health impacts. An attempt was made to reduce this uncertainty by considering a range of unit values. The unit values by Brandt et al. (2013) and Holland (2014) can be used within the EU, but differences among EU populations exist and more accurate values for the Dutch population specifically may improve the results.

7.4 - Double counting

Concentration levels of pollutants are often correlated to some extent, because they are emitted as a result of the same process. Estimating the effect of a single pollutant may then overestimate the effect, as some of the effect is attributable to other pollutants. Health outcomes such as premature mortality are caused by both PM and NO_2 exposure. Epidemiological studies that attempted to quantify the relation between the pollutants and the health outcome have not been conducted in controlled environments where only one pollutant was present. Simply adding the health outcomes or valuation of both pollutants will therefore likely overestimate the true impact. On the other hand, considering only the impacts of one pollutant, while more pollutants with the same health outcome are present, will likely underestimate the true impact. For the purpose of this thesis, health impacts of PM and NO_2 will be reported separately as well as aggregated, but it should be noted that the aggregated values will likely involve some double counting. While currently no specific guidelines exist for the proper aggregation of health impacts of both pollutants, a report by the Royal College of physicians (2016) provides an example of such aggregation. It reports number of attributable deaths of 28,861 and 23,500 for $PM_{2.5}$ and NO_2, respectively, but adopts 40,000 deaths as the combined effect. This represent a reduction of 23.6 percent of the combined deaths. The HRAPIE-project mentions that there could be an overlap of up to 33 percent. While both numbers are somewhat arbitrary, it gives an indication of the expected overlap. The overall conclusion appears to be that while there will be some overlap between $PM_{2.5}$ and NO_2, excluding on or the other will underestimate the total effects by more than the potential overlap.

8 - Conclusion

Waste incineration has become the predominate disposal method for most of our waste. The process of incineration reduces the volumes substantially, but also produces many pollutants that are emitted into the atmosphere. Through inhalation of these pollutants, humans experience adverse health effects. In this thesis, an attempt has been made to quantify those adverse health effects, both in terms of amount of cases as well as monetary costs. Most of the health impacts in terms of life years lost and hospital admissions are due to NO_2 emissions. The central estimates for hospital admissions and life years lost are 16.12 and 35.53, respectively. In comparison, the total health impact of PM is much lower. This is mainly due to the relatively small amounts of PM emissions from incinerators, at least compared to the amount NO_2 that is emitted.

The total health costs are estimated at around € 4.7 million (in 2017 prices). However, the uncertainty is large as it ranges from € 423,519 to € 10,925,772. The vast majority of the health costs can be attributed to the loss of life years due to NO_2 emissions. All remaining health costs are merely fractions of the total health costs. However, on per kg of emissions basis, $PM_{2.5}$ have the highest associated health cost, by a significant margin, € 5.96 compared to € 1.02 (PM_{10}) and € 1.47 (NO_2). Based on the average emission factors, a cost per ton of waste could also be calculated, which is € 0.61 per ton of waste.

In this thesis, the emissions from foreign waste incinerators have been ignored, as well as the contribution of emissions from Dutch incinerators to foreign concentration levels. For a more complete overview of the effects of emissions from waste incinerators on human health, future research that takes a more transnational approach may provide useful results. Furthermore, any research that attempts to accurately quantify the relation between exposure and health effects would be valuable for estimating health impacts from certain sectors as well as for new projects.

Based on the total health costs presented in this thesis, the benefits associated with reducing emissions are unlikely to outweigh the high investment costs associated with modern abatement technology.

9 - References

Ashworth, D. C., Fuller, G. W., Toledano, M. B., Font, A., Elliott, P., Hansell, A. L., & de Hoogh, K. (2016). Comparative assessment of particulate air pollution exposure from municipal solid waste incinerator emissions. *Air Quality,* 87-116. Apple Academic Press.

Bickel, P., & Friedrich, R. (2001). Estimating Environmental Costs using the Impact Pathway Approach. Retrieved on March 27 from: http://www.its.leeds.ac.uk/projects/unite/paris/bickel.pdf

Brandt, J., Silver, J. D., Christensen, J. H., Andersen, M. S., Bønløkke, J. H., Sigsgaard, T., ... & Hedegaard, G. B. (2013). Contribution from the ten major emission sectors in Europe and Denmark to the health-cost externalities of air pollution using the EVA model system–an integrated modelling approach. *Atmospheric Chemistry and Physics*, *13*(15), 7725-7746.

