Modelling of Concrete Waste Using Life Cycle Assessment & Damage Cost

نمذجة نفايات الخرسانة عن طريق تقييم دورة حياتها البيئية وتكلفة الضرر

by

MOHAMED DARWISH MOHAMED SAEED

A thesis submitted in fulfilment

of the requirements for the degree of

DOCTOR OF PHILOSOPHY IN PROJECT MANAGEMENT

at

The British University in Dubai

July 2019

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ABSTRACT

There is a massive quantity of concrete waste in the city of Dubai and in United Arab Emirates (UAE) landfills, due to the enormous growth of infrastructure in the last two decades that has resulted in an increase in the number of construction and demolition sites, and currently there is no effective management option to reduce the concrete waste in the landfills by sustainable methods. Furthermore, concrete waste accumulated in landfills can damage human health, resources and ecosystems. Therefore, in this study, a Life Cycle Assessment (LCA) was conducted between the current method of landfilling and its transportation of concrete waste to landfill, and the proposed recycling method. While the comparison helps in highlighting the ecological impact of concrete waste, the best waste management method, and the associated damage cost through the comparison of the relationship between ecological impacts and their costs, it also helps in reducing the environmental impacts and achieving project sustainability. Furthermore, the LCA was conducted through the utilisation of the ISO 14040:2006 framework and principles, which guided the study to follow the phases of LCA in an organised and systematic approach. In addition, the EcoInvent 3.4 database and SimaPro 8.5.2.0 software were used as tools to correlate and simplify LCA phases and results. Furthermore, the Handbook Environmental Prices 2017 was used to find the damage cost for each impact. Life Cycle Impact Assessment (LCIA) and damage cost results showed that concrete waste in the landfill has a considerable impact on the environment compared to the recycling method and transportation process. The study opens up opportunities for future research to identify which impact indicator and management method of concrete waste has a significant influence on the environment and its damage cost. Moreover, Life Cycle Cost Assessment (LCCA) can be conducted in future to compliment this study in finding the financial benefit of different management options of concrete waste beyond its damage cost. This study's contribution is predominantly the expansion of an understanding of modelling the damage cost, in addition to

providing LCA results for the management of concrete waste, which was based on different waste management options such as landfilling, recycling and transportation process.

الملخص

توجد كميات هائلة من المخلفات الخرسانية في مكبات دولة الامارات العربية المتحدة وتحديداً في إمارة دبي. وذلك بسبب تطور وتقدم مشاريع البنية التحتية والعمران في آخر عقدين من الزمن، مما أدى إلى ارتفاع عدد مواقع البناء والهدم. حالياً، لا توجد هناك أي طرق إدارية فعالة لتقليل المخلفات الخرسانية في المكبات من خلال استخدام الطرق المستدامة. بالإضافة إلى ذلك، المخلفات الخرسانية تزداد في مكبات مخلفات البناء والهدم مما تسبب العديد من الاضرار على صحة الانسان والمصادر المتاحة والنظام البيئي. في هذه الدراسة تم تطبيق تقييم دورة الحياة البيئية بين الطريقة المستخدمة الحالية في عملية التخلص من النفايات الخرسانية وهي طريقة الطمر وعملية توصيل النفايات الخرسانية عن طريق الشاحنات المخصصة إلى المكب مقارنةً بطريقة إعادة تدوير المخلفات الخرسانية. في حين أن المقارنة تساعد في بيان الاضرار البيئية الصادرة من مخلفات الخرسانة واختيار أفضل الممارسات وبيان التكلفة البيئية المرتبطة بالأضرار البيئية مما تساعد في الحفاظ على البيئة وتحقيق استدامة مشاريع البنية التحتية والعمران. تم استخدام عملية تقييم دورة الحياة البيئية بناءً على المنظمة الدولية للمقاييس 14040:2006 أنظمة وقواعد، مما توجه الدراسة في استخدام مراحل دورة الحياة البيئية بطريقة منظمة ومرتبة. كذلك تم استخدام قاعدة بيانات ايكو انفنت 3.4 وبرنامج تحليل دورة الحياة البيئية سيمابرو 8.5.2.0كأدوات لربط وتبسيط مراحل ونتائج دورة الحياة البيئية. أيضاً، تم استخدام كتاب الأسعار البيئية 2017 لإيجاد تكلفة الضرر لكل تأثير. إن نتائج تقييم دورة الحياة البيئية وتكلفة الضرر توضح بأن طمر المخلفات الخرسانية في المكب لها تأثير هائل على البيئة مقارنةً بعملية توصيل المخلفات الخرسانية عن طريق الشاحنات المخصصة إلى المكب وطريقة إعادة تدوير المخلفات الخرسانية. الدراسة تفتح آفاق وفرص مستقبلية لتحديد أي مؤشر وأي طريقة إدارية مثلى لمخلفات الخرسانة التي لها تأثير كبير على البيئة وبيان تكلفة الضرر. علاوة على ذلك، يمكن إجراء تقييم تكلفة دورة الحياة البيئية في المستقبل لتكملة هذه الدراسة في إيجاد الفائدة المالية لمختلف طرق الإدارة الخاصة بالنفايات الخرسانية بما يتجاوز تكلفة الضرر. تتمثل مساهمة هذه الدراسة في توضيح فهم نمذجة تكلفة الضرر من المخلفات الخرسانية، بالإضافة إلى توفير نتائج تقييم دورة الحياة البيئية لإدارة النفايات الخرسانية والتي تستند إلى خيارات مختلفة مثل الطمر وإعادة التدوير وعملية النقل.

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CHAPTER 1: INTRODUCTION

1.1. Introduction

The study consists of eight chapters. Chapter one discusses the theoretical research background related to concrete waste in terms of environmental aspects. In addition, the research gap and statement of problem clarify the lack of studies in the area of LCA of concrete waste in recent years. Moreover, the research aim and objectives, research questions, and significance and novelty of the study are included.

Chapter two introduces the literature review of waste in the construction industry such as the definition and concept of waste, waste management policy, construction and demolition (C&D) waste, the current method of C&D waste disposal, the composition of C&D waste and the impact of C&D waste on the environment.

Chapter three includes a literature review providing an overview of concrete usage, beginning with concrete as a construction material, waste concrete in the United Arab Emirates construction industry, the environmental impact of concrete waste, methods of assessing environmental impacts and monetising environmental indicators.

Chapter four describe the literature review of methods for modelling the ecological impact of concrete waste by using LCA steps, for example, life cycle goal and scope definition, life cycle inventory, life cycle impact assessment and interpretation of the results.

Chapter five elaborates the research framework, modelling methodology of concrete waste by LCA steps and LCIA, which is based on ReCipe2016 calculations that consist of 18 midpoint impacts and three endpoint impacts.

Chapter six introduces the LCIA by ReCipe2016. The environmental impact results for concrete waste used data from EcoInvent and SimaPro software. In addition, damage cost results are calculated by referring to the Handbook Environmental Prices 2017. The results consist of 18 midpoint impacts and three endpoint impacts of landfilling method, recycling method and transportation process. They include the mathematical relationship of LCA results and damage cost for 15 impacts and for three waste management methods, which are landfilling, recycling and transportation.

Chapter seven illustrates the research findings and discussion by identifying the objectives of the study.

Chapter eight concludes the research findings and discussion by identifying the knowledge contribution and research implication, research limitations and recommendations.

1.2. Research Theoretical Background

Effective waste management is important, as clarified by Won and Cheng (2017), who state that construction and demolition waste management have been researched in many different ways to find proper waste management options. Stages of waste management start with reduction as the first step in the 3Rs principle (reduce, reuse, recycle) to minimise the generated waste; although waste can be recycled or reused if production cannot be avoided or it can be disposed of in landfill if the 3Rs steps did not work properly (Won & Cheng, 2017). The disposal stage requires time estimation and counts the number of trucks needed to deliver the generated waste to the landfill, so the planning and management of waste are important (Won & Cheng, 2017).

The advantages of recycling waste are the economic benefit and achievement of project sustainability (Ghosh, 2016). On the other hand, construction waste has an impact on the environment due to transportation and the demolition process (Ghosh, 2016). The cost of construction is rising due to the increase in the cost of construction materials, which can be avoided by using materials such as steel and bricks which can be recycled and generate revenue for the contractors (Ghosh, 2016). Ghosh (2016) states that materials can generate revenue from the recycling method, such as steel, concrete and masonry blocks, which recommended taking actions before loaded into landfills.

In addition, Kulatunga (2006) suggests that applying best practice waste management can result in project quality and sustainability. Poon (2003) agrees that the massive increase in waste landfills is a burden on the environment. It was illustrated by Bossink and Brouwers (1996) in Yuan (2013) that the 3Rs can reduce the purchasing cost of materials and save transportation cost and disposal cost, and generate revenue from selling waste. Furthermore, Lu and Yuan (2010) believe that, when a country is not considering sustainable practices for construction waste management, this is reflected in the loss of budget and time, and the impact on the environment. Similarly, Kulatunga (2006) states that clear waste management practices in general in construction fields result in achieving sustainability, and better economy and quality. Lingard (2000) believes that proper waste management can decrease waste disposal and reduce the landfill areas. According to Lu (2016), construction and demolition waste management in China had reached 1.13 billion tonnes in 2014.

Ulubeyli, Kazaz and Arslan (2017) have stated that the recycling of construction and demolition waste is the most preferred environmentally friendly treatment method. Moreover, the importance of the waste management method in a number of countries is clear; for example, Europe has set a target to reuse, recycle and recover non-hazardous construction and demolition waste at 70% of total weight by 2020 (Tojo & Fischer, 2011). So far, this method has been successful in five countries, which have reached the target of recycling 70% of hazardous construction and demolition waste (Bohne, Brattebø & Bergsdal, 2008). Bohne, Brattebø & Bergsdal (2008)reported that the percentage of recycling in the Netherlands, Denmark, Estonia and Germany had achieved 98.1%, 94.9%, 91.9%, 86.3%, and 79.5% respectively. In the USA it has been demonstrated that the recycling method has reached 73.5% of construction and demolition waste, 35% of recycled mixed construction and demolition waste, 85% recycled of bulk aggregate, and 99% recycle of asphalt pavement (Kofoworola & Gheewala, 2009). In addition, Ortiz, Pasqualino and Castells (2010) stated that 50 to 95% of generated construction and demolition waste can be recycled depending on its nature.

Waste management methods have significant importance for many countries depending on the nature of the waste's utilisation (Sales & de Souza, 2009). For example, recycled aggregate is widely used in pavement construction in Brazil (Sales & de Souza, 2009). Similarly, recycled aggregate is used in public projects in Hong Kong (Lu & Tam, 2013). Furthermore, some countries have set certification schemes to ensure the quality of recycled materials in the construction sector (Weil, Jeske & Schebek, 2006). For instance, in Germany, the permission to use recycled materials in new construction works must be checked for quality by certified plants (Weil, Jeske & Schebek, 2006). The statistics show the average number of recycling plants in Germany, Austria, the Netherlands, the UK, Italy, Belgium, France, Sweden, Ireland and Spain to be 1000, 150, 120, 100, 92, 50, 30, 10, eight and six respectively (Symonds Group, 1999). From an economic point of view, the average rate of recycling waste in Europe is 47%, while, according to the Waste Framework Directive as reported by Torgal (2013), the percentage of recycling non-hazardous waste in 2020 should reach 70%. Moreover, there is a possibility of taking advantage of surplus construction and demolition waste to recycle concrete fractions into new aggregate and cement (Ismail & Ramli, 2013).

1.3. Research Gap

There are several points to make regarding the research gap relating to the ecological impacts and damage cost of concrete waste shows several points. For example, while Won and Cheng (2017) state that construction and demolition waste management have undergone many types of research to find the proper waste management option, there is no reliable evidence that can relate to it. Moreover, a LCA study of concrete waste was conducted by Mah (2017), where only a LCA was applied, with no consideration of the damage cost or the other essential impacts in terms of midpoint and endpoint LCA results. Bravo, Brito and Evangelista (2017) clarify that concrete has several advantages and disadvantages such as environmental footprint, but there is a general lack of research in terms of the ecological impacts of concrete waste itself. What is not yet clear is the impact of concrete waste on resources, human health and ecosystems, which was elaborated on by Richardson (2013), who stated that the consumption of concrete has grown from 1500 million tonnes in 1995 to an estimated 3500 million tonnes by 2020, but no previous study has investigated the forecasting of the LCA results and damage cost results of concrete waste for real impacts. Moreover, many studies have validated the economic point of view regarding the recycling of concrete waste. For instance, Torgal (2013) stated that the average rate of recycling waste in Europe is 47%; but this percentage is only related to certain waste products and limited to the recycling method only, without taking into account the LCA and damage cost. Kolay and Akentuna (2014) reported that some studies suggest CO2 emissions and energy consumption are appropriate benchmarks for the ecological influence of waste concrete, but, till recently, there has been no reliable evidence that demonstrates the proper benchmarking for the ecological influence of waste concrete based on all the related impacts of emissions on human health, ecosystems and resources.

1.4. Statement of Problem

It was identified by Fischer and Davidsen (2010) and Flower and Sanjayan (2007) that the second most consumed material worldwide is concrete, after water. Pade and Guimaraes (2007) and Huntzinger and Eatmon (2009) believe that the cement in concrete during production is accountable for 8% of global CO2 emissions. Furthermore, i Vieira and Pereira (2015) identified that reusing and recycling C&D materials has many benefits, in addition to sparing landfills, and saving and preserving natural resources. It was elaborated in detail by

Elchalakani and Elgaali (2012) that most of the concrete waste in the construction industry results either from the demolition of old concrete structures or from the surplus concrete dumped from construction sites. In Gulf Cooperation Council (GCC) countries, as clarified by Dubai Municipality (2010), 120 million tonnes of waste were generated in 2010, which ranked the country in the 10 top waste producers globally. Moreover, around 75% of the waste that goes into landfill is C&D waste (Elchalakani & Elgaali, 2012). In addition, Elchalakani and Elgaali (2012) state that it has been estimated that the city of Dubai generates about 76,000 tonnes of waste daily, which is the highest per capita in the world.

1.5. Research Aim and Objectives

This research aims to investigate the environmental impact and damage cost of concrete waste in the United Arab Emirates. This aim is accomplished through the following objectives:

- Find the environmental impact (eco-impact) and footprint by LCA of landfilling method, recycling method, and transportation process
- Find the environmental damage cost of landfilling method, recycling method, and transportation process
- Find the mathematical relationship between LCA results and damage cost results.

1.6. Research Questions

To achieve the objectives, the following questions were created:

1. What are the causes of concrete waste in the UAE?
2. What is the current method of concrete waste management in the UAE?
3. What is the level of ecological damage that concrete waste has on the environment in the UAE?
4. What is the level of cost damage that concrete waste has on the environment in the UAE?

1.7. Significance and Novelty of the Study

This study provides an exciting opportunity to advance our knowledge on analysing the environmental impact and damage cost of concrete waste by landfilling method and recycling method in parallel with the transportation process. Therefore, this study makes a significant contribution to research on LCA by demonstrating the findings of ecological and damage costs that correlate with concrete waste in terms of landfilling and recycling method.

The study offers some important insights into the beneficiaries from this research, such as building and road construction projects, consultants, contractors and manufacturers of concrete products. Governments and private authorities that are related to waste management and environment control will benefit from the method of this study because it allows them to minimise the environmental impact and damage cost of concrete waste in construction projects, and find the best waste management option that achieves sustainability by referring to the results of LCA and damage cost in this research.

1.8. Thesis Outline

The thesis consists of eight chapters and the outlines of the chapters are presented as follows:

- CHAPTER 1 provides a holistic overview of the research, including the background and the justification of the study, the aim and objectives, and the research methodology.
- CHAPTER 2 describes a basic understanding of construction waste, especially concrete waste, generated from the construction industry. The impact of concrete waste on the environment is also addressed.
- CHAPTER 3 presents a background of the most common materials used in the construction industry such as concrete. The discussion will include a brief introduction about their physical properties and application in construction. Their usage in the UAE construction industry and the effects and consequences on the environment will also be examined.
- CHAPTER 4 provides an in-depth explanation of the Life Cycle Assessment methodology and monetisation of concrete waste.
- CHAPTER 5 provides an in-depth explanation of modelling the Life Cycle Assessment and damage cost.
- CHAPTER 6 presents the results of the LCIA midpoint and endpoint in parallel with the damage cost of concrete waste based on the selected waste disposal method.
- CHAPTER 7 presents the research findings and discussion.
- CHAPTER 8 presents the knowledge contribution and research implications, research limitations and recommendations.

CHAPTER 2: LITERATURE REVIEW (WASTE IN THE CONSTRUCTION INDUSTRY)

2.1. Introduction

When constructing buildings and roads, the construction industry produces and consumes many aggregates to produce concrete (Edge Environment Pty Ltd, 2011). Henry and Kato (2014) state that concrete is the most used construction material in the world. There are different types of waste in the construction industry – steel, concrete, soil and masonry materials (Edge Environment Pty Ltd, 2011). The quantity of construction waste is increasing due to increased urbanisation development (Henry & Kato, 2014). This chapter discusses concerns with regard to the concept of waste and its impact on the environment. The chapter starts with a definition of waste followed by a review of critical issues related to construction and demolition (C&D) waste. A discussion of some precise problems related to construction waste is presented. This consists of the current waste management policies in some countries, current methods of disposing of the waste, and the cost and impact of construction waste on the environment.

2.2. Definition and Concept of Waste

Mercante (2012) describes the types of general waste as consisting of inert materials and non-inert materials that come from public landfills and other resources. Each type consists of different materials. Poon (2001) identified and categorised the construction and demolition (C&D) waste in Hong Kong into inert and non-inert materials. According to Poon (2001), inert materials, for example, soil, sand, brick and concrete, are disposed of in public landfills. Non-inert materials are timber, bamboo, glass and paper. Moreover, eShant et al. (2014) state that the waste is considered one of the leading issues in the construction industry that has financial and climate impacts, although, as illustrated by Larisa (2016), the problem nowadays is the increasing use of natural resources, which leads to depletion of these resources. Moreover, the increased use of natural resources has a negative effect on the environment. It was elaborated by Wu et al. (2017) in Tam (2008) that the challenge of effective and efficient waste minimisation in the building construction field is an issue. Based on the above statements, it is obvious that waste generation is a global problem. Many countries around the world have targets, plans and strategies such as recycling to reduce waste generation and protect the environment from greenhouse gases that produce the waste (et al. Wu, 2017).

On the other hand, some practices could lead to reducing the waste. For example, Saez, (2013) pointed out that waste sorting is important to identify and segregate the types of waste for recycling. The waste sorting method considers the effective measures to minimise waste and increase material recovery. In his report, Magalhães (2017) clarified that most generated waste comes from the construction industry as it is known as the source of a negative impact on the environment which is associated with depletion of raw material. Extracting the natural raw materials to process with different chemical components to produce the final product has to use energy, which pollutes the environment (Magalhães, 2017). Peng, Scorpio and Kitbert (1997) introduced the concept of a hierarchy for handling construction waste at the most preferred stage to reduce the waste from the beginning, as illustrated in Figure 2.1. For example, order the precise amount of material required (reduce). The second stage is to reuse the waste if possible to avoid disposing of it. The third stage is to follow the recycling method, which is an important stage to help achieve project sustainability, reduce the debris going into landfill and protect the environment. The least preferred stage is to dispose of the waste in landfills or dump it without consideration of the advantages of the top preferred stages in the waste minimisation hierarchy (Peng, Scorpio & Kitbert, 1997).

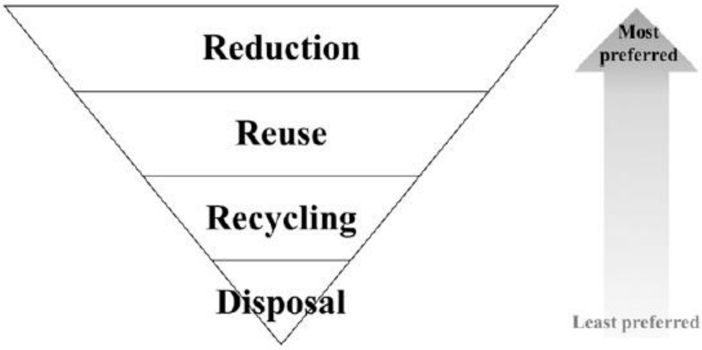


Figure 2.1: Hierarchy of handling construction waste (Peng, Scorpio & Kitbert 1997, p. 2)

Furthermore, reduction and reuse of waste are the preferred stages that help to avoid debris accumulating in landfill (Peng, Scorpio & Kitbert 1997, p. 2). Moreover, the recycling method is the green and sustainable stage which reduces the waste in landfills, provides more area and regains some energy from the recovery of waste, which reduces the impact of waste on the environment from greenhouse gas compounds such as carbon dioxide (CO2), methane (CH4), nitrous oxide (N2O) and fluorinated gases (Peng, Scorpio & Kitbert, 1997).

2.2.1. Current Local Practices

In the UAE, as it is one of the fastest-growing countries in terms of constructing building and road projects in the last two decades, there is a huge quantity of generated waste, which accumulates in landfills (UAE Government Waste Management, 2017). According to the UAE Government Waste Management Policy, due to rapid construction, population and economic growth in the UAE, cites such as Abu Dhabi, Dubai and Sharjah have set their target and strategy for waste management by converting the waste into energy and reducing the greenhouse gases and municipal waste at the landfills or dumpsites (UAE Government Waste Management 2017, p. 1). Dubai Municipality (2012) set the integrated master plan 2012 for Dubai waste management, which aimed to reduce the amount of waste sent landfill to zero in 20 years by relying on the recycling method approach. On the other hand, in 2008, the city of

Abu Dhabi established a centre for waste management called Tadweer, which is responsible for t waste management policies and strategies (Waste Statistics in the City of Abu Dhabi, 2011). In addition, in 2007, the city of Sharjah established a waste management company called Beeah as a public, private partnership (UAE Government Waste Management, 2017). The company utilises waste management methods such as recycling of different types of waste (UAE Government Waste Management 2017, p. 1-4). On the other hand, construction waste, which counts as the highest percentage of the waste in landfills that has a negative impact on the environment, such as concrete waste that has not been treated efficiently to reduce the millions of tonnes of it across landfills (UAE Government Waste Management 2017, p. 1-4).

Dubai Municipality (2015) classified the types of waste; for example, domestic waste consists of residential, commercial and institutional waste. Each category of domestic waste has a different type of generated debris from different resources, as illustrated in Table 2.1.

Table 2.1: Classification of wastes in the city of Dubai (Dubai Municipality 2015, p. 9)

Domestic Waste	***Residential*** • Households (single and multi-family units) • Apartments • Villas/Colonies	• Organics • Paper & cardboard • Metal • Plastic • Glass • Wood • Textile • Sand & stones • Others • Uncounted
	Commercial • Retail and wholesale • Shopping malls • Hotels • All other business establishments	
	Institutional • Government offices • Schools, universities • Mosques, other churches	

Furthermore, C&D waste in Dubai, as shown in Table 2.2, is a mixture from building and road activities and consists of different types of debris, such as concrete, glass, wood, gypsum board, asphalt paving, plastics, soil, etc. There are also different categories of waste such as tyre waste, horticultural waste, hazardous waste, medical waste and sewage waste, as also illustrated in the table (Dubai Municipality, 2015).

Table 2.2: Classification of wastes in the city of Dubai (Dubai Municipality 2015, p. 9)

Construction & Demolition Waste	Construction sites, road repair/renovation sites, broken pavement, demolition companies	Wood/timber, carpets, textiles, rubber, glass, plastics, metals, ceramics, soil, stones, boulders, concrete, bricks asphalt paving & roofing, gypsum board
Horticultural Waste	Landscaping & maintenance of public & private cultivated areas such as parks, plazas, gardens, beaches & recreational areas	Leaves, grasses, twigs/branches, flowers, tree trimmings
Hazardous Waste	Healthcare facilities such as: Hospitals, laboratories, clinics, industries, i.e., wastes related to manufacturing and industrial processes and from construction and demolition activities.	Sludge from on-site effluent treatment; chemicals containing acids, paints, varnishes, adhesive and sealants, solvents, lead batteries, photographic films and papers, oil wastes, flammable liquids; toxic, infectious and corrosive substances; packaging materials, animal tissue waste, fluorescent tubes and other mercury containing waste
Medical Waste	Hospitals, clinics, physician's offices, dental offices, blood banks, veterinary, research facilities and laboratories, pharmaceuticals	Infectious fluids, bloods & body parts, sharps, expired medicines
Sewage Sludge	Sewage Treatment Plants from	Sewage solids/sludge from

To conclude this part, based on the above statements and the concept of waste, construction waste is a serious issue worldwide. However, there are different types of waste in landfill. Increasing the debris has a serious effect on the environment. Furthermore, C&D waste represents the largest amount of waste in landfill. Furthermore, some countries have certain strategies to reduce waste generation and reduce its impact on the environment. Based on the construction waste hierarchy, it is recommended to focus on the most preferred stages – reduce, reuse and recycle debris – rather than the disposal option. The more waste that is sent to landfill, the greater the effect on the environment.

2.3. Waste Management Policy

The European Union Waste Framework Directive (WFD) legislation includes distinguish, prevention, preparing for reuse, recycling, recovery, and landfill on a superior scale (EC, 2008). On the other hand, Ewijk and Stegemann (2016) have compared USA and European waste policies, and state that waste hierarchy is more entrenched in Europe than in the USA. There are some comparisons and practices of waste in some countries. For example, Kourmpanis (2008) mentions that the size of waste differs in each country according to culture and the country's economic situation. Manowong (2012) sets out the process of dealing with waste: that recycling of concrete begins in the early 1970s in the USA and Europe. The rules and policies are critical to construction waste management. Although Jia et al. (2017) identified some statistical results in China, for example, one construction development report has shown that 1.5 billion tonnes of construction waste has been generated and 5% is the reuse materials.

In addition, Ewijk and Stegemann, (2016) agree that the waste policy remains as a performance criterion. Moreover, other authors have shared their opinions on waste management policy. For example, Ponnada and Kameswari, (2015) state that the rules regarding waste management in India failed in implementation due to the authorities. This indicates that some authorities cannot implement effective waste management (Ponnada & Kameswari, 2015). Lingard (1997) stated that waste management plans could reduce the consumption cost of materials. Since it is identified by Lingard (1997) that there is still no effective implementation of the waste management plan.

2.3.1. Current Local Practices

The waste management policy in the city of Abu Dhabi is controlled by the Environment Agency, which means this agency has the authority to identify the strategies, laws and regulations for waste management (Environment Agency in Abu Dhabi, 2013). To reduce the waste, the Environment Agency in Abu Dhabi (2017) has identified one of their targets as being to divert from 60% to 80% of C&D waste by 2020. Although Federal Law No. 24 of 1999 acts as the overall umbrella law for environmental protection, the Environment Agency in Abu Dhabi (2013) initiated Law No. 16 of 2005 concerning the re-organisation of identifies, while the Waste Management Law No. 21 of 2005 identifies the Environment Agency in Abu Dhabi as the competent authority for waste management. Furthermore, the practice of the treatment method of C&D waste conducted in Abu Dhabi by the establishment of two recycling plants (Environment Agency in Abu Dhabi, 2013). On the other hand, Dubai Municipality (2018) has initiated circulars and technical guidelines for waste management in relation to fees and fines, waste disposal service, tracking system for municipal solid waste, safe transportation of solid waste and C&D waste collection, which might help in reducing the waste (Dubai Municipality, 2018). On the other hand, the government sector responsible for the management of waste indicated that there are regulations for the control of construction waste (Dubai Municipality, 2018). For example, there is a fine for accumulated construction waste, which means all types of building waste should be removed and the construction area must be cleaned, although segregation of waste should be adopted in all construction project areas (Dubai Municipality, 2018). With regard to the quantity of waste allowed in landfill, the government sector has no limit for it, and there is no regulation to insist that all stakeholders reuse or recycle concrete waste (Dubai Municipality, 2018). Government sectors have provided landfills across UAE to dispose of the waste (Dubai Municipality, 2018).

2.4. Construction and Demolition (C&D) Waste

Bovea and Powell (2016) highlighted that there is a significant quantity of generated C&D waste worldwide. In addition, most of the C&D waste is sent to landfill without considering better waste management options. Rodrigues et al. (2013) stated that C&D waste could be reused as raw materials in manufacturing secondary products. Furthermore, Vieira and Pereira (2015) identified that reusing and recycling C&D materials are beneficial because they avoid the waste being disposed of in landfill, and thus save natural resources. It was clarified by Lu and Yuan (2011) and Yuan and Shen (2011) that management of C&D waste has become an exciting and important area for both practitioners and researchers globally, as the management of practical resources is a challenging worldwide issue. However, as assessed by Cheng and Ma (2013), the management of C&D waste in many countries is not adequate. Bovea and Powell (2016) stated that there are considerable opportunities to enhance C&D waste management in some areas, for example, technical, environmental and economic. These opportunities lead to reducing the waste and obtaining some advantages from waste management areas (Bovea & Powell, 2016). To elaborate on the situation of C&D waste management, Yu et al. (2013) showed that life cycle assessment is important to evaluate the environmental performance of construction waste management to have an efficient and effective decision-making process. Most of the time, as stated by Lu and Yuan (2012), C&D waste consists of both inert and non-inert materials, which means it is helpful to sort the waste before it is disposed of to the public landfills or landfills.

Kabir, AlShayeb and Khan (2016) stated that most of the construction waste is consider as inert material, which can be recycled. For example, construction debris in the Kingdom of Saudi Arabia contains solid waste generated from excavation and construction which consists of concrete pieces, marble, plastic, petrochemicals, papers, asphalt, and products of paint, aggregates and small quantities of steel that cannot be separated from concrete waste (Kabir, AlShayeb & Khan, 2016). However, in a landfill site located to the east of Jeddah, construction waste was found to contain significant portions of marble, gypsum, ceiling panels and ceramics which could be reused without a recycling method (Kabir, AlShayeb & Khan, 2016). Moreover, more than 50% of solid debris in the Gulf region comes from construction sites

(Kabir, AlShayeb & Khan, 2016). Furthermore, Saudi Arabia is the largest country in the Gulf area and it is one of the main countries that generate solid waste in the Gulf region (Kabir, AlShayeb & Khan, 2016). According to Eurostat (2015) statistics, in Europe the total quantity of waste generated from the C&D industry in 2012 exceeded 2.5 billion tonnes, while 34% resulted from C&D waste.

The use of C&D waste in different Segregation is possible, as shown in Figure 2.2 taken from Das and Swamy (2014), as reclaimed aggregate material can be used in the construction of asphalt pavement. However, it should be tested and evaluated first to ensure its properties.

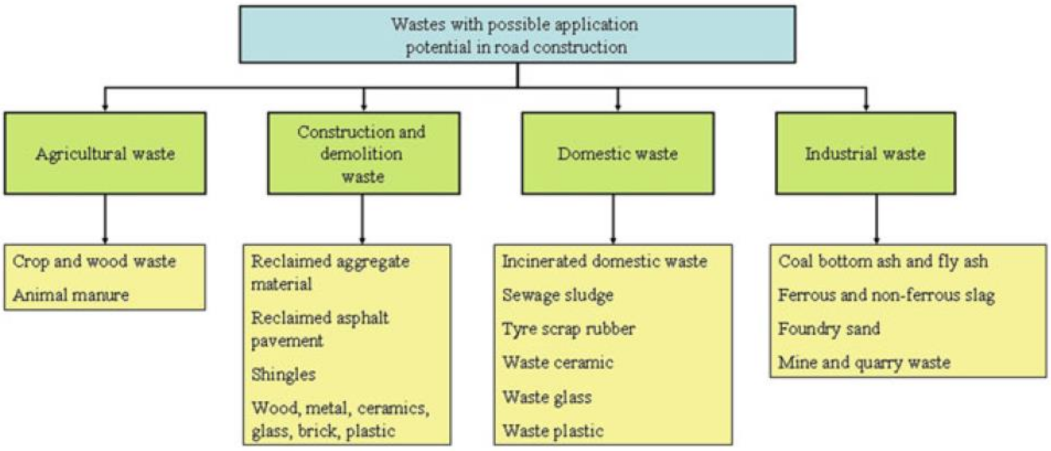


Figure 2.2: Segregation of C&D waste (Das & Swamy 2014, p. 421)

As stated by et al. Udawattaa (2015), the generation of construction waste is one of the main problems in the industry. Moreover, it affects the environment and the construction field. Shen et al. (2004) explained that construction waste consists of building waste, earth, concrete, steel, wood, composites of filed materials, rubble, and numerous amounts of construction debris such as from land excavation, civil activities, road development, demolition, building refurbishment. The following are waste statistics for a number of developed countries: in the USA the total construction waste quantity reached 136 million tonnes and in the United Kingdom it was 70 million tonnes yearly (Yuan, 2012), while in Australia it was 14 million tonnes (Site Waste Management Plans in Construction Industry 2013, p. 1). The above results show that the quantity of construction waste is huge and it requires an efficient waste management method to decrease the amount of debris and protect the environment (Site Waste Management Plans in Construction Industry 2013, p. 1).

As states by Abdelhamid (2014), the waste from C&D sites have been an issue in Egypt. Although, efficient waste management required support from the public. As states by Arslan, Coşgun and Salgin (2012), natural resources are typically consumed in large quantities in the construction field and this produces a high quantity of C&D waste. Furthermore, C&D waste creates the largest quantity of all solid waste (Coşgun & Salgin, 2012). In fact, diverse studies have shown that there is high quantity of C&D waste generated. Moreover, dumping the C&D waste without control has a negative effect on the environment, and a financial cost (Coşgun & Salgin, 2012).

On the other hand, Das and Swamy (2014) stated that there are some activities that produce waste, such as construction, renovation or demolition of numerous structures, for example, buildings, bridges and roads. Moreover, these types of waste consist of cracked pieces of cement concrete, timber, ceramics from tiles, glass from window panels, asphalt shingles from villa roofs, metals from the reinforcement of roofing and truss structures, bricks and plastics, although the sources of these materials may differ; for example, some of the waste can be recycled and reused, such as timber and metals (Das & Swamy, 2014). Furthermore, cracked pieces of cement concrete mainly contain cracked aggregates with hardened cement paste on the aggregate surface area. Ferrari, Miyamoto and Ferrari (2014) stated that aggregates can be recovered from waste concrete (Das & Swamy, 2014). As claimed by Das and Swamy (2014) that (Ferrari, Miyamoto & Ferrari (2014) could refer to reclaimed or recycled concrete aggregates (RCA).

Figure 2.3 shows the grouping of C&D waste as identified by Arslan, Coşgun and Salgin (2012). The figure consists of four types of C&D waste, which are excavation soil, roadwork wastes, demolition and complex wastes (Arslan, Coşgun & Salgin, 2012). On the other hand, it clarifies each type of C&D waste with the type of waste, resources and components (Arslan, Coşgun & Salgin, 2012). From an overall perspective, materials such as soil, sand and concrete are represented in most of the waste types as resources and components (Arslan, Coşgun & Salgin, 2012). For example, roadwork, demolition and complex wastes include concrete waste, which means concrete is the main material used in most construction activities. In addition, that means there is a high percentage of concrete waste in C&D stages (Arslan, Coşgun & Salgin, 2012).

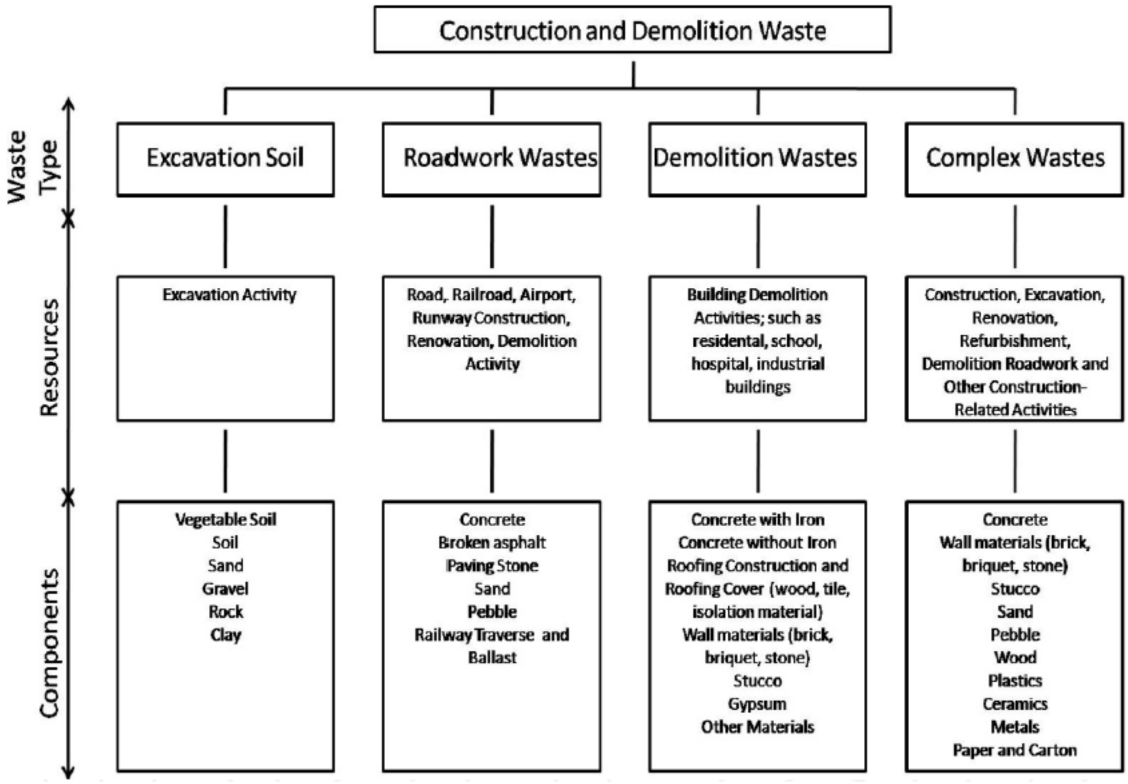


Figure 2.3: Grouping of C&D waste as identified by Arslan, Coşgun and Salgin (2012, p. 9)

The term C&D waste refers to uncontrolled materials produced from building and infrastructure activities of construction, renovation and demolition (HKEPD, 2013; USEPA, 2013). As stated by Wua, Yu and Shen (2017), the generation of C&D waste can be decreased through efficient management. However, there is a large probability that C&D waste will generate useful resources after undergoing a suitable treatment method (Wua, Yu & Shen, 2017). Moreover, millions of tonnes of construction and demolition waste can undergo a treatment method (Wua, Yu & Shen, 2017). An increase in the number of construction projects around the world has led to an increase in construction waste, which is mainly disposed of into landfill (Wua, Yu & Shen, 2017). Based on statistics clarified by Tam, Tam and Le (2010), all wastes, which count 42%, are associated with C&D waste in Australia. Moreover, 81% of this 42% is concrete waste. On the other hand, Liang, Ye and Xi (2015) state some statistics from the USA that the quantity of concrete waste is around 30 million tonnes per year, and this is 29% of the solid waste total (Tam, Tam & Le, 2010).

Table 2.3 shows the quantity of C&D waste in India, as illustrated by Ponnada and Kameswari (2015). The highest type of wastes generated are soil, sand and gravel at 4.20 to 5.14 million tonnes, while 4.40 million tonnes of brick and masonry waste and 2.40 to 3.67 million tonnes of concrete waste are generated.

Table 2.3: Quantity of C&D waste in India (Ponnada & Kameswari 2015, p. 4)

Constituent	**Quantity generated in Million Tonnes**
Soil, Sand and Gravel	4.20 to 5.14
Bricks and Masonry	3.60 to 4.40
Concrete	2.40 to 3.67
Metals	0.60 to 0.73
Wood	0.25 to 0.30
Others	0.10 to 0.15

The quantity of generated waste indicates that construction waste represents a huge quantity compared to the other types of waste (Ponnada & Kameswari, 2015). Although, it was stated that concrete and masonry are the highest types of waste generated in the construction industry at more than 50% (Ponnada & Kameswari, 2015).

Moreover, Table 2.4 shows that recycling of these wastes into aggregates could reduce the waste accumulating in landfills and reduce the use of virgin materials in construction work (Ponnada & Kameswari, 2015).