Brown, J. S., Gordon, T., Price, O., & Asgharian, B. (2013). Thoracic and respirable particle definitions for human health risk assessment. *Particle and fibre toxicology*, *10*(1), 1-12.

CBS, PBL, RIVM, WUR (2015). Gezondheidseffecten van fijn stof en ozon, 1992 - 2013 (indicator 0340, versie 11 , 13 mei 2015). Centraal Bureau voor de Statistiek (CBS), Den Haag; PBL Planbureau voor de Leefomgeving, Den Haag; RIVM Rijksinstituut voor Volksgezondheid en Milieu, Bilthoven; en Wageningen University and Research, Wageningen. Retrieved on July 3 from: https://www.clo.nl/indicatoren/nl0340-gezondheidseffecten-van-fijn-stof-en-ozon

CBS (Statistics Netherlands) (2018). Municipal Solid Waste: Quantities. Retrieved on May 13 from: https://statline.cbs.nl/Statweb/publication/?DM=SLNL&PA=83558NED&D1=1-36,109-119,121-122,173-177&D2=0&D3=18-24&HDR=G1,G2&STB=T&VW=T

CBS and Kadaster (2018). Wijk- en Buurtkaart 2017. Retrieved on June 4 from: https://www.cbs.nl/nl-nl/dossier/nederland-regionaal/geografische%20data/wijk-en-buurtkaart-2017

CBS (2019). StatLine: Gemeentelijke afvalstoffen, hoeveelheden. Retrieved on May 4 From: https://opendata.cbs.nl/statline/#/CBS/nl/dataset/83558NED/table?ts=1562839716703

Chilton et al. (2004) 'Valuation of health benefits associated with reductions in air pollution. Final report'. http://randd.defra.gov.uk/Document.aspx?Document=EP01006_4723_FRP.pdf

Costa, S., Ferreira, J., Silveira, C., Costa, C., Lopes, D., Relvas, H., ... & Paulo Teixeira, J. (2014). Integrating health on air quality assessment—Review report on health risks of two major European outdoor air pollutants: PM and NO2. *Journal of Toxicology and Environmental Health, Part B*, *17*(6), 307-340.

COWI (2000). A Study on the Economic Valuation of Environmental Externalities from Landfill Disposal and Incineration of Waste. Report to European Commission, DG Environment. Retrieved on May 3 from: http://ec.europa.eu/environment/waste/studies/pdf/econ_eva_landfill_report.pdf

DEFRA (2019). Impact Pathways approach: Guidance for air quality appraisal. Department for Environment, Food and Rural Affairs, London, UK.

Dijkgraaf, E., & Vollebergh, H. R. (2004). Burn or bury? A social cost comparison of final waste disposal methods. *Ecological Economics*, *50*(3-4), 233-247.

European Commission. (2008). Directive 2008/50/EC of the European Parliament and of the Council of 21 May 2008 on Ambient Air Quality and Cleaner Air for Europe. *Official Journal of the European Union.*

EEA (2016). Indicators - Waste Recycling. Retrieved on June 17 From: https://www.eea.europa.eu/data-and-maps/indicators/waste-recycling-1/assessment

EEA (2018). Air Quality in Europa – 2018 Report. European Environment Agency, Copenhagen, Denmark. doi: 10.2800/777411.

EMEP/EEA (2016). Air pollutant emission inventory guidebook 2016. EEA Report No 21/2016. Available from: https://www.eea.europa.eu/publications/emep-eea-guidebook-2016/part-b-sectoral-guidance-chapters/5-waste.

Eshet, T., Ayalon, O., & Shechter, M. (2005). A critical review of economic valuation studies of externalities from incineration and landfilling. *Waste management & research*, *23*(6), 487-504.

Giusti, L. (2009). A review of waste management practices and their impact on human health. *Waste management*, *29*(8), 2227-2239.

Hall, J. V., Brajer, V., & Lurmann, F. W. (2010). Air pollution, health and economic benefits—lessons from 20 years of analysis. *Ecological Economics*, *69*(12), 2590-2597.

Harris, J.M., & Roach, B. (2015). Environment and Natural Resource Economics – 3rd ed. Routledge, Ney York, NY.