Table 2.4: Recyclable waste materials in demolition sites in India (Ponnada & Kameswari 2015, p. 6)

Architectural salvage	**Non-ferrous metals**	**Land-clearing residuals**
Doors and door frames	Wiring/conduit	Trees, stumps, brush
Windows and frames	Plumbing (pipes, fixtures)	Soil
Millwork	HVAC (ductwork, motors)	**Ferrous metals**
Furniture and Furnishings	**Asphalt**	Structural steel
Office furniture	**Aggregate**	Steel framing members
Partition systems	Concrete (with & without rebar)	**Porcelain fixtures**
Medical/lab equipment	Brick	**Ceiling tiles**
Reception/casual furniture	Concrete block	**Gypsum wallboard**
Lockers/athletic equipment	**Wood**	**Roofing**
Carpeting	Dimensional lumber	Shingles
Broadloom	Panels (plywood, OSB, MDF)	Commercial membrane
Carpet tiles	Engineered beams (glu-lam, etc.)	Wood, metal, slate

Tam, Tam and Le (2010) illustrate some important statistical results for China; for example, in Hong Kong, 38% which counts 14 million tonnes of C&D waste is produced annually but 11 million tonnes is reused for reconstruction, repair and earthwork activities, while the other 3 million tons ends up in landfill (Tam, Tam & Le, 2010). On the other hand, China is more concerned about waste minimisation, and the country produces only 750,000 thousand tonnes, which count 16% that related to C&D waste (Tam, Tam & Le, 2010). This gives a perfect indication of waste management in China, a country that is considering reducing the impact of C&D waste on the environment (Tam, Tam & Le, 2010). Globally, demolition waste represents 70% of the construction and demolition waste (Martínez, Nuñez & Sobaberas, 2013). Globally, C&D presents the highest amount of waste in the industry because of the enormous growth of infrastructure projects (Martínez, Nuñez & Sobaberas, 2013).

2.4.1. Current Local Practices

In the UAE, most of the C&D waste is sent to landfill without any thought about effective treatment methods. In the main, there are certain types of C&D waste in UAE landfill, such as reinforced concrete, un-reinforced concrete, precast concrete, ready-mix plant waste concrete and concrete blocks. The generation of C&D waste in the UAE is very high due to the effective growth of infrastructure development (Dubai Municipality, 2018). The reasons for the increase in C&D waste, as clarified by Wua, Yu and Shen (2017), are, for example, more production, weak handling, improper storage and ordering, changes to design drawings, etc.. Some global statistics show that, in the USA, 1 to 1.2 million dollars per project can be lost on debris (Woo et al. 2006). While, to compare with UAE, the most used material in the construction field is concrete. Furthermore, most of the buildings in the UAE are constructed from concrete (Dubai Municipality, 2018). In addition, concrete is one of the most represented wastes in landfills across this country (Dubai Municipality, 2018). There are a number of causes of construction and demolition waste, such as ordering too much material (Glass et al., 2008). For instance, in the city of Dubai, it is calculated that C&D waste in 2011 reached 6,638,471 tonnes, which made it the highest type of waste compared to general waste, horticultural waste and liquid waste, which were calculated at 2,689,808, 175,022 and 154,119 tonnes respectively (Envirocities eMagazine, Issue 4, January 2013).

Table 2.5, as described by Dubai Municipality (2013), shows the quantity of waste generation in the city of Dubai in 2011. The highest debris type is C&D waste at 6.6m tonnes compared to general waste, horticultural waste and liquid waste at 2.7m, 175,022 and 154,119 respectively. The table results indicate that C&D waste took up a large proportion of the space in landfills (Dubai Municipality 2013, p. 5).

Table 2.5: Quantity of waste generation in the city of Dubai in 2011 (Dubai Municipality 2013, p. 5)

Waste Type	Quantity (tones)
General Waste	2,689,808
C & D Waste	6,638,471
Horticultural Waste	175,022
Liquid Waste	154,119

Based on data from the Statistics Centre in Abu Dhabi, Table 2.6 shows the quantity of C&D waste in the city of Abu Dhabi generated between the years of 2011 to 2016 (Statistics Centre 2016, p. 5). The table identifies different types of waste such as industrial and commercial waste, agricultural waste and municipal waste. It shows that the quantity of C&D waste is the highest compared to other types of waste (Statistics Centre 2016, p. 5). The table illustrates that C&D waste was increasing until 2013 and then started to decrease slightly from 2014 to 2016. There are two types of C&D waste, normal and mixed C&D waste. The generated quantity of C&D waste in 2011 was 7.6m tonnes and this increased in 2012 to 9.6m tonnes, while in 2013 tit decreased slightly to 7692921 tonnes, and then continued decreasing until in 2015 it stood at 2.9m tonnes. Suddenly, in 2016 the quantity of C&D waste increased, to reach 4,532,379 tonnes (Statistics Centre 2016, p. 5).

Table 2.6: Quantity of generated waste in the city of Abu Dhabi between the years of 2011 to 2016 (Statistics Centre 2016, p. 5)

Source	2011	2012	2013	2014	2015	2016
Total	10,430,031	12,705,902	11,762,602	9,918,590	8,420,998	9,598,969
Construction and demolition waste	**7,624,575**	**9,628,309**	**7,692,921**	**4,419,665**	**2,876,313**	**4,532,379**
C & D waste	5,075,325	5,721,367	2,767,342	1,723,497	2,042,883	2,580,913
C & D mixed waste	2,549,250	3,906,942	4,925,579	2,696,168	833,430	1,951,465
Industrial and commercial waste	**643,338**	**804,174**	**1,305,556**	**3,312,125**	**3,306,644**	**2,692,768**
Agriculture waste	**816,069**	**898,258**	**999,239**	**561,991**	**493,106**	**745,644**
General agriculture waste	63,345	305,749	339,078	320,689	380,465	503,001
Mixed agriculture and Animal waste	752,724	592,509	660,161	241,302	112,641	242,643
Municipal waste	**1,105,602**	**1,272,668**	**1,528,093**	**1,466,590**	**1,678,983**	**1,561,680**
Households, streets, and public gardens waste	859,329	1,059,219	1,234,336	1,298,955	1,420,323	1,397,292
Bulky waste	246,273	213,449	293,757	167,635	258,660	164,387
Other*	**240,447**	**102,493**	**236,793**	**158,219**	**65,952**	**66,499**

Source: The Centre of Waste Management - Abu Dhabi, ADNOC.
*Include Oil and Gas sector

To conclude this part, in general the percentage of C&D waste is increasing yearly due to an increase in construction activity. The type of material that is usually present in high quantities in landfill is C&D waste. However, some countries are following good practice in dealing with C&D waste, for instance, by establishing recycling plants to reduce the waste accumulated in landfill and to reuse the recycled materials for different purposes. Some statistics indicate that the large quantity of C&D waste is an issue in many countries because of the space required to provide areas for landfill sites and the negative effect of waste on the environment. On the other hand, in the UAE some cities contribute to waste management such as recycling C&D waste and other types of waste. However, the effective implementation of waste management is not sufficient. Moreover, the use of recycled C&D waste is still not efficient in the UAE and some other countries, such as India.

2.5. Current Method of C&D Waste Disposal

Kabir (2013) claimed that the problem in the GCC is that most of the countries have a high percentage of waste disposal. Moreover, developing countries have two practices regarding construction and demolition waste, which are fly tipping and landfilling (Kabir, 2013). Townsend, Wilson and Beck (2015) identified the percentage results for building and demolition waste, which they claimed were around 25-45% in the discharge. In contrast, in Kuwait, 90% of the construction and demolition waste is disposed of in landfill (Kartam, 2004). This is also the case in Hong Kong, where most of the construction and demolition waste is disposed of in landfills (Tam & Tam, 2006). It has been shown in a study that 21-30% is the cost of construction project waste materials (Ameh & Daniel, 2013). Arulrajah (2017) confirms that C&D waste materials are excavated materials, asphalt debris, bricks, concrete, plasterboard timber, asbestos and mixed contaminated soil. Moreover Abrelpe et al. (2015) assert that in Brazil almost 45 million tonnes of construction waste was generated in 2015, which was 57% of the total amount of solid waste.

Coelho and de Brito (2013) divide C&D waste into two different types; for instance, the first category is inert materials such as sand, bricks and concrete while the second type is non-inert materials such as glass, paper, plastic, wood and organic materials. Moreover, this declaration excludes excavation activity, which is not considered as C&D waste. Dajadian and Koch (2014) noted that construction waste includes concrete, steel, rebar, electricity cable and equipment, aluminium, copper, brick, plywood, metal, paint and paper. Furthermore, it has been asserted that 40% of natural resources are consumed yearly by the construction industry worldwide, although the reason for this use of virgin resources is because of the enormous amount of development, which causes an increase in the amount of waste going to landfill. In addition, 50% of waste excluded from primary treatment is sent to landfill directly. It was stated by Panos and Danai (2012) that many studies have shown that the design stage is important to level. In addition, 33% of on-site waste occurs because the designers failed in the reduction of waste during the design process.

The term the 3Rs stands for reduce, reuse and recycle (Peprah, Amoah & Achana, 2015). Reduce the generated waste, reuse the material and recycle the waste (Huanga et al. 2012;Viswanathan & Norbu, 2008). The concept of 3Rs could be used as a policy or assessment tool (Peprah, Amoah & Achana, 2015). To reduce the debris, there should be some regular and general practices in society (Tojo, 2010). According to Tojo (2010), there are unsorted waste landfill bans and recycling materials bans in the Netherlands, Germany, Belgium and Switzerland. According to Tam (2009), it is mandatory to use certain materials in the demolition process in Japan. The recovery of C&D waste has been considered by many experts in order to find different management models, which focus on improving the recycling method (Shen, 2010). Malia (2013) cites evidence from Yuan & Shen (2011), in which they clarify that the importance of establishing construction waste and demolition indicators has increased in recent years.

Figure 2.4 below illustrates the process of construction and demolition waste management in Shenzhen city in China (Yuan, 2017). The process of building materials and building demolition in construction and demolition stages is considered as C&D waste which is going to be collected and sorted if it is only inert and non-inert waste. These two types of procedures are disposed of to either landfill or an appropriate dumping area, while mixed C&D waste, considered as residential or sellable waste, is disposed of in landfill or dumping area. Moreover, there is a process for reusing or recycling building waste materials before dumping it or disposing of it in landfill (Yuan 2017, p. 4).

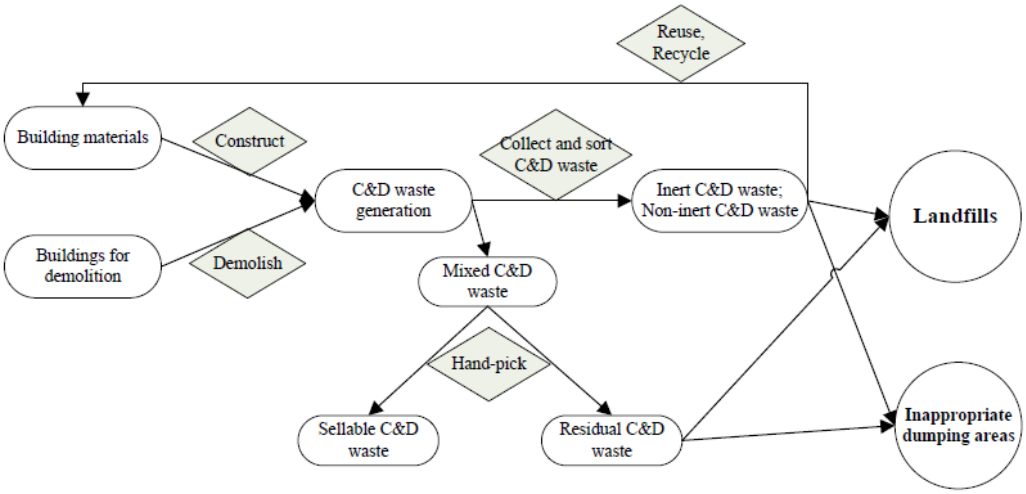


Figure 2.4: The process of construction and demolition waste management in Shenzhen city in China (Yuan 2017, p.4)

2.5.1. Current Local Practices

The current disposal method for C&D waste in GCC countries is the landfilling method. Moreover, the current treatment method for waste is recycling general waste and some construction and demolition waste, which obviously shows that there are some waste management procedures in the region (Dubai Municipality, 2018).

To conclude this part, C&D waste has different compositions, such as inert materials and non-inert materials. The quantity of C&D waste in landfill is high, which requires the establishment of proper waste management methods to decrease and avoid construction waste. Moreover, the impact of accumulated waste in the landfill has a negative impact on the environment.

2.6. Impact of C&D Waste on the Environment

Polat (2017) elaborates that C&D waste generated by the construction industry has a negative effect on the environment, exploits natural resources, and increases the pollutants in the atmosphere. Moreover, as Babak (2017) clarifies, the impact of construction activity on the environment occurs in all stages. The disposal of waste to dumps leads to accumulation of the debris and the availability of fewer waste spaces for future disposal. On the other hand, de Magalhães, Danilevicz and Saurin (2017) believe that the construction industry is producing waste that affects the environment and its impacts rise when the development process increases.

Global statistics produced by Wiedmann (2014) show that buildings influence and affect 30% of the global carbon footprint and this will increase in the future. There is a common policy by the European Commission, as clarified in EU Directive 2002/91, released in 2002 for sustainable buildings and to reduce the influence of materials on the environment to promote energy efficiency and reduce greenhouse gas (GHG). Some methodological analysis tools, as described by Ahlroth et al. (2011), such as LCA and carbon footprint have been initiated to reduce the environmental impact and during the manufacture of building materials, recycle the materials and minimise the emissions from total transportation. Based on evaluation of the environmental impact of construction materials, Giama and Papadopoulos (2015) have selected materials such as bricks, cement, steel, concrete and cement plaster. The reason for selection is because they are the most used building materials in Europe, and they are the most embodied energy in the buildings at 50%, as clarified by Muneer & Kelley (2007) and Chen et al. (2001).

Management of the construction and demolition waste is recommended to decrease the amount of waste and save the environment (Babak, 2017). Moreover, the negative impact on the environment comes from the extraction of natural raw materials, which leads to a violation of the landscape and increases air pollution (Babak, 2017). The example of the main natural material used in construction activities is aggregate, which requires rocks to be crushed to obtain the material, which causes depletion of natural resources (Babak, 2017). For more clarification, Estanqueiro (2010) states that, due to the evolution of civil construction, production of the C&D waste has risen and consequently, its management is tougher. However, he added that the construction techniques and materials used are immediately related to the nature of this waste and one of all its major sources is the demolition of buildings at the end of their life. C&D waste contributes appreciably to worsening the environmental issues, especially regarding its transport and deposition (Estanqueiro, 2010). Moreover, the increasing amount of waste is one of the primary problems that developed societies are dealing with nowadays (Estanqueiro, 2010). Furthermore, one of the solutions discovered for a maximum amount of the waste is giving it a second lifetime (Estanqueiro, 2010). In addition, if the waste is properly treated, it may constitute added benefit to the economy and environment (Estanqueiro, 2010). Lastly, the recycling of C&D waste helps to decrease the quantity of debris in landfill, and reduces the extraction of virgin aggregates and its influence on the environment (Estanqueiro, 2010).

Furthermore, Fuertes (2013) believes that core construction industries are not considering utilising an Environmental Management System to solve environmental matters. There are some types of materials that can still be used instead of being sent to landfill. For example, Kralj (2008) states that C&D waste materials that can be reused and recycled to reduce the environmental impact are wood, steel, bricks and concrete.

Generation of construction industry waste cannot be zero because some individual stages of waste generation cannot be avoided (Galan, 2013). Delivery of building materials to the site represents 10-20% waste of the total construction building weight (Poon, Yu, & Ng, 2001). In 2000, the overall rate of global carbon dioxide emissions of cement was 8.6% (Kleijn, 2012).

2.6.1. Current Local Practices

In the city of Dubai, the generation of C&D waste is increasing yearly because of the huge development of infrastructure projects, which requires the production of building materials such as concrete to be increased, which affects the environment; for example, it increases the percentage of carbon dioxide in the environment (Dubai Municipality, 2018). Moreover, extracting and crushing aggregates from the mountains as the main raw materials for concrete production results in depletion of natural resources (Dubai Municipality, 2018).

2.7. Summary

This chapter has provided the importance and overview of concrete waste in the construction industry. Moreover, it includes different waste policies used in different countries around the world. In addition, the chapter has discussed the composition of C&D waste and types of waste disposal methods that might be used to reduce the concrete waste. Finally, the impact of concrete waste on the environment has a huge effect in terms of greenhouse gases and other harmful substances.

CHAPTER 3: LITERATURE REVIEW (OVERVIEW ON CONCRETE USAGE)

3.1. Introduction

This chapter provides an overview of concrete usage in the construction industry. Moreover, the chapter start with use of concrete as a construction material and the waste involved in its use. In addition, this chapter includes a literature review about the ecological damage that concrete waste might cause. Moreover, many Life Cycle Impact Assessment methods are presented and explained in this chapter. On the other hand, an explanation of the monetisation of the environmental impact is also included, which shows that different types of methods can be used to find the environmental damage in addition to LCA.

3.2. Concrete as a Construction Material

It was stated by Elchalakani, Basarir and Karrech (2016) that, in the current century, one of the priority challenges is to meet the growth of infrastructure and building activities, which is required for the rapid urbanisation and industrialisation of cities for the future. Concrete is one of the main and most important materials used in most civil engineering construction industries (Elchalakani, Basarir & Karrech, 2016). It is used for the structure of buildings in the construction of foundations, walls, roofs, slabs, columns and beams, and it is used in some countries in road construction as rigid pavements and for airport runways (Elchalakani, Basarir & Karrech, 2016). Moreover, concrete is widely used in many areas such as drainage projects, production of paving blocks and solid blocks as it is light and heavy weight (Elchalakani, Basarir & Karrech, 2016). The compounds of concrete are sand, fine and coarse aggregates, cement and waster (Elchalakani, Basarir & Karrech, 2016). These raw materials are mixed together with a specific ratio to generate concrete from the batching plant as the final product (Elchalakani, Basarir & Karrech, 2016). Fischer and Davidsen (2010) and Flower and Sanjayan, (2007) have identified concrete as the second most consumed material worldwide, although Henry and Kato (2014) state that concrete is the most used construction material in the world. Pade and Guimaraes (2007) and Huntzinger and Eatmon (2009) believe that the cement in concrete during production is accountable for 8% of global CO2 emissions. Moreover, cementitious products also produce CO2 emissions. Essentially, it was stated by Eisa (2014) that there is a strong link between concrete and global warming, which is the main issue with recent infrastructure activities. Furthermore, the quantity of concrete is increasing due to the huge development of the construction industry, as illustrated by Richardson (2013), in that the consumption of concrete has grown from 1500 million tonnes in 1995 to an estimated 3500 million tonnes by 2020.

The high quantity indicates that concrete usage has an effect on the environment. On the other hand, it was stated by Sadati et al. (2016), Saravanakumar and Dhinakaran (2013) and Maholtra and Mehta (2008) that manufacturing of Ordinary Portland Cement (OPC) requires high energy levels, which releases greenhouse gas emissions. Although the production of 1 tonnes of OPC needs 1.5 tonnes of raw material, it was clarified by Sadati et al. (2016) and Maholtra and Mehta (2008) that the global production of OPC in one year is 3.7 billion tonnes, which releases around 3 billion tonnes of CO2, which is almost 7% of the total emissions in Earth's atmosphere. Moreover, it was stated by Collins and Sanjayan (2002) that OPC production produces 5% of global greenhouse gas emissions.

It has been estimated that 896 million tonnes of C&D waste are produced in Europe (Fischer & Davidsen, 2010). In addition, the composition of these residues differs between countries (Fischer & Davidsen, 2010). Furthermore, the concrete percentage could range from 20% to 80%, as clarified by Klee (2009). The range shows that there is a huge quantity of concrete waste in landfill. In addition, Tam (2008) states that, although half the quantity of the concrete waste is disposed of in landfill, 100% of concrete waste can be recycled based on recent technology. However, most of the recycled concrete aggregates are used as backfilling material or in road activities. In addition, types of concrete are used for several purposes (Elchalakani & Elgaali, 2012). Moreover, in contrast to standard bricks, hollow blocks of concrete have been discovered to have higher constant quality, more strength, are very fast to construct and require low labour (Elchalakani & Elgaali, 2012). In addition, these hollow blocks of concrete are used in many construction fields (Elchalakani & Elgaali, 2012).

According to Akpinar and Khashmanb (2017), concrete is a compound material that consists of cement and other types of binders such as slag and fly ash, used together by adding an adequate ratio of water, fine aggregates (i.e. sand), coarse aggregates (i.e. gravel) and chemical admixtures that are used to improve certain concrete properties. There is a huge generation of the construction industry, as stated by Elchalakani and Elgaali (2012), that aggregates and freshwater have consumed a million tonnes in the construction industry. The results can indicate that construction development is increasing, which requires high consumption of building materials such as aggregates, which are used to produce concrete products (Elchalakani & Elgaali, 2012). Most of the waste comes from the construction field, as clarified by Fisher and Werge (2010), the EPA (2009) and Dubai Municipality (2010): significant amounts of materials are sent to landfill. In addition, it was elaborated in detail by Elchalakani and Elgaali (2012) that most of the concrete waste in the construction industry results either from the demolition of old concrete structures or from the extra concrete waste that represents in the sites or concrete fleet. Furthermore, many types of research focus on how to use the recycled aggregates and water, which helps to develop sustainable concrete with a long-term performance and which is environmentally friendly (Elchalakani & Elgaali, 2012).

On the other hand, Fib (2009) classified concrete as low strength and high strength; the normal strength of concrete is between the range 20 and 50 MPa and any other higher or lower between this range classified as low or high strength. Kim (2016) provides some characteristics of building structure properties, for example, mix design and strength of concrete types vary according to the building classifications and region, which is because concrete is produced by ready-mix plants located in different areas and cities. It was stated by Tam and Tam (2010) and Han and Thakur (2012) that the annual production of aggregates in the USA had reached 2 billion tonnes and it is expected to reach 2.5 billion tonnes by 2020. It was estimated that the quantity of raw materials consumed annually worldwide is around 3 billion tonnes (Saghafi & Teshnizi, 2011).

Concrete is a combination of materials; it consists of 80% of aggregates such as gravel and sand, while the other 20% is cement, as identified by Menard et al. (2013). It was stated by Yanik (2016) that the world demand for aggregates material for construction was expected to reach above 51 billion metric tonnes by 2019. Davis et al. (2016) mentioned that many aggregates require mining, processing and transportation, which consume high amounts of energy and affects the environment. Furthermore, Kumar (2017) stated that the greater demand for aggregates has effects on the environment as it depletes natural resources such as river rock and extracted sand, which is a risk worldwide (Kumar, 2017). Moreover, the demand for aggregates is getting in risk, which depends on economic growth and region (Kumar, 2017), although the sustainability of concrete can be achieved by reducing the consumption of natural aggregates by utilising a recycling method for C&D waste. It was stated by Medda (2016) that concrete is the second most used material in the world after water.

3.3. Waste Concrete in the UAE Construction Industry

The number of concrete block factories in the city of Dubai is 15 manufacturers and 57 concrete ready-mix plants (DCL 2018 p. 3-5). In the UAE, concrete is the main material utilised in the construction of buildings, bridges, road barriers and drainage. Furthermore, all buildings in the UAE, whether residential or commercial, are constructed from concrete and mostly this is the same for other countries worldwide (DCL, 2018).

Concrete is the most used material in the UAE as precast concrete or ready-mix concrete in construction activities (Elchalakani & Elgaali, 2012). Furthermore, concrete is used in road barriers and kerbstones, blocks, paving blocks and drainage construction projects (Elchalakani & Elgaali, 2012). In Gulf Cooperation Council (GCC) countries, as clarified by Dubai Municipality (2010), 120 million tonnes of waste were generated in 2010, which means that the region ranked in the 10 top waste producers globally. Moreover, around 75% is C&D

waste that goes to landfill (Elchalakani & Elgaali, 2012). In addition, Elchalakani and Elgaali (2012) state that it was estimated that the city of Dubai generated 76,000 tonnes of waste daily, which ranked at the top per capita in the world.

Table 3.1 shows a comparison between GCC countries, the USA and the 28 countries of the EU in terms of C&D waste. The results indicate that GCC countries produced more waste than the USA and EU-28 at 36 to 40%. Moreover, the quantity of waste generated per tonnes per capita in the year for GCC countries was 2.25 compared to the USA and EU-28 at 0.42 and 1.170 respectively, although there is a massive difference in the population numbers between the countries listed in the table. In GCC countries, the population in 2010 was 120 million, while in the USA it was 380 million and in the EU-28 it was 2.742 billion. The results in the table illustrate that GCC countries generate more waste compared to the other, mostly developed, countries (Fisher & Werge 2009, p. 2; EPA, 2008); (Dubai Municipality, 2010).

Table 3.1: The quantity of C&D waste in three regions (Fisher & Werge 2009, p. 2; EPA, 2008; Dubai Municipality, 2010)

Region	EU-28	USA	GCC
Population in 2010 (million)	501	310	40
Total Waste (million ton/year)	2742	380	120
C&D Waste (million ton/year)	850	130	90
C&D Waste (ton/capita/year)	1.70	0.42	2.25
C&D Waste / Total Waste (%)	31%	35%	75%

There are many types of concrete products which are disposed of as waste in landfill (Dubai Municipality, 2018). Moreover, the worth of hollow concrete blocks could be between 8 and 10% of the total cost of construction work, assuming that 20% of the construction work would use hollow blocks (Dubai Municipality, 2013). Furthermore, statistical results show that there are large amounts of wasted materials yearly (Dubai Municipality, 2013). Moreover, Gulf Cooperation Council (GCC) countries have been placed in the top 10 global waste producers at 120 million tonnes of debris per year, and the statistics were expected to reach 350 million tonnes by the end 2014 (Dubai Municipality, 2010). Although, 75% of the 120 million tonnes is C&D waste that consists of different types of debris (Dubai Municipality, 2010). Moreover, the city of Dubai generates more than 76,000 tonnes daily with highest per capita in the world (Dubai Municipality, 2010).

Furthermore, most of the waste is sent to landfill, which leads to an accumulation of waste in landfill and a negative effect on the environment (Dubai Municipality, 2018). The types of concrete waste in UAE landfills are reinforced concrete, normal concrete, hollow blocks of concrete, and waste concrete from construction sites (Dubai Municipality, 2018). Furthermore, these types of concrete are accumulating in the landfills daily. In addition, the waste management options such as recycling concrete waste to reduce the quantity of waste in the landfills are still not 100% effective. However, there is an initiative to use the recycled concrete waste in construction activities (Dubai Municipality, 2018). The collection and segregation of concrete waste are performed at the site of construction projects by collecting the debris in waste skips (Dubai Municipality, 2018). With regard to the disposal method for concrete waste, it is sent to the Al Bayada dump area (landfill) in Dubai, with a fee of 10 AED per truck entry (Dubai Municipality, 2018). Furthermore, management of construction waste is not sufficient at some levels because the huge percentage of waste and the absence of an effective treatment method in UAE is an issue (Dubai Municipality, 2018).

3.4. Environmental Impact of Concrete Waste

Concrete is one of the most common construction materials that emit a bulky volume of hazardous emissions into the environment during their production, construction, maintenance and demolition (Kim, 2016). Furthermore, Kim (2016) states that concrete emits large amounts of CO_2 during the production of raw materials such as aggregate and cement. Also, he added that transportation distance and energy in each stage of concrete production are used to calculate the CO_2 emissions. On the other hand, Roh (2013) concludes that CO_2 emissions from concrete account for 70% of materials such as reinforcing rod and steel section. Moreover, he agreed that generation of concrete can be divided into stages such as raw material, transportation and manufacturing, although. he mentioned that emission building could be divided into construction, operation and maintenance. Essentially, it was identified by Bravo, Brito and Evangelista (2017) that concrete has several advantages and disadvantages, such as its environmental footprint is a disadvantage due to its impact on the environment. This problem needs to be solved to have a sustainable construction industry (Bravo, Brito & Evangelista, 2017).

3.5. Methods of Assessing Environmental Impacts

There are many methods of assessing the environmental impact, for example, the Eco-indicator 95 methodology designed to LCA weighting method for product design development (Ministry of Housing, Spatial Planning and Environment, 2000). Eco-indicators are numbers that express the product or process of the total environmental load, which help to analyse it over the life cycle (Ministry of Housing, Spatial Planning and Environment, 2000). For example, Yahya and Boussabaine (2010) used Eco-indicator 95 to calculate brick waste and its influence on the environment.

On the other hand, Goedkoop et al. (2000) have stated some obstacles to weighting the results of LCA; for example, most Life Cycle Impact Assessment (LCIA) approaches cannot solve the weighting issues as they are established in a bottom-up approach, which means that inventory outcomes are considered as the starting point (Goedkoop et al. 2000). Essentially, the indicators are distinct without reflecting on weighting issues (Goedkoop et al. 2000). Moreover, to find out the Eco-indicator 99 scores, there are three required phases, as illustrated in Figure 3.1 (Goedkoop et al. 2000). The first stage is to identify all emission types that involve all the processes of extraction and land-use by adopting an LCA procedure (Goedkoop et al. 2000). The second stage is to compute the kinds of damage these flows cause to Human Health, Ecosystem Quality and Resources (Goedkoop et al. 2000). The third and last stage is to weight the damage categories (Goedkoop et al. 2000). The three steps will help to find the Eco-indicator scores required to complete the assessment (Goedkoop et al. 2000).

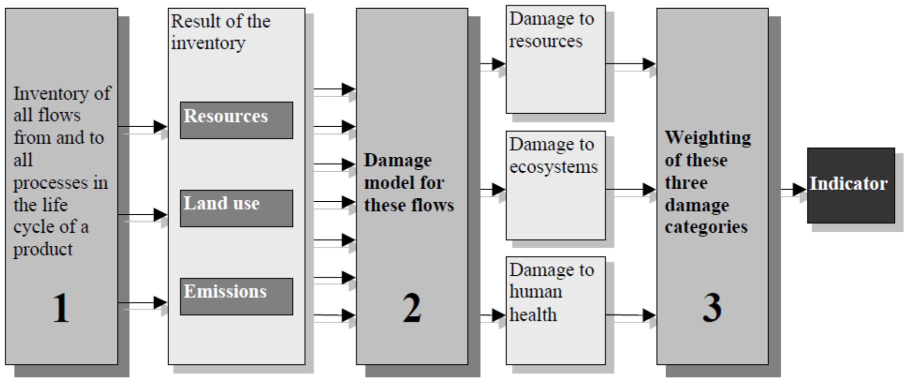


Figure 3.1: Phases of Eco-indicator 99 scores (Goedkoop et al., 2000, p. 4)

The weighting step is the most significant controversial step in the Eco-indicator methodology (Goedkoop et al. 2000). Essentially, emission and resource extraction in LCA has more than 10 different impact categories, such as ecotoxicity, acidification, ozone layer reduction and resource extraction (Goedkoop et al. 2000). In addition, the Eco-indicator 99 methodology process, as stated by Goedkoop et al. (2000), starts with the design of the weighting process and type of data that the panel can switch in a weighting process. The authors have concluded that different kinds of damage are caused by impact categories. On the other hand, Sharaai, Mahmood and Sulaiman (2010) have stated that impact assessment is used to identify substantial possible environmental influence by using life cycle impact assessment (LCIA) results. Furthermore, LCIA is distinguished from environment impact assessment (EIA) and risk assessment techniques; the reason is that the approach depends on the functional element (Sharaai, Mahmood & Sulaiman, 2010). In addition, LCIA includes four features: classification, characterisation, normalisation and weighting (Sharaai, Mahmood & Sulaiman, 2010). However, the optional elements are normalisation and weighting (ISO14040:2006). LCIA includes a midpoint approach and an endpoint approach (ISO14040:2006). In more detail, midpoint approaches include CML 2001, EDIP 97 and TRACI (Sharaai, Mahmood & Sulaiman, 2010), while endpoint methodology means a damage-oriented approach, as identified by Dreyer, Niemann and Hauschild (2003). In addition, Heijungs et al. (2003) clarified that the endpoint method is the elements in the impact pathway that made an independent assessment of the culture. On the other hand, Rydberg (2004) stated the examples of endpoint methodology, which are Eco-indicator 95 and 99 that are used in environmental assessment, and there are also methods such as EPS 92, 96 and 2000 and LIME 2003.

For more clarification about LCA methodology, Sharaai, Mahmood and Sulaiman (2010) explained that LCA assessment has four fundamental phases based on ISO14040. These are goal scope definition as is mentioned in ISO14040, life cycle inventory as stated in ISO14041, life cycle impact assessment as mentioned in ISO14042 and, finally, life cycle assessment and interpretation as included in ISO14043. By considering the four main phases, this gives a comprehensive evaluation, and it is necessary to use some Eco-indicators to assess the damage that might have an effect the environment. Concrete was assessed in the study of Sharaai, Mahmood and Sulaiman (2010) by Eco-indicator 99.

Although, Peña et al. (2003) state that the impact categories classification of LCI is by an impact pathway approach, which starts from LCI results to the endpoint. For more clarification and details, Figure 3.2 below, as described by Goedkoop (2000), shows the Eco-indicator 95 weighting principle based on the European scale which consists of 11 impacts, nine effects and three damages. For instance, products or ideas are analysed to find the main causes of environmental pollution and discover the improvements and opportunities (Goedkoop, 2000). The damage caused by materials and products can be in terms of impairment to human health, fatalities and the impairment to the ecosystem (Goedkoop, 2000).

The effects that cause the damage are the ozone layer, heavy metals, carcinogens, summer smog, winter smog, pesticides, greenhouse effect, acidification and eutrophication (Goedkoop, 2000). The use of Eco-indicators is applicable in many areas, where they help to find the impact on the environment and to find better weighting of a material or product to choose better options based on LCA results and Eco-indicator (Goedkoop, 2000).

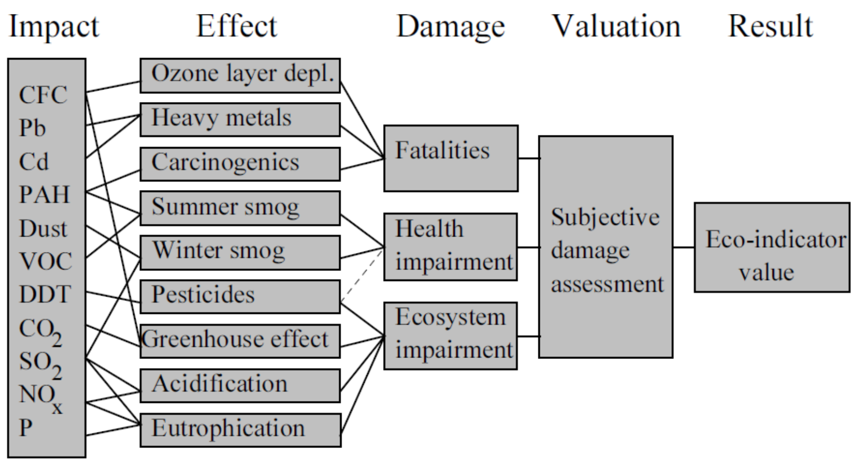


Figure 3.2: Eco-indicator 95 impact assessment method (Goedkoop, 1995, p. 1)

On the other hand, Guardigli, Monari and Bragadin (2011) have stated that Eco-indicator 99 has three categories of damage: Human Health, Ecosystem Quality and Resources, although each damage category is subdivided into impact categories (Guardigli, Monari & Bragadin, 2011). In addition, Guardigli, Monari and Bragadin (2011) used Eco-indicator 99 for reinforced concrete and wood structures to assess the environmental impact. Furthermore, Eco-indicator 99 is valid for the European perspective, but the adaption is based on personal experience (Guardigli, Monari & Bragadin, 2011). Moreover, the indicators help to combine the results of LCA in more coherent and practical parameters (Guardigli, Monari & Bragadin, 2011). Some modifications can be made to Eco-indicators, as Neri (2007) has done to ensure the assessment is perfectly effective in the evaluation. For example, he has added waters in the minerals impact category and has excluded superficial waters, to take into consideration that high consumption of water that unlimited elements always involves high energy for its extraction (Neri, 2007). In addition, materials such as gravel and sand have been added to the minerals category, as essential elements for the manufacture of building materials and energy (Neri, 2007). Furthermore, the elements of nitrogen and phosphorus, COD (Chemical Oxygen Demand) and BOD (Biochemical Oxygen Demand) were added to the Acidification and Eutrophication impact category, because they produce eutrophication of water (Neri, 2007). In addition, iron emissions were added to the Carcinogens category (Neri, 2007). With regard to energy demand, cumulative energy was included in Eco-indicator 99 to take into consideration the global energy combination process (Neri, 2007).

As described by Toxopeus, Lutters and Houten (2006), it could be abnormal that familiar indicators, such as CML-92, Eco-indicator 95 or Eco-indictor 99, are all based on weighting factors, which indicates the use of subjective substances. Moreover, designers appear to agree on these particular elements with the explanation that only comparative assessments between different conceptions are obligatory (Toxopeus, Lutters & Houten, 2006). Accordingly, the complete assessment of the influence is no longer an issue (Toxopeus, Lutters & Houten, 2006). The reason is the selection of a certain impact assessment method, or an exact use of weighting features is not constrained by the design process, while any specified indicator of the predictable environmental impact might be valuable in several circumstances (Toxopeus, Lutters & Houten, 2006). Koroneos and Dompros (2009) carried out another use of Eco-indicator 95 for the environmental assessment of a cement product and concrete life cycle. They clarified that the Eco-Indicator 95 weighting method was conducted for the objective of this study as stated in Goedkoop (2000). Moreover, the authors stated that the Eco-indicator is one method of aggregation or, as described in ISO 14042, that weighting over categories leads to an individual result. In addition, the Eco-indicator method is the Weighing factor (Wf) conducted on environmental impact indices, for example, greenhouse effect, ozone depletion, etc., which relate to the main damage triggered by the influence of the ecological

(Koroneos & Dompros, 2009); although the primary damages that might occur are 5% damage to the ecosystem, additional deaths per million people in a year, and health issues because of smog incidents (Koroneos & Dompros, 2009).

On the other hand, in their environmental inventory analysis Pushkar, Becke and Katz (2005) used the SimaPro database tool, LCA and Eco-indicator 99 for designing optimal buildings. The SimaPro database is recognised as a full database instrument (Becke & Katz, 2005). Moreover, it has a complete database of materials and processes in different fields (Becke & Katz, 2005). Furthermore, all processes are editable and can be improved to fit different circumstances or to form new areas. The SimaPro database contains a bulky amount of evaluation measures such as CML 92, CML 2 baseline 2000, Eco-indicator 95, Eco-indicator 99, Eco points 97, EDIP/UMIP 96 and EPS 2000 (Becke & Katz, 2005). It also includes sensitivity analysis for waste scenarios, recycling allocation methods, standardisation and weighing features (Becke & Katz, 2005). Moreover, Nicoletti, Notarnicola and Tassielli (2002) used Eco-indicator 95 and 99 for evaluating a LCA of ceramic and marble flooring tiles. The outcomes of their study show that arsenic emissions have an impact on the environment (Nicoletti, Notarnicola & Tassielli, 2002).

Figure 3.3, taken from Goedkoop et al. (1998), shows inventory table, part 1 and 2, and types of damage. The inventory table consists of resource use, land use and emissions. Each type of inventory table moves to part 1, which is the main part of modelling of the cause-and-effect chain. For more clarification, part 1 has three types of damage, which are damage to resources, damage to ecosystem health and damage to human health. These damages are related to the types of products and materials that have been selected for modelling in part 1. Finally, the last stage is part 2, which is used to assess the results of the product or material and its damage results based on the indicator (Goedkoop et al. 1998).