Monks, P. S., Archibald, A. T., Colette, A., Cooper, O., Coyle, M., Derwent, R., ... & Stevenson, D. S. (2015). Tropospheric ozone and its precursors from the urban to the global scale from air quality to short-lived climate forcer. *Atmospheric Chemistry and Physics*, *15*(15), 8889-8973.

OECD (2012), "Recommended Value of a Statistical Life numbers for policy analysis", in Mortality Risk Valuation in Environment, Health and Transport Policies, OECD Publishing, Paris.

OECD (2014), The Cost of Air Pollution: Health Impacts of Road Transport, OECD Publishing, Paris, http://dx.doi.org/10.1787/9789264210448-en.

OECD (2016). The Economic Consequences of Outdoor Air Pollution, OECD Publishing, Paris.

Pelucchi, C., Negri, E., Gallus, S., Boffetta, P., Tramacere, I., & La Vecchia, C. (2009). Long-term particulate matter exposure and mortality: a review of European epidemiological studies. *BMC public health*, *9*(1), 453.

Pope III, C. A., & Dockery, D. W. (2006). Health effects of fine particulate air pollution: lines that connect. *Journal of the air & waste management association*, *56*(6), 709-742.

Porta, D., Milani, S., Lazzarino, A. I., Perucci, C. A., & Forastiere, F. (2009). Systematic review of epidemiological studies on health effects associated with management of solid waste. *Environmental health*, *8*(1), 60.

Rabl, A., Spadaro, J. V., & Zoughaib, A. (2008). Environmental impacts and costs of solid waste: a comparison of landfill and incineration. *Waste Management & Research*, *26*(2), 147-162.

Rijkswaterstaat (2018). Afvalverwerking in Nederland, gegevens 2017. National Institute for Public Health and the Environmental (RIVM), Utrecht, Netherlands.

Sauter, F., van Zanten, M., van der Swaluw, E., Aben, J., de Leeuw, F., & van Jaarsveld, H. (2018). The OPS-model, description of OPS 4.5.2. National Institute for Public Health and the Environmental (RIVM).

Silveira, C., Roebeling, P., Lopes, M., Ferreira, J., Costa, S., Teixeira, J. P., ... & Miranda, A. I. (2016). Assessment of health benefits related to air quality improvement strategies in urban areas: An Impact Pathway Approach. *Journal of environmental management*, *183*, 694-702.

van der Swaluw, E., de Vries, W., Sauter, F., Aben, J., Velders, G., & van Pul, A. (2017). High-resolution modelling of air pollution and deposition over the Netherlands with plume, grid and hybrid modelling. *Atmospheric environment*, *155*, 140-153.

Velders, G.J.M., Aben, J.M.M., Geilenkirchen, G.P., den Hollander, H.A., Megens, L., van der Swaluw, E., de Vries, W.J. & van Zanten, M.C. (2016) Grootschalige concentratie- en depositiekaarten Nederland. Rapportage 2016. Rapport 2016-0068, Rijksinstituut voor Volksgezondheid en Milieu, Bilthoven.

Velders, G.J.M., Aben, J.M.M., Geilenkirchen, G.P., den Hollander, H.A., Nguyen, L., van der Swaluw, E., de Vries, W.J., & Wichink Kruit, R.J. (2018). Large-scale Concentration and Deposition Maps The Netherlands Report 2018. RIVM Rep. 2018-0104. doi: 10.21945/RIVM-2018-0104.

WHO. (2006). Air quality guidelines: global update 2005: particulate matter, ozone, nitrogen dioxide, and sulfur dioxide. World Health Organization, Geneva, Switzerland.

WHO (2013). Recommendations for Concentration-response Functions for Cost-benefit Analysis of Particulate Matter, Ozone and Nitrogen Dioxide, Health Risks of Air Pollution in Europe - HRAPIE Project. World Health Organization (WHO) Regional Office for Europe, Copenhagen, Denmark.

WHO (2016). Ambient Air Pollution: A global Assessment of Exposure and Burden of Disease. World Health Organization, Geneva, Switzerland.

10 - Appendix

10.1 Appendix A

Table A1. Waste incinerator location and incinerated waste 2011-2017.

Name	Location	2011	2012	2013	2014	2015	2016	2017
REC	Harlingen	153,793	227,733	242,418	248,368	231,917	256,373	260,993
EEW Energy From Waste	Delfzijl	248,798	316,608	357,475	374,587	372,814	376,953	343,789
Attero Noord	Wijster	639,323	677,239	704,347	730,161	702,201	711,987	658,000
Twence	Hengelo	613,100	607,899	617,454	599,188	643,897	637,061	622,198
ARN	Weurt	261,050	294,060	285,596	291,937	303,608	276,002	272,384
HVC	Alkmaar	608,042	639,666	680,829	671,001	667,946	670,521	677,940
HVC	Dordrecht	288,308	301,039	310,266	308,747	307,291	310,525	290,801
AVR Rijnmond	Botlek Rotterdam	1,242,077	1,292,757	1,288,026	1,295,168	1,332,595	1,372,204	1,282,970
AVR BV	Duiven	397,498	383,093	385,733	378,614	380,336	386,973	390,340
AEB	Amsterdam	1,473,065	1,473,120	1,427,083	1,395,444	1,352,196	1,483,122	1,477,126
AEC	Moerdijk	984,807	924,087	910,135	962,271	913,409	958,479	1,017,110
SITA ReEnergy	Roosendaal	288,014	334,040	330,054	335,718	347,382	345,096	321,127
	Total	**7,197,875**	**7,471,341**	**7,539,416**	**7,591,204**	**7,555,592**	**7,785,296**	**7,614,778**

Table A2. Incinerator emissions and emission factors 2015.

		2015			emission factors		
Location	Waste (t)	NO2 (kg)	PM10 (kg)	PM2.5 (kg)	e_{fno2} (kg/t)	e_{fpm10} (kg/t)	$e_{fpm2.5}$ (kg/t)
Harlingen	231,917	84970	432	428.2	0.366381	0.001863	0.001846
Delfzijl	372,814	115200	1563	1550	0.309001	0.004192	0.004158
Wijster	702,201	210100	5678	5628	0.299202	0.008086	0.008015
Hengelo	643,897	271300	3216	3193	0.421341	0.004995	0.004959
Weurt	303,608	134400	1300	1289	0.442676	0.004282	0.004246
Alkmaar	667,946	316800	1475	1159	0.474290	0.002208	0.001735
Dordrecht	307,291	118000	1259	1248	0.384001	0.004097	0.004061
Botlek Rotterdam	1,332,595	523600	2321	2303	0.392918	0.001742	0.001728
Duiven	380,336	164500	1928	1911	0.432512	0.005069	0.005025
Amsterdam	1,352,196	653400	13970	13970	0.483214	0.010331	0.010331
Moerdijk	913,409	318800	1340	1328	0.349022	0.001467	0.001454
Roosendaal	347,382	137400	715.7	709.4	0.395530	0.002060	0.002042
	7,555,592	**3,048,470**	**35,198**	**34,717**			
				Average	**0.3958407**	**0.0041994**	**0.0041333**
				Recomme	1.071	0.003	0.003

Table A3. Incinerator emissions and emission factors 2016.

Location	Waste (t)	2016 NO2 (kg)	PM10 (kg)	PM2.5 (kg)	emission factors e_{fno2} (kg/t)	e_{fpm10} (kg/t)	$e_{fpm2.5}$ (kg/t)
Harlingen	256,373	95740	758	751.4	0.37344	0.00296	0.00293
Delfzijl	376,953	122600	1539	1525	0.32524	0.00408	0.00405
Wijster	711,987	177500	3328	3299	0.24930	0.00467	0.00463
Hengelo	637,061	283800	3360	3336	0.44548	0.00527	0.00524
Weurt	276,002	135400	2200	2181	0.49058	0.00797	0.00790
Alkmaar	670,521	319500	1267	995.5	0.47650	0.00189	0.00148
Dordrecht	310,525	114000	1260	1248	0.36712	0.00406	0.00402
Botlek Rotterdam	1,372,204	523700	2758	2738	0.38165	0.00201	0.00200
Duiven	386,973	143800	1740	1725	0.37160	0.00450	0.00446
Amsterdam	1,483,122	691600	14240	14240	0.46631	0.00960	0.00960
Moerdijk	958,479	353300	1272	1261	0.36860	0.00133	0.00132
Roosendaal	345,096	141000	703.8	697.6	0.40858	0.00204	0.00202
	7,785,296	**3101940**	**34425.8**	**33997.5**			
				Average	**0.3937007**	**0.0041984**	**0.0041370**
				Recommen	1.071	0.003	0.003

Table A4. Relative risks and 95% confidence intervals for all considered health effects.