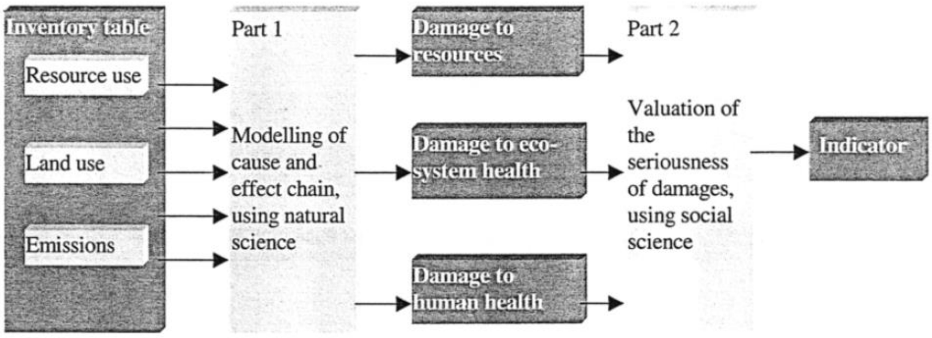


Figure 3.3: Inventory and types of damage (Goedkoop et al., 1998, p. 353)

Some authors, such as Simoes, Xara and Bernardo (2010), have stated that many studies have shown that they are using Eco-indicator 99 or CML 2 (2000), a method to assess the environmental impact of products and materials. On the other hand, Evangelista and Brito (2006) used Eco-indicator 99 for concrete made with recycled aggregates as a final analysis of the environmental score. The importance of the Eco-indicator 99 is that it can evaluate the environmental damage in many areas such as public health and ecosystems (Evangelista & Brito, 2006). Based on Evangelista and Brito's (2006) analysis, the recycled aggregate is better than natural aggregates in concrete products from the environmental damage scores.

SimaPro LCA software and EcoInvent LCA database were used in the environmental evaluation of concrete, plaster and brick component manufacturing (Giama & Papadopoulos, 2015). Moreover, for the environmental impact assessment, two indicators were used, Eco-indicator 95 and CML 2 baseline 2000 method (Giama & Papadopoulos, 2015). In addition, the selected functional unit of materials for evaluation of the environment was in kg for emissions and building materials and MJ/kg for embodied building energy (Giama & Papadopoulos, 2015). It was clarified in detail by Peuportier et al. (2009) that SimaPro is a comprehensive tool to develop LCA studies of a varied range products, activities or services. However, although it is not designed precisely to conduct LCA of buildings, its comprehensive databases and the flexibility of the impact assessment approaches involved make it appropriate for this objective (Peuportier et al., 2009). Furthermore, SimaPro provides a native user boundary by following ISO14040, and it can be used to create comprehensive modelling with scenario analysis (Peuportier et al., 2009). Moreover, the SimaPro tool helps to find direct impact assessment results from every stage of the studied system and it can analyse a composite of waste treatment and recycling methods (Peuportier et al. 2009). Although, SimaPro bring collaborative outcomes analysis, which help to trace, the results back to their origins in the real period (Peuportier et al. 2009). Furthermore, it presents a weak point analysis by using the 'tree' procedure to find any 'hot spots' (Peuportier et al. 2009). There are a number of methods of assessing the environmental impacts, as stated by Peuportier et al. (2009), as shown in Table 3.2.

Table 3.2: Methods of assessing the environmental impacts (ENSLIC_BUILDING-State of the Art Report, p. 6)

Method	Characteristics
Impact 2002+	Damage approach; it is similar to Eco-indicator 99, but totally recalculates toxicity influences
TRACI 2002	Established by US EPA and is based on the midpoint method
CML 2 baseline 2000	Updated revision of the 1992 method; more developed models and presence of fate analysis
EPS 2000	Damage approach, by conducting monetarisation (willingness to pay) in place of weighting by a panel
Eco-indicator 99	Damage approach, uses classification indicators at endpoint stage. Three types are involved by using different moulds
Ecopoints 97 (UBP)	Distance to objective, which is based on Swiss policy targets, which is referred to as the Ecoscarcity method or UBP
EDIP/UMIP 97	Characterisation and normalisation method established for Danish EPA; it also has a 2003 version
Eco-indicator 95	Includes damage approach, and distance to objective method based on scientific targets
CML 92	Very commonly conducted for midpoint method, moderately simple characterisation; it does not include fate or exposure, and has several normalisation sets

For more details of environmental assessment methods, IMPACT 2002+ is a combination of IMPACT 2002 (Pennington et al., 2004), Eco-indicator 99 (Goedkoop & Spriensma, 2000), CML (Guinée et al., 2002) and IPCC. In addition, IMPACT 2002 features mostly replace Human Health cancer and non-cancer factors and Aquatic and Terrestrial ecotoxicity factors (Peuportier et al. 2009), while Eco-indicator 99 factors primarily replace Respiratory effects, Ionising radiations, Terrestrial acid/nutri, Land use and Mineral extraction. Furthermore, CML factors replace Aquatic acidification and Aquatic eutrophication (Peuportier et al. 2009), although Aquatic eutrophication CF applied in this method is the one for a P-limited watershed (Peuportier et al. 2009). On the other hand, the impact categories such as Aquatic acidification and Aquatic eutrophication are midpoint indicators, which are not included in the endpoint by Peuportier et al. (2009).

The second is the TRACI 2002 method, which is a temporary application of the TRACI 2.0 method, which depends on preliminary data (Peuportier et al. 2009).

Thirdly, CML 2 baseline 2000 is an update of the CML 92 method and contains many radical models (Peuportier et al. 2009). The impact categories are abiotic resources, global warming, ozone layer depletion, toxicity as for humans will be aquatic and terrestrial ecosystems, photochemical oxidation, acidification and eutrophication (Peuportier et al., 2009). There are several sets of normalisation for the Netherlands 1997, Western Europe 1995 and Worldwide 1990 or 1995 (Peuportier et al. 2009).

Fourthly, the EPS 2000 method, which focuses on the most important environmental strategies in product design and assessment from an economic aspect and refurbishment of environmental damage caused by products (Peuportier et al. 2009). It is a tool that is mainly appropriate for the development process of a product in a corporation (Peuportier et al. 2009), although several impact types are considered in the group of four damage categories: human health, renaissance capacity of ecosystems, resource assets and biodiversity by Peuportier et al. (2009).

Fifthly, Eco-points 97 is an update of a 1990 method illustrated by the Swiss Ministry of the Environment, in cooperation with established environmental rules of that country (Peuportier et al. 2009). Mainly, it includes a practical number of impact categories which potential emphasise the levels of NOx, SOx, CO2, Pb, CD, Zn and Hg in the air, and levels of Cr, Zn, Cu, CD, Hg, Pb and Ni in water, pesticides, wastes, etc. Three types of existing method vary in normalisation factors by (Peuportier et al. 2009).

Sixthly, the EDIP/UMIP 97 method was introduced by the Environmental Design Centre of Industrial Products (EDIP) of Holland. It is similar to CML 92 but several features have been updated and developed (Peuportier et al. 2009). The c impact categories are global warming, aquatic and terrestrial ecosystems, ozone layer depletion, human toxicity, acidification, eutrophication, photochemical smog, dangerous pollution, wastes, radioactive waste and resources (Peuportier et al. 2009). On the other hand, there are two impact evaluation methods based on Eco-indicators 95 and 99, which are the most commonly used. Eco-indicators are numbers that represent the total environmental impact of a product or service, whose construal is relatively as high indicator had more impact and associated with the environmental (Peuportier et al. 2009).

Seventhly, Eco-indicator 95 consists of 10 impact categories, which are greenhouse effect, ozone layer depletion, soil acidification, eutrophication, heavy metals, carcinogenic substances, pollution, pesticides, energy resources and solid wastes (Peuportier et al. 2009). The normalisation influences are based on European data from 1990 (Peuportier et al. 2009). In addition, two sets of normalisation exist, which are Europe g and Europe e which use different hypotheses when inferring information by (Peuportier et al. 2009).

Eighthly, the Eco-indicator 99 method is an update of the Eco-indicator 95 method, which has three versions of Eco-indicator 99. The assumptions in the environmental models measure a Qualitative Perspective (E), in which the selected time span is the extreme long term; ingredients are included if there is the least indication as to their influence (Peuportier et al. 2009), while damages cannot be excluded and could cause catastrophic effects. In addition, for fossil fuels, it is calculated that they cannot simply be replaced (Peuportier et al. 2009). The Individualistic Perspective (I) is the time span for the short term, for example, 100 years or less. Constituents are included if the evidence is clear as to their effect (Peuportier et al. 2009).

Moreover, damages can be improved by technological and economic enhancement (Peuportier et al. 2009), although it is assumed that fossil fuels cannot be shattered, and they are missing from the evaluation (Peuportier et al. 2009). Furthermore, the Hierarchist Perspective (H) has a long-term time perspective (Peuportier et al. 2009). Constituents are counted if there is an agreement as to their effect (Peuportier et al. 2009). Usually, the

Hierarchic Perspective (H) is selected randomly, because it is the weighting typical of the group of experts who designed the process (Peuportier et al. 2009). Figure 3.4 shows the assessment method of Eco-Indicator 99 (Goedkoop et al. 2000, p. 5).

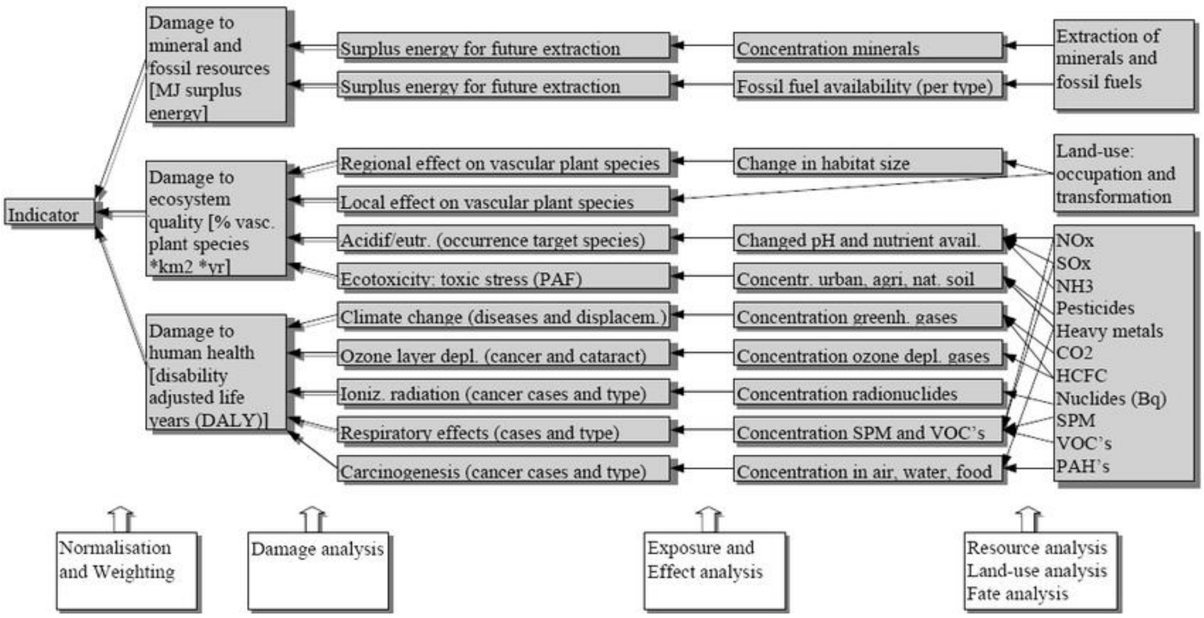


Figure 3.4: Graphical presentation of the Eco-indicator 99 methodology (Goedkoop et al. 2000, p. 5)

Finally, the CML 92 method was established by the Environmental Training Centre (CML) of the University of Leiden in Holland (Goedkoop et al. 2000). The impact categories that are measured are moderately easy to recognise: greenhouse effect conservatory, the ozone layer, ecotoxicity and human toxicity, power eutrophication, acidification, pollution, resources and solid wastes (Goedkoop et al. 2000). In addition, the most common methodology used is Eco-indicator 99 (Goedkoop et al. 2000). Itsubo and Inaba (2012) make a comparison between different methods of assessing the environment in Table 3.3, which illustrates the key differences of 11 methods of assessing the environment.

Table 3.3: Comparison between different methods of assessing the environment (Itsubo & Inaba 2012, p. 13)

Method	Country of development	Year of development/ renewal	Object of assessment			Remarks
			Characteri -zation	Damage assessment	Integration	
CML	Holland	2002 renewal	○			
EDIP	Denmark	2003 renewal	○			
TRACI	US	2003	○			
Eco-scarcity	Switzerland	2007 renewal			○	Midpoint modeling
JEPIX	Japan	2003			○	Midpoint modeling
Ecoindicator'95	Holland	1995			○	Midpoint modeling
Ecoindicator'99	Holland	2000		○	○	Endpoint modeling
EPS	Sweden	2000 renewal		○	○	Endpoint modeling
Impact2002	Switzerland	2002	○	○		
ExternE	Europe	2005 renewal			○	Endpoint modeling
LIME2	Japan	2008 renewal	○	○	○	Endpoint modeling

Figure 3.5 shows the details of the LIME2 method of assessing the environment (Itsubo & Inaba 2012), which consists of 19 inventory and 15 impact categories. There are some differences between Eco-Indicator 95 and 99 compared to LIME2, the Japanese method (Itsubo & Inaba 2012). The differences are in inventory, impact category and damage impacts (Itsubo & Inaba 2012).

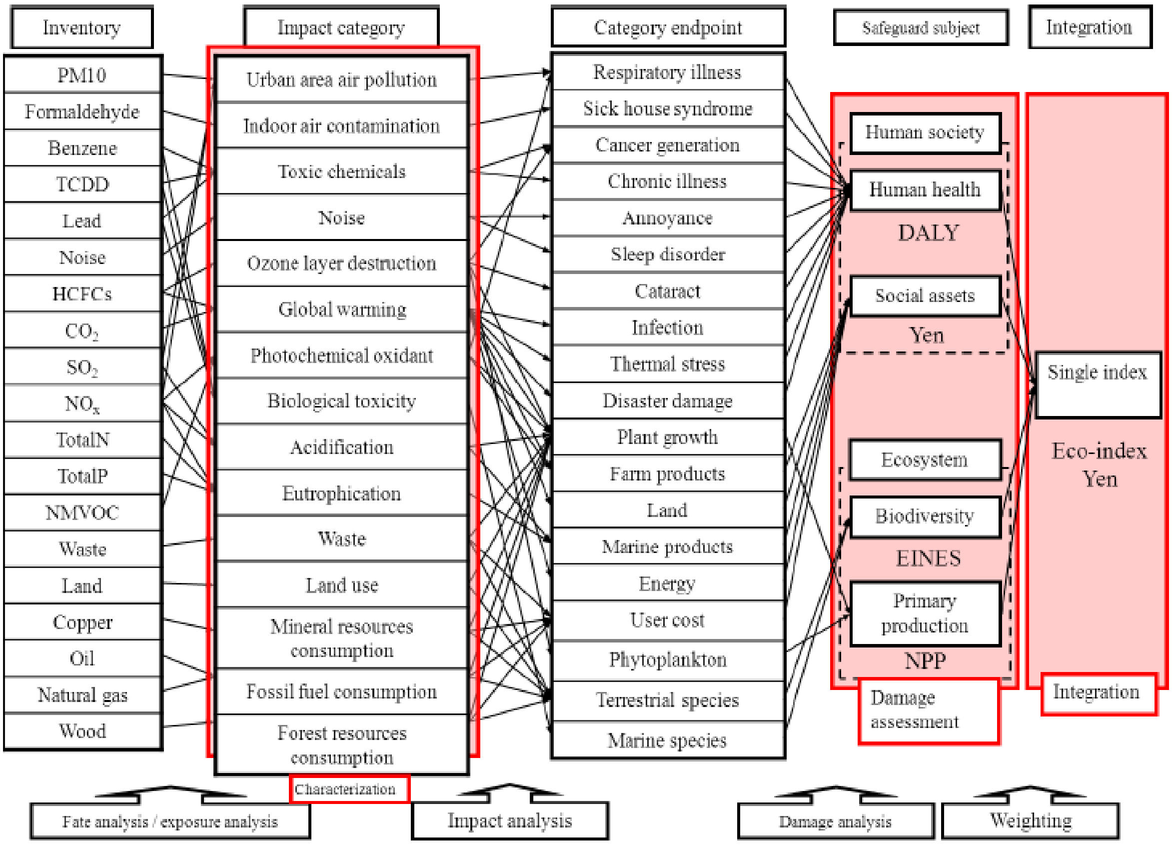


Figure 3.5: Assessment method of the LIME2 method (Itsubo & Inaba 2012, p. 2)

3.6. The Importance of LCA in Waste Concrete

Estanqueiro (2011) has elaborated in detail the importance of LCA, explaining that the construction industry is of essential financial and social importance for the development of all countries; however, it has severe environmental impacts due to its consumption of large amounts of natural assets and production of a massive volume of construction and demolition waste. A vast percentage of the C&D waste is deposited illegally, which causes problems for human health and the environment; consequently, correct management is essential (Estanqueiro, 2011).

As described by Wang and Gangaram (2014), LCA is a systematic method for assessing potential environmental problems and impacts throughout a product's life from raw material acquisition to production, use and disposal. Moreover, it helps to provide metrics that can be used to measure progress towards environmental sustainability (Wang & Gangaram, 2014).

Bovea and Powell (2016) clarified that, nowadays, analysis of construction materials based on LCA and an environmental point of views is very important in European recycling research. Oh et al. (2014) state that efforts have been made in Japan to use recycled materials in cement manufacture to reduce the CO2 emissions. Kolay and Akentuna (2014) observe that some studies suggest CO2 emission and energy consumption are appropriate benchmarks for the ecological influence of waste concrete.

Figure 3.6 shows the framework for the formal structure of LCA by the International Standards Organisation (ISO) 14040:2006. It shows three primary phases: goal and scope definition, inventory analysis and impact assessment. Interpretation of each phase is required to evaluate the completion of the analysis, which gives a conclusion, recommendation and limitation, as interpretation is beneficial for the product included in the LCA to find better options, strategies and applications.

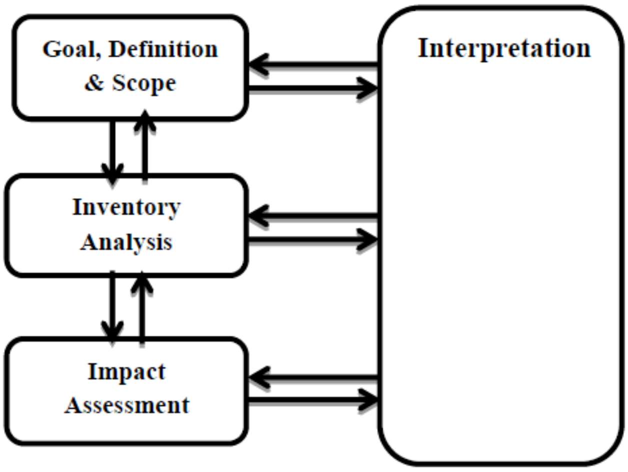


Figure 3.6: Framework of LCA by the International Standards Organisation (ISO) 14040:2006

The first and fundamental process in LCA is defining the meaning of the goal and scope of the procedure. In any LCA process, the objective is to evaluate and describe the stream of the considerable number of materials associated with the method, which helps in distinguishing the ecological effect of each material, and locating an effective way to deal with it to decrease the effect. LCA has risen as a broadly rehearsed procedure to diminish the destructive natural impacts, and it has given numerous helpful outcomes. Characterising the goal of any procedure is thought to be the most basic approach in starting an LCA assessment (Wang & Gangaram, 2014). Although, the goal is to characterise the inquiries that are to be addressed trailed by selecting the assessments of scope (Wang & Gangaram, 2014). The scope includes what and how the entire procedure will be depicted, and what choices should be characterised. The appraisal of the assets ought to likewise be done, which can also be connected to analysis. The phases include characterising the framework limits, suspicions and constraints of the system (Wang & Gangaram, 2014).

UNEP (2009) states that depletion, use and disposal of construction materials are affecting the environment. Kleijer et al. (2016) declared that concrete is the most used product worldwide, adding that the use of concrete on the planet leads to high CO2 emissions. LCA has been included in the international standard, for example, ISO 14040, as a principle and framework of environmental management (UNEP, 2009). LCA helps to analyse many materials regarding energy and environmental impact, which helps to find better solutions to protect the environment (UNEP, 2009). For more details, the Institute for Advanced Study of Sustainability (2016) states that cities account for 67% of global energy consumption, accounting for 70% of greenhouse gas emissions and 75% of natural resources depletion (Federal Ministry for Economic Cooperation and Development, 2014). On the other hand, the building sector represents 40% of natural worldwide energy consumption and 30% of greenhouse gas emissions (Federal Ministry for Economic Cooperation and Development, 2014). LCA identifies the environmental aspects and impacts of product life cycle that include use and end of life levels (Weiler et al. 2017). Furthermore, ISO 14040 and 14044 have defined the LCA requirements (Institut für Normung, 2006 & 2009).

Figure 3.7, reproduced from Weiler et al. (2017), shows the LCA divided into stages and sub-stages. It starts with production, which is extraction and manufacturing, transportation and assembly. Secondly, the use of the material itself follows, and then the last stage, which is the end of life, which consists of sub-stages, for instance, demolition, transportation and treatment method. All stages require energy use and emit greenhouse gases into the atmosphere, whether during processing, using or disposal. The main three phases of LCA have release different levels of emissions and have different effects on the environment due to energy consumption (Weiler et al. 2017).

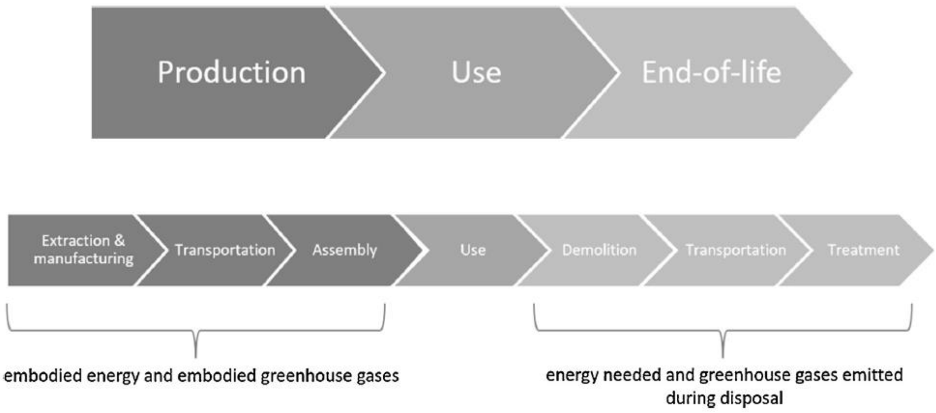


Figure 3.7: Division of LCA (Weiler et al. 2017, p. 321)

As stated by Khasreen (2011) , LCA can evaluate the environmental impact of building materials. Building emissions such as CO2 and SO2 have a possible effect on the environment (Khasreen, 2011). Operation construction process, demolishing and depletion of natural resources have a severe impact on climate (Khasreen, 2011). Furthermore, Porhinčák and Eštoková (2013) declare that building materials have an adverse impact on occupational health and the environment. It has been shown, as illustrated by Bribian, Capilla and Uson (2011), that 24% is the percentage of construction that used as depletion of raw materials.

On the other hand, Figure 3.8, from Weiler et al. (2017), shows the cradle to grave of the LCA process. From an overall perspective, this is part of a circular process, where the final product is created and in use until the end of life, which means disposal of the material or product. Moreover, during the end life of the product, emissions are released. The whole process, based on Figure 3.7, is obvious the life cycle process of material or product starting from manufacturing stage to the disposal after being used (Weiler et al. 2017). It is obvious that emissions are emitted from the preparation stage and disposal stage (Weiler et al. 2017). The use of the LCA process helps to trace and monitor the life cycle of a material or product and identify how much energy is consumed and the impact on the environment, which helps to find better management process, material and energy consumption, to reduce the influence on the environment (Weiler et al. 2017).

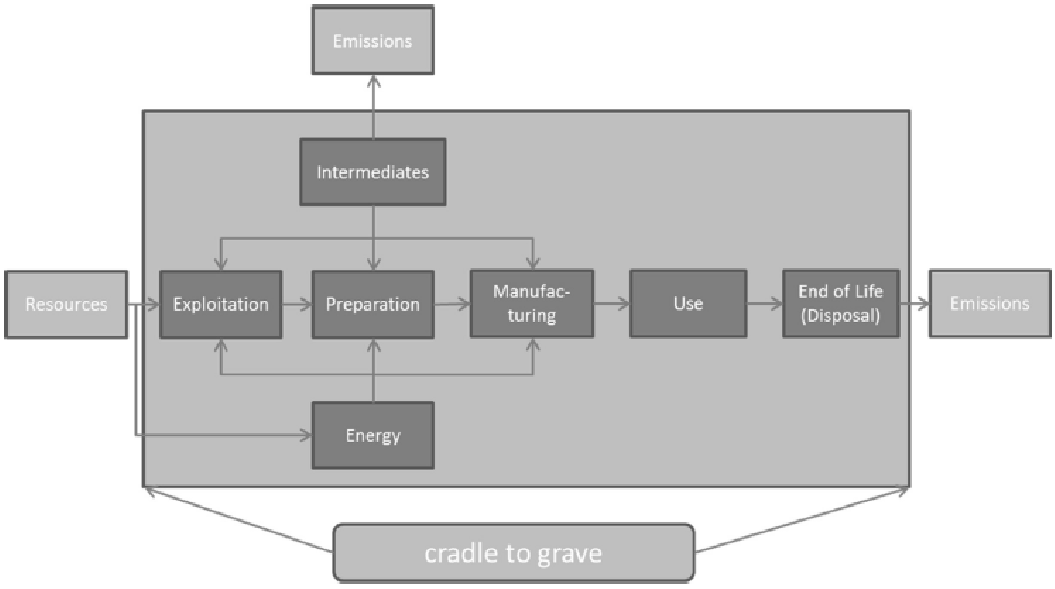


Figure 3.8: Cradle to grave of LCA process (Weiler et al. 2017, p. 321)

Wang and Gangaram (2014) defined the inventory analysis, otherwise called a life cycle inventory (LCI). An inventory analysis evaluates the inventory flow for an item or process from cradle stage to end stage. It incorporates contributions of water, energy and raw materials to air, water and soil. The inventory model is developed as a flow diagram, and it incorporates the input and output data into a framework that is viewed as a flow model which utilises the information of the specialised framework. This data is gathered by the technical system boundaries. The information is comprised of products starting from raw material to the end of life/recycle stage. Data are identified with the goal characterised for the LCA (Wang & Gangaram, 2014).

Life cycle assessment of aggregate, which is called cradle-to-grave and cradle-to-gate, and is illustrated in Figure 3.9 from Ghanbari, Abbasi and Ravanshadnia (2017), shows the first step, which is extracting the aggregate from the mountain. The second step is to transfer the raw materials extracted from the mountain to the crusher plant, which is the place for crushing, processing and material storage. The third step is to use the aggregate, whether in concrete or asphalt plant for production. The fourth step is the use of the final product for the construction site or road activities. The fifth stage is the disposal of C&D waste to landfill. The last stage is the use of waste material for the recycling method, which recycles the waste and reuses it again in concrete or asphalt plant as recycled aggregate (Ghanbari, Abbasi & Ravanshadnia, 2017).

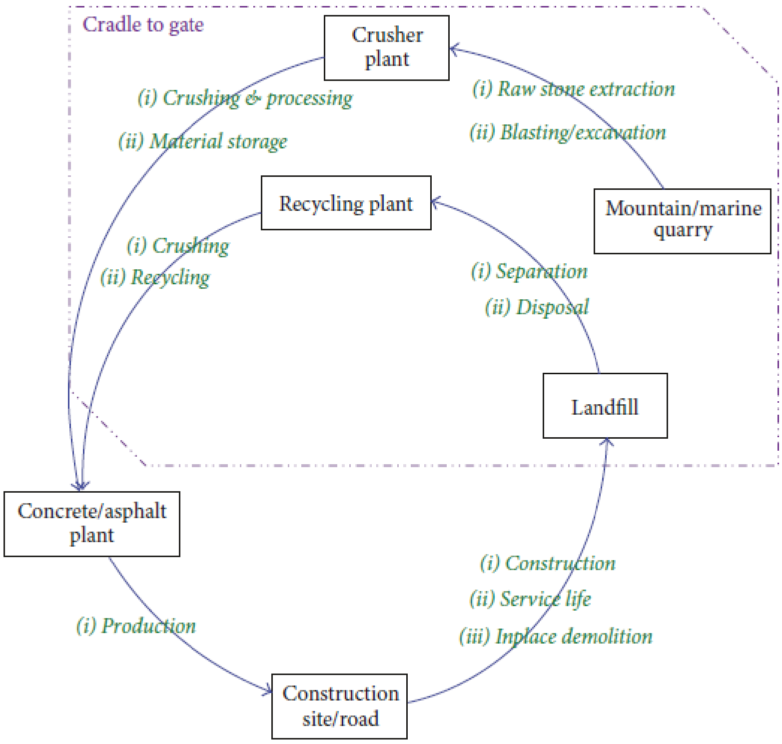


Figure 3.9: LCA of aggregate form cradle-to-grave and cradle-to-gate, (Ghanbari, Abbasi & Ravanshadnia 2017, p. 3)

Figure 3.10 from Ghanbari, Abbasi and Ravanshadnia (2017) shows the input of energy, CO2 and cost of the aggregates process in two scenarios. Scenario one starts from the extraction phase, which is as excavation and transportation of raw material to crushing. Second is the crushing phase, which includes storing the aggregates after they have been crushed. Scenario two is pre-crushing of C&D waste, which supplies the crushing process phase, then storage (Ghanbari, Abbasi & Ravanshadnia, 2017).

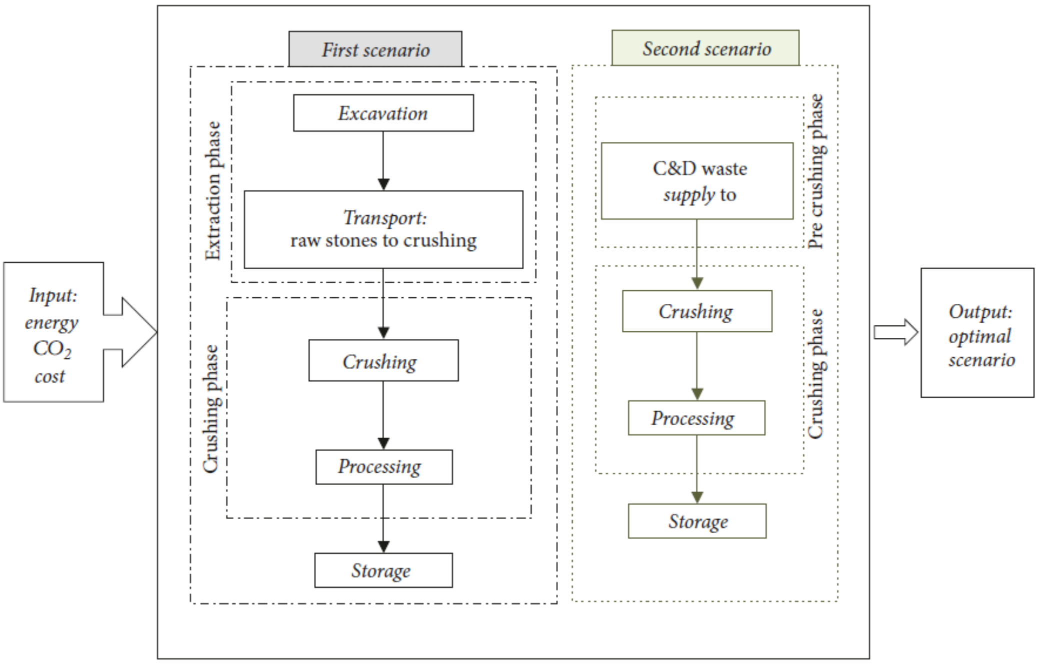


Figure 3.10: Input of energy, CO2 and cost of aggregates process in two scenarios (Ghanbari, Abbasi & Ravanshadnia 2017, p. 3)

On the other hand, Figure 3.11 from Estévez, Aguadoa and Josaa (2006) illustrates the complete life cycle of concrete products. It starts from the extraction of raw material, which is primary aggregates, then moves on to the manufacturing of cement, to concrete as the final product used in the construction of the building, which requires rehabilitation and maintenance during its life. In addition, each step is linked by the transportation system to transport the raw material and final product, and take the concrete waste after the demolition stage to a landfill site or recycling plant; in the latter, it will be recycled for reuse as secondary aggregates (Estévez, Aguadoa & Josaa, 2006).

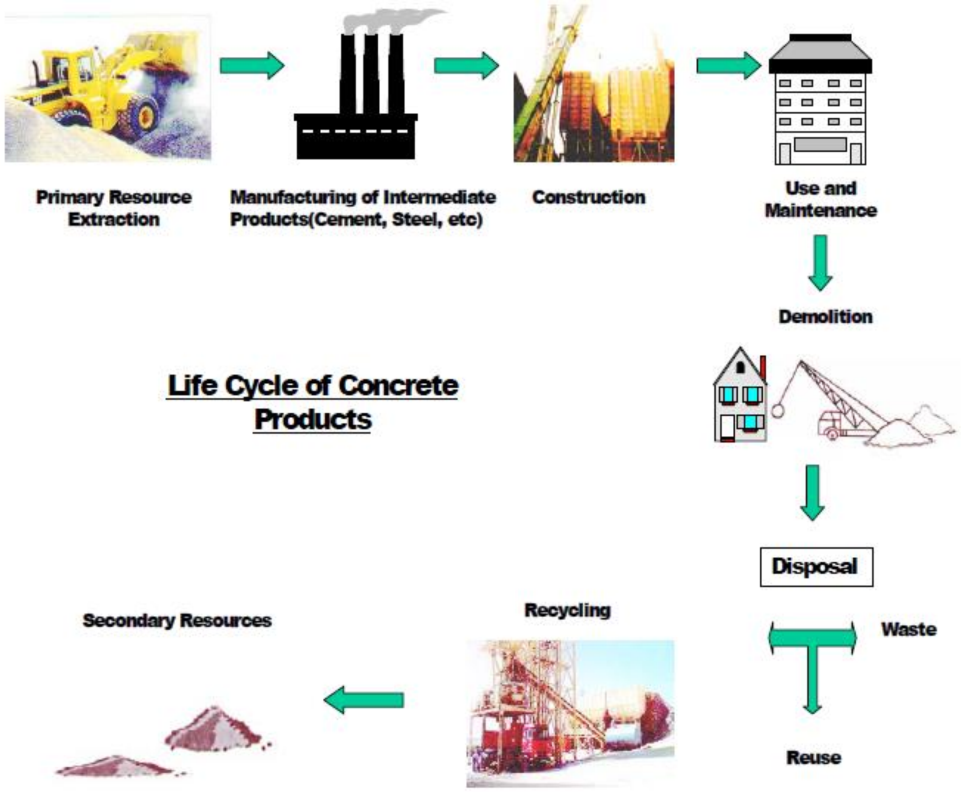


Figure 3.11: Complete LCA of concrete products (Estévez, Aguadoa & Josaa, 2006, p. 3)

Based on the advantages of the recycling method, Zaho, Leeftink and Rotter (2010) have argued that recycling of aggregates can reduce the energy consumption and greenhouse gases such as CO2 that are related to industrial activities, and this will result in economic and environmental benefits. Moreover, the recycled aggregates can be used in concrete production (Sabai et al. 2013) and road construction (Petkovic, 2004).

On the other hand, a number of authors (Vold & Ronning, 1995; Young et al., 2002; Björklund & Tillman, 1996-1997; Häkkinen & Mäkelä, 1996; Lundström et al., 1996; Lundström, 1997; Geem, 1998; Nisbet & Geem, 1997) have shown that numerous LCA studies have been conducted on cement, concrete and concrete products related in phases of their life cycle that have a main influence on the environment. Moreover, Greece is producing 14 million tonnes of cement per year, 50% of which is exported (Koroneos & Dompros, 2009). In addition, 7 million tonnes of that used for internal consumption is used for manufacturing 30 million m3 of concrete (Koroneos & Dompros, 2009). The cement industry’s consumption emits a high percentage of CO2 and energy, which results in important quantities of both energy-related and process (Price et al., 1999; Hendriks et al., 1998; Martin et al., 1999); Worrell et al., 2001). The efficient use of energy results in reduced costs and decreased emissions. Moreover, as illustrated by Bahr et al. (2003), Gabel et al. (2004) and the IPPC (2000), the increasing awareness of sustainable development leads to an increase in the importance of assessing the corresponding environmental performance of the huge manufacturing segment. Furthermore, Koroneos and Dompros (2009) state that raw materials and rational energy management will limit the emission of harmful contamination into the environment and save energy. To ensure its effectiveness, it requires systematic decision-making tools that will provide the compulsory data for identification of potential enhancements in the concrete life cycle (Koroneos & Dompros, 2009). The composition of cement and its role in LCA is important because it contributes to CO2 emissions (Koroneos & Dompros, 2009). Cement is a fine, granular powder that has hydraulic features. It consists of calcium oxides, silicon, aluminium and iron, which join together to represent 90% of its mass. When cement is mixed with water, it becomes hard and durable in the air or below water Figure 3.12 illustrates the system boundaries of the cement and concrete life cycle (Koroneos & Dompros, 2009).

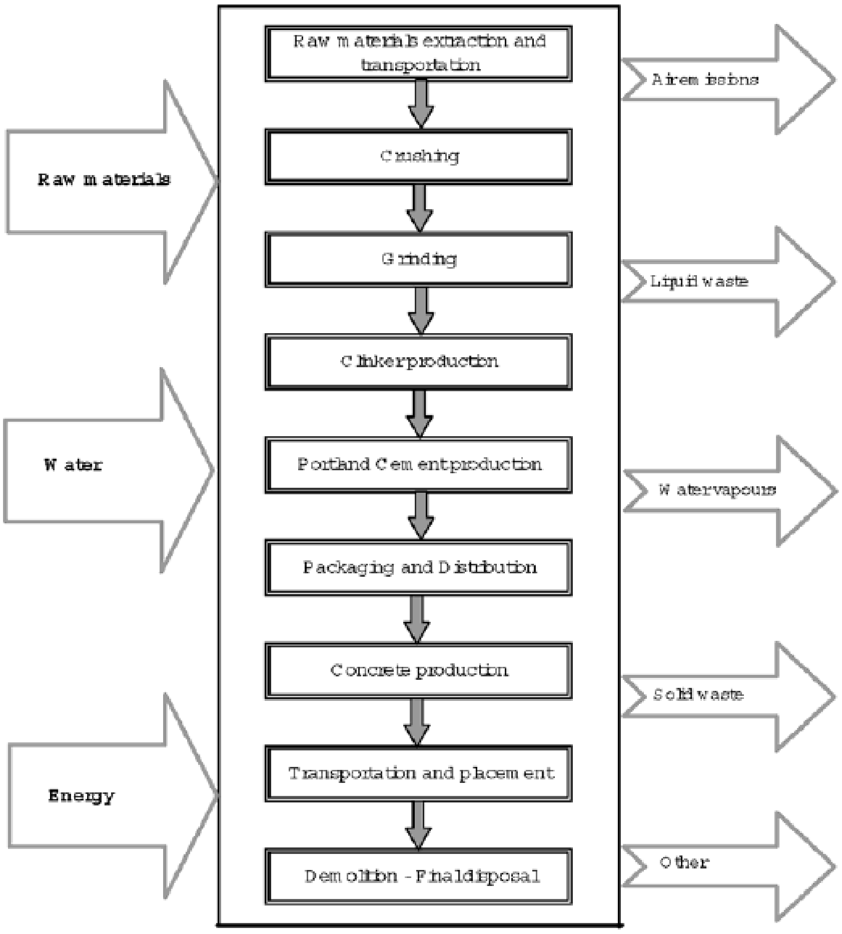


Figure 3.12: System boundaries of cement and concrete life cycle (Koroneos & Dompros 2009, p. 74)

The first stage in Figure 3.12 starts with raw material extraction and transportation, which causes emissions into the air during crushing, grinding and clinker production for cement, which causes liquid waste. The second stage is the process of Portland cement production packaging and distribution and concrete production. This stage includes water vapour and solid waste generation. The last stage is energy use, which requires transportation and placement of concrete or demolition and disposal, which cause pollution (Koroneos & Dompros, 2009).

3.7. Methods for Monetising Environmental Indicators

Monetisation is an approach used to signify the impact category based on monetary value (Pre-sustainability 2018, p. 1). In addition, the value is based on costs related to preventing or refurbishing the damage; for instance, one kind of impact could be more expensive to prevent or fix than another. The monetisation impact category help to estimate how willing people and society need to be to spend money to prevent certain damage. For example, people might be willing to pay to protect human health more than to minimise resource depletion to prevent impacts and damages. On the other hand, the issue with the monetisation model is that it fundamentally involves an answer to the question of how much the damage is satisfactory and how much human life is valued, while the answer is at best subjective. Furthermore, the monetisation process signifies the impacts that are linked with social and natural capital in monetary value, which locates them as more tangible. In addition, monetising environmental influences are known as natural capital valuation, which is considered a kind of sustainability return on investment analysis and delivers a different financial perception of sustainability (Pre-sustainability 2018, p. 1).