Pollutant	Health effect	Exposure	Relative risk (per 10 µg/m^3)	95% confidence interval	Group
PM2.5	Respiratory hospital admissions	Short	1.0190	0.9982 - 1.0402	A*
PM2.5	CVD hospital admissions	Short	1.0091	1.0017 - 1.0166	A*
PM2.5	All cause mortality	Long	1.0620	1.040 - 1.083	A*
PM2.5	Restricted Activity Days (RADs)	Short	1.0470	1.042 - 1.053	B*
PM2.5	Work days lost	Short	1.0460	1.039 - 1.053	B*
PM2.5	Mortality	Short	1.0123	1.0045 - 1.0201	A
PM10	Incidence of asthma symptoms in asthmatic children	Short	1.0280	1.006 - 1.051	B*
PM10	Incidence of chronic bronchitis	Long	1.1170	1.04 - 1.189	B*
PM10	Prevalance of bronchitis	Long	1.0800	0.98 - 1.19	B*
PM10	Post -neonatal infant mortality	Long	1.0400	1.02 - 1.07	B*
NO2	Respiratory hospital admissions.	Short	1.0180	1.0115 - 1.0245	A*
NO2	All cause mortality	Long	1.0550	1.031 - 1.08	B*
NO2	Bronchitic symptoms in asthmatic children	Long	1.0210	0.99 - 1.06	B*

Table A5. Population and incidence data for all considered health effects.

Pollutant	Health effect	Unit	Pop at risk	Pop at risk (N)	Incidence rate
PM2.5	Respiratory hospital admissions	cases	all ages	17181000	0.00767
PM2.5	CVD hospital admissions	cases	all ages	17181000	0.01364
PM2.5	All cause mortality	life years lost	30+	11418000	0.008327
PM2.5	Restricted Activity Days (RADs)	days	all ages	17181000	19
PM2.5	Work days lost	days	age 18-64	10340884	10
PM2.5	Mortality	life years lost	all ages	17181000	0.008327
PM10	Incidence of asthma symptoms in asthmatic children	days	age 5-19	2728000	0.17
PM10	Incidence of chronic bronchitis	cases	27+	11852000	0.0039
PM10	Prevalance of bronchitis	cases	age 6-12	1124000	0.186
PM10	Post -neonatal infant mortality	deaths	1-12 months	155833	0.00372
NO2	Respiratory hospital admissions.	cases	all ages	17181000	0.00767
NO2	All cause mortality	life years lost	30+	11418000	0.008327
NO2	Bronchitic symptoms in asthmatic children	cases	age 5-14	1691000	0.0472

Table A6. Overview of sources for unit values of health effects.

Health effect	Lower Holland (2014)	Upper Holland (2014)	Brandt et al. (2013)	DEFRA (2019)
Respiratory hospital admissions	2637	2637	9404	9750
CVD hospital admissions	2637	2637	9404	9750
All cause mortality	69171	159440	91535	49655
Restricted Activity Days (RADs)	110	110	155	
Work days lost	156	156		
Mortality	69171	159440	91535	25663
Incidence of asthma symptoms in asthmatic children	50	50		
Incidence of chronic bronchitis	64256	64256	62,797	
Prevalance of bronchitis	705	705		
Post -neonatal infant mortality	1918080	3956040	3756098	
Respiratory hospital admissions.	2637	2637	9404	
All cause mortality	69171	159440	91535	49655
Bronchitic symptoms in asthmatic children	705	705		

10.2 - Appendix B

Detailed overview of steps taken in OPS-pro 2018 and QGIS 2.18.