Another method is to find the costs correlated with a particular activity, for example, the cost of damage or the costs to replace the service (Pre-sustainability 2018, p. 1). For instance, the social cost of carbon (SCC) signifies the economic damages related to an increase of CO2 impacts. However, monetary methodologies might be variable and include original value decisions. Moreover, critics argue that monetisation could oversimplify complex concerns and it is difficult to quantify or monetise every ecosystem value (Pre-sustainability 2018, p. 1).

There are costs associated with ecological damages imposed on society (Nguyen et al. 2016). For instance, there are several existing methods to evaluate the monetisation of environmental damages as classification endpoints or safeguarded subjects such as ecosystems, humans and resources. In addition, the methods of evaluating monetisation of environmental damages are European models such as EPS 2000 by Steen (1999), the ExternE project by Bickel and Friedrich (2005), Ecotax 2002 by Finnveden et al. (2006) and Stepwise2006 by Weidema (2009). Moreover, LIME is a Japanese model by Itsubo et al. (2004), which is a non-European model. On the other hand, most of the monetary assessment models are not fully completed, since not all ecological impacts are subjected to monetisation (Bos & Vleugel, 2005).

The Handbook Environmental Prices (2017) was used as a method for cost valuation of ecological impacts by CE Delft (2018). Moreover, the handbook contains the environmental prices of many impacts. In addition, environmental prices are applicable for social cost or pollution, which are expressed in euros per kilogram pollutant. Moreover, the environmental prices specify the damage to economic welfare that occurs when one added kilogram of the pollutant makes its way into the atmosphere. Environmental prices can also be determined for immaterial forms of pollution such as noise nuisance and ionising radiation. Furthermore, they provide average values for the Netherlands, for instance, emissions from a common source of emission with an average emission site in the year 2015. The handbook presents the prices at three levels Firstly, at the pollutant level, which gives the environmental emissions values of damaging substances. Secondly, at the midpoint level, which gives values for environmental themes, for example, climate change or acidification. Thirdly, at the endpoint level, which a value for the environmental impacts of pollution, for instance, damage to human health or ecosystem services (Environmental Prices Handbook, 2017).

The methodology used in this Environmental Prices Handbook is designed to harmonise the values at pollutant, midpoint and endpoint level, to achieve reliable estimation of the influences or pollution in the Netherlands. Figure 3.13 shows an overview of the relationships covered in this handbook, with each arrow representing a relationship that has been mapped.

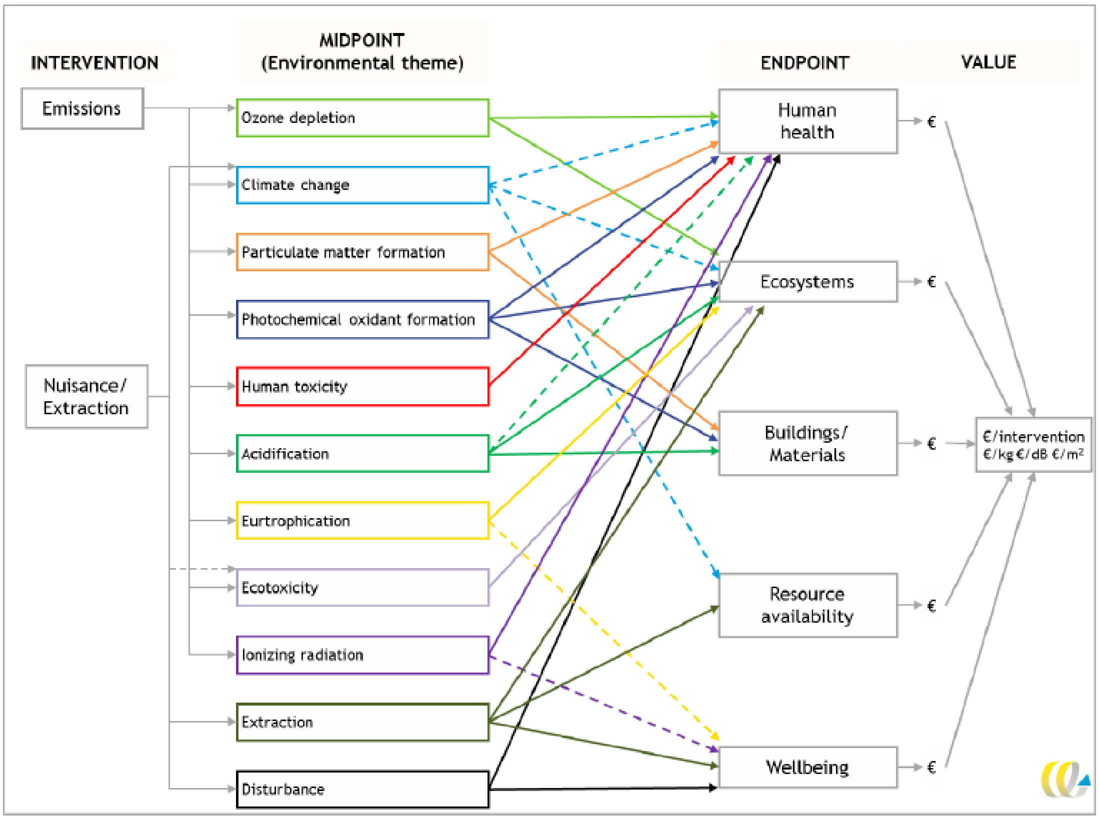


Figure 3.13: The relationship mapping and methodology of Environmental Prices Handbook (2017, p. 4)

To use weighting factors in LCAs, two values have to be established (Handbook Environmental Prices, 2017). Firstly, a value based on external costs, which is applicable for an individualist perspective in ReCiPe (Handbook Environmental Prices, 2017). Secondly, a value for use as a weighting factor that is entirely applicable for the hierarchist perspective in ReCiPe (Environmental Prices Handbook, 2017).

The relationship between the many stages involved in environmental prices, as per the Handbook Environmental Prices (2017), is shown in Figure 3.13, which consists of emissions, midpoints, endpoints, valuation and related fields of study such as ReCiPe, which is used in this study (Environmental Prices Handbook, 2017).

Table 3.4: Environmental price in euros for each impact category as per each unit (Handbook Environmental Prices, 2017)

No.	Impact category	Unit	Price in Euros
1	Climate change	€/kg CO2-eq.	0.057
2	Ozone layer depletion	€/kg CFC-eq.	30.4
3	Human toxicity	€/kg 1,4 DB-eq.	0.214
4	Acidification	€/kg SO2-eq.	5.4
5	Freshwater eutrophication	€/kg P-eq.	1.9
6	Marine eutrophication	€/kg N	3.11
7	Land use	€/m2a	0.0261
8	Terrestrial ecotoxicity	€/kg 1,4 DB-eq.	8.89
9	Freshwater ecotoxicity	€/kg 1,4 DB-eq.	0.0369
10	Marine ecotoxicity	€/kg 1,4 DB-eq.	0.00756
11	Human toxicity	€/kg 1,4 DB-eq.	0.214
12	PM2.5	€/kg PM2.5 eq.	79.5
13	Nitrogen oxides (Nox) (Human health)	€/kg NOx eq.	18.7
14	Mineral resource scarcity (Atmospheric)	$/kg Cu eq	4.2
15	Mineral resource scarcity (Soil)	$/kg Cu eq	0.239

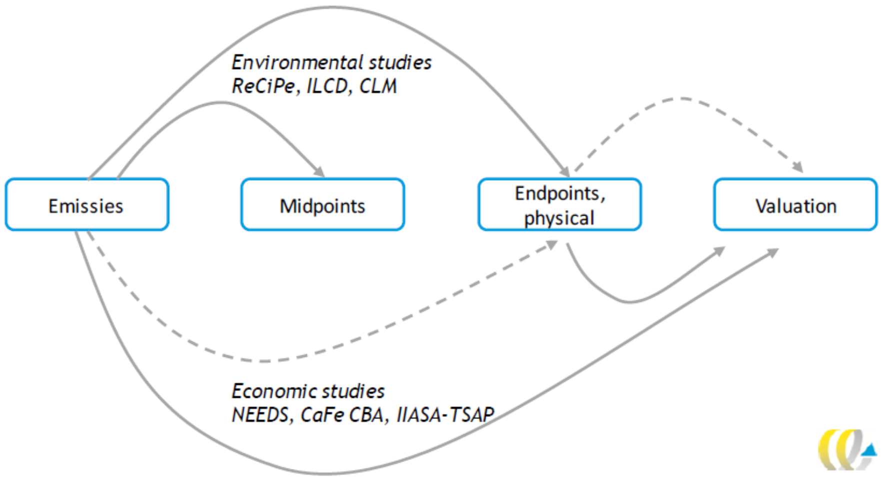


Figure 3.14: Relationships between emissions, midpoints, endpoints, valuation and related fields of study (Handbook Environmental Prices 2017, p. 26).

The objective of ecological prices is to find the damage costs that impact on the environment, which helps to consider better solutions in order reduce the ecological impacts and costs, which results in achieving sustainability (Environmental Prices Handbook, 2017).

Figure 3.15: Challenges and solutions to achieve sustainability (Handbook Environmental Prices 2017, p. 35)

LCA studies are growing, and the integration of translating the planning and process in real measures is a challenge (Environmental Prices Handbook, 2017). On the one hand, monetisation of the ecological impact helps to express the emissions in cost units, which is based on incentives such as taxes, fees and bounces reduction (Beckenbach et al. 1998). On the other hand, there are two main approaches to monetisation (Beckenbach et al. 1998). One approach is willingness to pay the environmental damage cost, which determines the amount of money that an individual or organisation should pay to cover the expenses of environmental impacts (Beckenbach et al. 1998). However, willingness is individualist and depends on certain criteria, for example, education, region or existing finances (Krieg, Albrecht, Lindner & Jäger, 2013).

Moreover, the subjectivity of monetary can fluctuate from the value of actual monetary (Reap et al. 2008). The second approach measures environmental damage cost by the concrete damage triggered by emissions (Krieg, Albrecht, Lindner & Jäger, 2013). However, as they note, there are many uncertainties related to this approach, which leads to complexity regarding predictions of cost and variety in impact values. inhuman activities are causing ecological impacts to fluctuate at an extraordinary level (Krieg, Albrecht, Lindner & Jäger, 2013). In addition, the amount of greenhouse gases and ozone depletion in the climate causes a reduction in the lifetime of species, interruption of biogeochemical phases, deforestation and depletion of natural resources associated with human activity (Spangenberg, 2007). The demand of how economic activities distress on the environment has become general between scholars in 1960s (Spangenberg, 2007). Moreover, historical records show the link between human activity and environmental damage (Asıcı, 2012). The sustainability of the environment along with social and economic sustainability creates three pillars of sustainability (Moldan et al., 2011). Although, it was defined by Goodland (1995) that enhancing human health and wealth can be achieved by saving the raw material sources which are used for human requirements. In addition, Goodland (1995) states that the human waste should not exceed the limit in order to avoid any harm and damage to humans themselves. In addition, the growth of the economy has concerns for the environment both domestically and globally (Asıcı, 2012). The Environmental Kuznets Curve (EKC) is an approach to analyse the impacts of economic growth on different environmental quality measurements (Asıcı, 2012). However, environmental quality has numerous measurements and grouping those dimensions into a single indicator is unreliable (Asıcı, 2012). Furthermore, there are a variety of studies about EKC that have initiated different indicators, for example, carbon dioxide, sulphur dioxide emissions (Boulatoff & Jenkins, 2010; Grossman & Krueger, 1991), urban air quality (Esty & Porter, 2005), deforestation (Martinez et al., 2002), heavy metal contamination (Grossman & Krueger, 1995) and waste (Mazzanti et al., 2009). In addition, it is not applicable for a single curve to explain all kinds of environmental degradation, because it raises uncertainties about the generalisability of the EKC hypothesis (Özler & Obach, 2009).

Decision support tools such as Cost-Benefit Analysis (CBA) have been established to support the process by decision-making. Moreover, the importance of CBA in site remediation was initiated by Bonnieux et al. (1998) and presently it is considered to be a valuable tool based on the decision-making process (European Commission, 2014; Guerriero & Cairns, 2011a, 2011b). There are some monetisation approaches; for example, Stepwise 2006 is a monetisation technique that can be comprehensively applied in the previous SimaPro software version (European Commission, 2014; Guerriero & Cairns, 2011a, 2011b). Furthermore, it provides the users with three valuable values: Human well-being, Biodiversity and Resource productivity, which are associated with the three pillars of sustainability, people, planet and profit (European Commission, 2014; Guerriero & Cairns, 2011a, 2011b). In addition, the calculation process can be linked with the LCIA stage and the results signified in Quality Adjusted Life Years (QALY) for human well-being, Biodiversity Adjusted Hectare Years

(BAHY) for biodiversity and euros for Resource Productivity (European Commission, 2014; Guerriero & Cairns, 2011a, 2011b). Another monetisation approach is Ecotax 2002, which is based on the Swedish eco-taxes and fees that are correlated with emissions and resource use by assuming decision making on the part of politicians reflects on the societal values of environmental impacts (Huysegoms, Rousseau & Cappuyns, 2018).

Environmental Priorities Strategies (EPS) in product design is one of the oldest monetisation models (European Commission, 2014; Guerriero & Cairns, 2011a, 2011b). Furthermore, the EPS model is one of the most primitive monetary valuation models that has been established to simplify the assessment of environmental impacts between the concept of the product in product improvement (European Commission, 2014; Guerriero & Cairns, 2011a, 2011b). While the primary version of the EPS model was initiated in 1991 to 1992, the most topical version was introduced in 2000 (Steen, 1999). In addition, the model estimates the external costs of products using inventory data such as data based on emissions, energy consumption, etc., while characterisation factors, for instance, influence per unit of environmental interference and weighting factors, for instance, weights of impact, are expressed in monetary units (Nguyen et al., 2016). The EPS model is a top-down one, which allocates the highest importance to the effectiveness of the scheme (Nguyen et al., 2016). Moreover, the principle is used in certain deals with a complex model, in which parts are not recognised or unbearable to include because of limited resources (Nguyen et al., 2016). Furthermore, Environmental Priority Strategies (EPS) in product design express the results based on damage cost of emissions and use of natural resources by using Environmental Load Units (ELUs) (ivl 2019, p. 1). In addition, one ELU signifies an externality equivalent to one euro environmental damage cost (ivl 2019, p. 1).

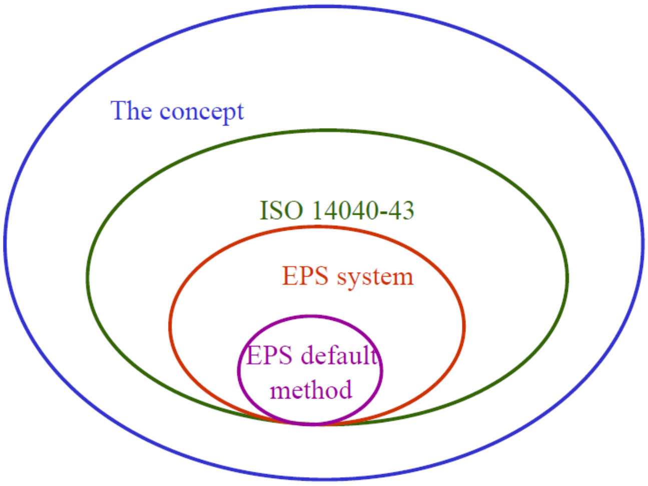


Figure 3.16: The link between LCA concept, ISO standard, EPS system and EPS default method (CPM report 1999:5)

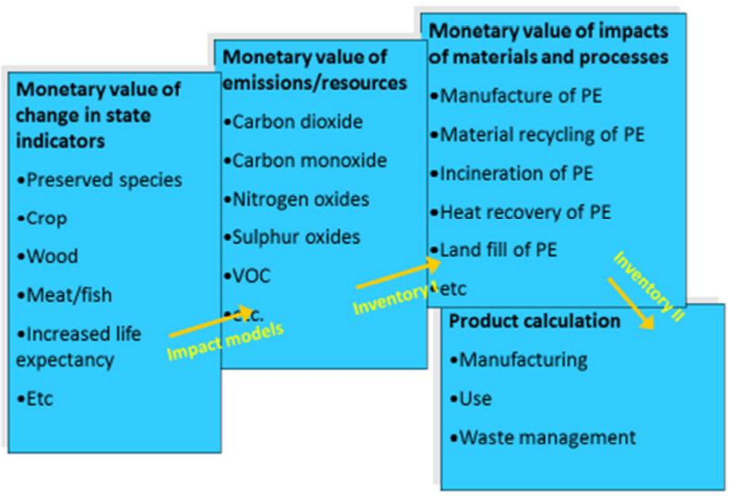


Figure 3.17: The structure of the EPS system (Steen, 2015)

EPS 2015 consists of three features such as 24 kinds of characterisation, and six damage assessments and weightings, which are Ecosystem Services, Access to Water, Biodiversity, Building Technology, Human Health and Abiotic Resources (Inaba, 2013). In addition, uncertainty is connected with the factors that are appropriate for monetising weight in different categories of impact or damage, because of subjective preference regarding the value (Inaba, 2013). In ReCiPe 2016 LCIA, two midpoint impact categories are associated with monetisation, which are Mineral Resources and Fossil Resources. Furthermore, the impact of Mineral Resources causes increases in extraction cost, which leads to damage to resource availability (RIVM Report 2016-0104), while Fossil Resources is correlated to the cost of oil, gas, coal and energy, which lead to damage to resource availability (RIVM Report 2016-0104).

3.8. Summary

This chapter identified some important aspects of concrete as a construction material. First of all, it has discussed the high percentage of the actual usage of concrete worldwide and the issues behind it. Secondly, it has examined the environmental impact of concrete, from manufacture with raw materials to demolition stage. Thirdly, it was found that concrete products can be recycled to save the environment and reduce the waste going into landfill. Finally, the effective methods of assessing concrete waste were introduced, such as Eco-indicator 95 and 99, which can predict some of the causes and effects of different types of waste. Furthermore, some assessment methods were used on concrete products and analysed by software called SimaPro. There are over 11 methods for assessing the environmental impact. In addition, the Handbook Environmental Prices (2017) was found to be the most suitable and applicable source to find the damage cost from LCA results by using ReCiPe2016 LCIA, since the units are similar for both, which will be discussed in Chapter 4 of this research.