OPS-Pro 2018 steps:

1. Use the file "Maak_bronbestand-v1.8" to insert data on incinerator locations and emissions. An example entry for a single location is shown in figure 1. Both for NO_x (as NO_2) and PM_{10} emissions are entered. Based on PM_{10} emissions, concentration maps for both PM_{10} and $PM_{2.5}$ are generated.
2. Define a run and load the emissions file created in step 1.
3. A resolution of 500m by 500m is chosen. (four cells equate to one square kilometre).
4. Meteorological and surface roughness data is included in the model, long term annual average over the period 1995-2004 is chosen as meteorological input.
5. Output is a map of the Netherland showing concentrations of the selected pollutant and a report providing values for all grid cells.
6. Use the "Importeer_OPS_output_v1.0" to generate tables with 'rijksdriehoek' coordinates and concentration values.
7. Save as a CSV file, which can be used in GIS software.

"Maak_bronbestand-v1.8" and "Importeer_OPS_output_v1.0" are available from:
https://www.rivm.nl/operationele-prioritaire-stoffen-model/tools

QGIS steps:

1. Load the layer "buurt_2017" from the file "Wijk- en buurtkaart 2017", made available by Statistics Netherlands.
2. Create a new field which holds the population density per km^2 per 'buurt'. Note that the layer "Buurt_2017" measures area in ha, not square kilometre.
3. Rasterize the population density value in the 'buurt' polygons. Multiplying the population density value by the area of the grid cell gives the population per cell.
4. Add delimited text layer, the CSV file. The result is a large amount of grid points that hold the concentration value for the three pollutants. Locations of the incinerators can also be added through a CSV file.
5. To convert the points value to a raster of value, triangulated irregular network (TIN) interpolation is used such that the points are in the middle of the created grid cells, this requires an offset of 250m in the layer extent. This is done for all three pollutants. This results in a grid of cells containing concentration values.

Two approaches can be taken. Calculate the number of additional cases per grid cell and sum across all cells for the total number of additional cases in the Netherlands, or determine the population weighted average mean concentration and use this value to determine additional cases for the Netherlands as a whole directly.

6. For per cell calculation of additional cases, use raster calculator to make the following calculation:
   = (POP_DENSITY / 4) * Incidence rate * (RR-1) * (CONC_POLL / 10)
   Use zonal statistics to sum across all grid cells for the total.
   For population weighted annual average approach use raster calculator the calculate the population weighted average concentration values as follow:

= ((POP_DENSITY / 4) / 17070735) * (CONC_POLL)
Use zonal statistics to sum across all grid cells for the weighted average concentration. With the result, the additional cases for all health impacts can be calculated.

The second approach is preferred as the calculation has to be executed within GIS only three times (once for all pollutants), whereas the first approach will need to be done for all pollutants and health impacts individually in GIS. However, both approaches can be compared as a way to verify one another.

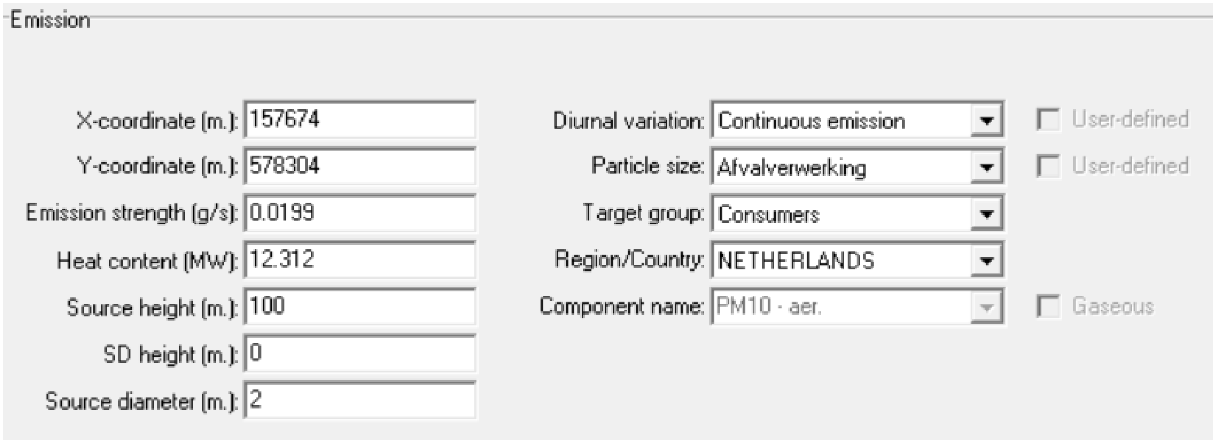

Figure 1B. Example of a single entry.