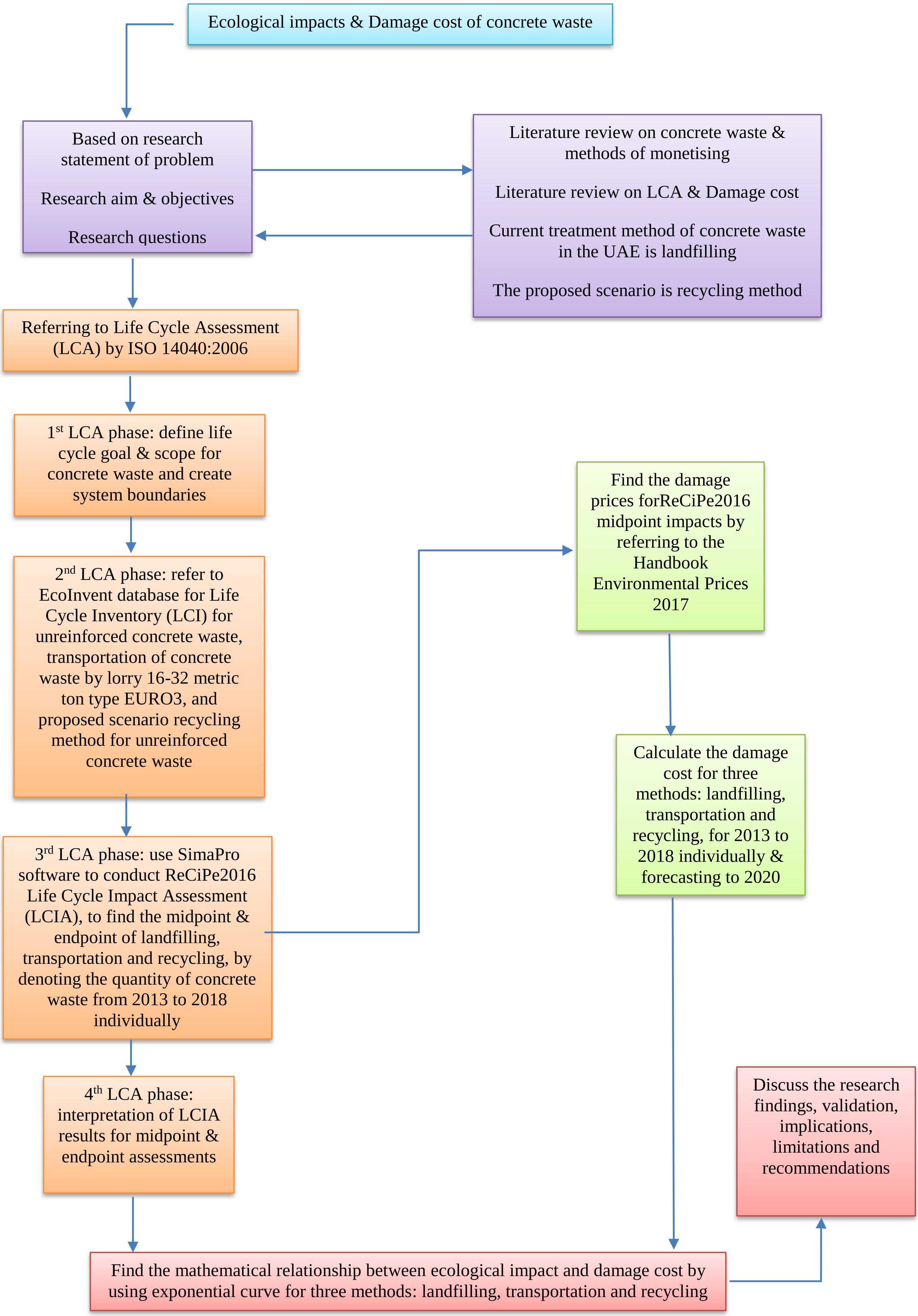
CHAPTER 4: METHODS FOR MODELLING THE ECOLOGICAL IMPACT OF CONCRETE WASTE BY USING LCA

4.1. Introduction

This chapter discusses the available methods for modelling LCA used in various studies. In addition, the chapter introduces the importance of LCA and its recommended phases, which are based on the commonly used international standard ISO14040:2006. Around 1970, life cycle assessment (LCA) was invented in the USA at the Midwest Research Institute.

It became the main topic for analysis of environmental protection and energy saving. The meaning of LCA, as described by Kolpffer (1997), is that it connects all ecological products or services, from raw material to waste disposal. In addition, LCA, as it is known from the cradle to the grave, has been used in many countries and for different types of building material products to assess the effects of these materials on the environment. As described by Barbieri and Cajazeira (2009), LCA is an environmental management tool that can be applied to selected services of a particular company. LCA is used to identify the environmental elements at the product level to determine key areas of environmental improvements, which view the new product. Nowadays, the LCA method is widely used and is practical for the C&D waste management segment, although it can also be applied to recover and reuse materials (Hossain et al., 2016; Butera et al., 2015; Dahlbo et al., 2015; Kucukvar et al., 2016). Dahlbo et al. (2015) used LCA to assess the C&D waste performance in Finland with regard to the environment and economic aspects, in order to identify the potential of recycling materials which allow the country to achieve the EU target of 70% by 2020.

Figure 4.1: Research modelling framework presenting the thesis roadmaps


Ecological impacts & Damage cost of concrete waste
Based on research statement of problem
Research aim & objectives
Research questions
Literature review on concrete waste & methods of monetising
Literature review on LCA & Damage cost
Current treatment method of concrete waste in the UAE is landfilling
The proposed scenario is recycling method
Referring to Life Cycle Assessment (LCA) by ISO 14040:2006
1st LCA phase: define life cycle goal & scope for concrete waste and create system boundaries
2nd LCA phase: refer to EcoInvent database for Life Cycle Inventory (LCI) for unreinforced concrete waste, transportation of concrete waste by lorry 16-32 metric ton type EURO3, and proposed scenario recycling method for unreinforced concrete waste
3rd LCA phase: use SimaPro software to conduct ReCiPe2016 Life Cycle Impact Assessment (LCIA), to find the midpoint & endpoint of landfilling, transportation and recycling, by denoting the quantity of concrete waste from 2013 to 2018 individually
4th LCA phase: interpretation of LCIA results for midpoint & endpoint assessments
Find the damage prices forReCiPe2016 midpoint impacts by referring to the Handbook Environmental Prices 2017
Calculate the damage cost for three methods: landfilling, transportation and recycling, for 2013 to 2018 individually & forecasting to 2020
Discuss the research findings, validation, implications, limitations and recommendations
Find the mathematical relationship between ecological impact and damage cost by using exponential curve for three methods: landfilling, transportation and recycling

4.2. LCA Phases

Based on ISO14040:2006 Environmental Management of Life Cycle Assessment: Principles and Framework, the LCA consists of four phases, which are goal and scope definition, inventory analysis, impact assessment and, finally, interpretation of results. Furthermore, the phases should be conducted sequently (ISO14040:2006).

4.2.1. Life Cycle Goal and Scope Definition

The main reason for using LCA in this research is to determine the environmental, economic and social impacts of the three disposal options for C&D waste, which are natural aggregates and recycling concrete aggregates from the production plant, sorting plant and landfill. The research considers concrete as the selected construction material waste. Moreover, the study will include an emphasis on emissions from three waste disposal options for concrete activities such as transportation, cost, energy consumption and CO2 emission. Furthermore, concrete waste will be calculated based on government statistics report on the quantity of concrete waste quantity in the landfills in the UAE.

It was stated by Baumann et al. (1994) that system boundaries should be detailed in many scopes, for example, boundaries among nature and technological system, geographical zone, period horizon, production of investment goods, limitations between the life cycle of the studied product and correlated life cycles of other products.

The modelling framework of construction waste (Yahya, Boussabaine & Alzaed, 2016), as illustrated in Figure 4.8 starts with a definition of the goal and scope, which includes functional unit and inventory requirements, by EcoInvent database. The second stage is to classify and characterise the metal waste. The modelling framework of this research is provided in Chapter 5.

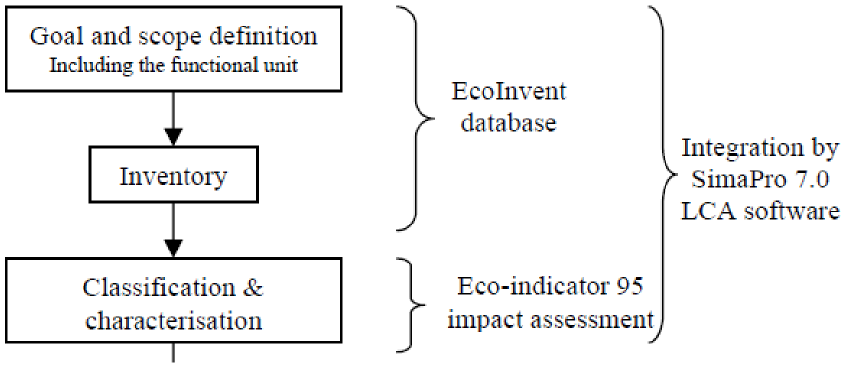


Figure 4.2: Modelling framework for construction waste (Yahya, Boussabaine, & Alzaed 2016, p. 6)

4.2.2. Life Cycle Inventory

The Life Cycle Inventory (LCI) is one of the important stages in the LCA (ISO14040:2006). The Swiss database EcoInvent inventory has been selected for this research because the methodology used in developing the database includes the input and output parameters that are eligible for current conditions in the GCC countries, including the United Arab Emirates, such as geographical area, technology, markets, products and services consumption. The full inventory for this research is provided in Chapter 5.

4.2.3. Life Cycle Impact Assessment

The third step in LCA stages is the Life Cycle Impact Assessment (LCIA), the objective of which is to measure the potential environmental impacts and their consequences, based on LCI results (Verones, Hellweg &, Huijbregts, 2015). Moreover, LCIA is used to ensure that the performed assessment is valuable environmentally (ISO14040:2006). This phase will be conducted in this research in Chapter 5 as midpoint and endpoint assessments by referring to the ReCipe2016 method.

4.2.4. Interpretation

The last phase after the LCIA generates results is an interpretation of the results, which gives a comprehensive evaluation of the assessment parallel with the benefits to develop or find a product with better options, strategies and applications. Moreover, it shall reflect based on LCIA results (ISO 14040, 2006). This phase of LCA will be discussed in Chapter 6 in this research.

4.3. Monetisation of Concrete Waste

Damage price will be used by referring to the Handbook Environmental Prices (2017), which was used as a method and cost for valuation of ecological impacts by CE Delft (2018). Moreover, the handbook contains the environmental prices of many impacts. In addition, environmental prices are applicable for social cost or pollution, which is expressed in euros per kilogram pollutant. Further, the environmental prices specify the damage to economic welfare that occurs when one added kilogram of the pollutant makes its way into the atmosphere (Handbook Environmental Prices, 2017). Environmental prices can also be determined for immaterial forms of pollution such as noise nuisance and ionising radiation (Handbook Environmental Prices, 2017).

The only prices that are applicable for the global method of LCIA by ReCipe2016 as selected for this research are in the Handbook Environmental Prices (2017), since it was initiated by the CE Delft organisation in the Netherlands to reflect global prices, which is the same country that developed the ReCipe 2016 method. In the UAE, there is no local LCIA method or environmental prices developed specifically for the region. Moreover, it has been found that 15 prices in the Handbook Environmental Prices are applicable for 15 impacts out of 18 in LCIA by ReCipe2016 in terms of impact type and impact unit.

Furthermore, environmental prices provide average values for the Netherlands, for instance, emissions from a common source of emission with an average emission site in the year 2015 (Handbook Environmental Prices, 2017). The handbook present the prices at three levels. Firstly, at the pollutant level, which provides environmental emissions values for damaging substances (Handbook Environmental Prices, 2017). Secondly, at the midpoint level, which provides values for environmental themes, for example, climate change or acidification (Handbook Environmental Prices, 2017). Thirdly, at the endpoint level, which provides values for the environmental impacts of pollution, for instance, damage to human health or ecosystem services (Environmental Prices Handbook, 2017).

To use weighting factors in LCAs, two values need to be established. Firstly, a value based on external costs, which is applicable for the individualist perspective in ReCiPe2016. Secondly, a value for use as a weighting factor that is entirely applicable for the hierarchist perspective in ReCiPe2016 in the Environmental Prices Handbook (2017), which will be used in Chapter 6 and explained in Chapter 5.

4.4. Summary

This chapter has illustrated the concept and process of LCA phases, starting from goal and scope definition, inventory analysis and impact assessment based on ISO 14040. Moreover, it has provides an overview of LCA of concrete products, which gives the complete cycle in the production of concrete, although the methodology itself for this research will be explained in detail in Chapter 6. Additionally, it was decided to base monetisation of concrete waste and the mathematical relationship on the Environmental Prices Handbook (2017), which is applicable for ReCipe2016 LCIA.

CHAPTER 5: MODELLING METHODOLOGY FOR CONCRETE WASTE USING LCA

5.1. Introduction

This chapter demonstrates the modelling methodology that will be used in detail to find out the impact of concrete waste. The discussion begins with an explanation of the LCA methodology that will be used in this study including the ReCipe2016 impact assessment method selected to determine the emissions for three methods of concrete waste. Furthermore, the research adapted the methodology of ISO 14040. The discussion elaborates in detail the data collection for LCA and the process for life cycle impact assessment (LCIA) based on the research objectives. Moreover, the discussion continues with a clarification of selected environmental indicators and data collection and processing in each LCA phase.

5.2. Data Processing for LCA

A simulation method will be selected for this study, because it was found to be the most reliable and applicable for finding the LCA results for waste concrete and its damage cost to achieve the aim and objectives of this research. This is LCA requires the use of software that can simplify and organise the results, although hard data is required to analyse the impact of concrete waste on the environment. For example, the quantity of concrete waste in the city of Dubai is included. Moreover, the type of concrete chosen for this study is unreinforced concrete waste. The waste management option practised in the UAE and especially in the city of Dubai is to dispose of the concrete waste into landfill. There is a recycling method for concrete waste management in the UAE, but the method is not efficiently used.

The study has identified the quantity of unreinforced concrete waste in landfill in the city of Dubai to assess the environmental impact by using specific analysis tools as identified earlier in the study. In addition, LCA methodology and ReCipe2016 will be used to assess and calculate the ecological impact of the concrete waste. The impact of ReCipe2016 will help to assess the environmental impact of concrete waste management based on selected options, which are recycling method and final disposal method. Furthermore, the analysis results will contribute to the implementation of sustainable strategies that will help to reduce the quantity of concrete waste in the construction field and landfills, which will result in protecting the environment by reducing pollutants such as greenhouse gases and other impacts on ecosystem, resources and human health. It was clarified by Assamoi and Lawryshyn (2012) that LCA is a valuable tool to assess the management of solid waste performance. The LCA of this study will be conducted by using SimaPro software and EcoInvent database with ReCipe2016 environmental assessment, which are important tools to correlate the inventory and assess the ecological impact of concrete waste.

The research modelling framework as illustrated in Figure 5.1 has been created based on the research aim and objectives that help to find the LCA and damage cost results of waste concrete by landfilling and recycling methods. The methodology starts with phase one of the LCA, which is the definition of the goal and scope, including the data functional unit, which is 1 tonne for concrete waste, 1 tonne for recycling concrete waste, and 1 km per tonne for transportation method. The inventory requirements have been taken from the EcoInvent database.

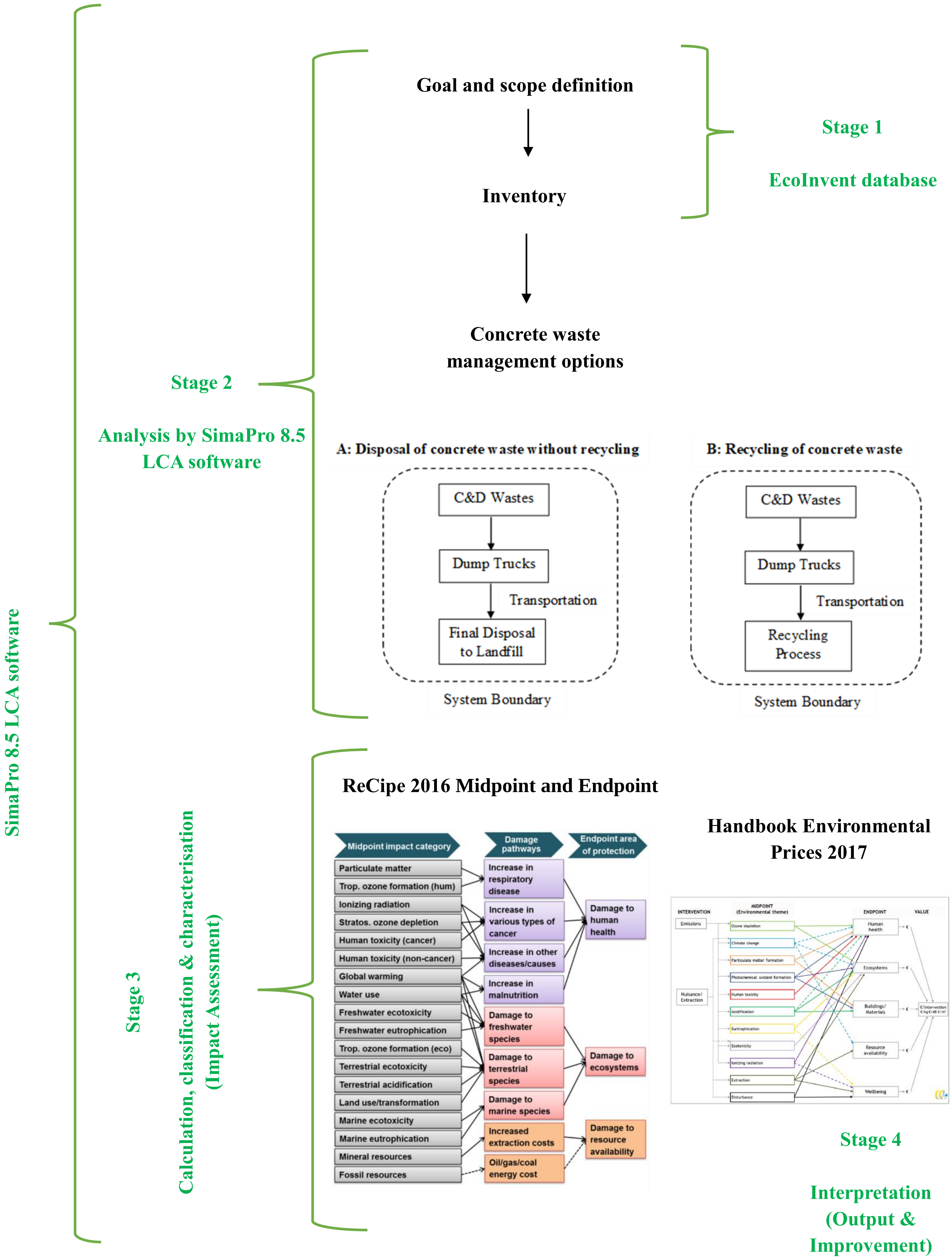
Based on the research modelling framework shown in Figure 5.1 the second phase is to integrate stage one by using SimaPro software 8.5.2 to help to analyse the impact of concrete waste based on the quantity of concrete waste in landfill. Moreover, another scenario of concrete waste management will be applied, which is the direct recycling option. For example, the amount of concrete waste in landfill will used as a recycling method for concrete waste, which will be calculated to compare the impact on the environment of both options in parallel with the method of transporting the concrete waste to landfill.

The research modelling framework in Figure 5.1 also shows the third phase of LCA, which is to assess the LCIA and find midpoint and endpoint results by using ReCipe2016 LCIA, which classifies and characterises the concrete waste impacts.

5.3. Research Modelling Framework

The research modelling framework shown in Figure 5.1 consists of four stages, as per ISO 14040 LCA framework and principles, which are parallel with the research aim and objectives, which are to find the ecological impact of landfilling and recycling methods of concrete waste and the damage cost that can impact on human health, ecosystem and resources.

Figure 5.1: Research modelling framework


SimaPro 8.5 LCA software
Goal and scope definition
Inventory
Stage 1
EcoInvent database
Concrete waste
management options
Stage 2
Analysis by SimaPro 8.5
LCA software
A: Disposal of concrete waste without recycling
C&D Wastes
Dump Trucks
Transportation
Final Disposal
to Landfill
System Boundary
B: Recycling of concrete waste
C&D Wastes
Dump Trucks
Transportation
Recycling
Process
System Boundary
Stage 3
Calculation, classification & characterisation
(Impact Assessment)
ReCipe 2016 Midpoint and Endpoint
Midpoint impact category
Damage pathways
Endpoint area of protection
Particulate matter
Trop. ozone formation (hum)
Ionizing radiation
Stratos. ozone depletion
Human toxicity (cancer)
Human toxicity (non-cancer)
Global warming
Water use
Freshwater ecotoxicity
Freshwater eutrophication
Trop. ozone formation (eco)
Terrestrial ecotoxicity
Terrestrial acidification
Land use/transformation
Marine ecotoxicity
Marine eutrophication
Mineral resources
Fossil resources
Increase in respiratory disease
Increase in various types of cancer
Increase in other diseases/causes
Increase in malnutrition
Damage to freshwater species
Damage to terrestrial species
Damage to marine species
Increased extraction costs
Oil/gas/coal energy cost
Damage to human health
Damage to ecosystems
Damage to resource availability
Handbook Environmental
Prices 2017
INTERVENTION
MIDPOINT
(Environmental theme)
ENDPOINT
VALUE
Emissions
Nuisance/
Extraction
Ozone depletion
Climate change
Particulate matter formation
Photochemical oxidant formation
Human toxicity
Acidification
Eutrophication
Ecotoxicity
Ionizing radiation
Extraction
Disturbance
Human health
Ecosystems
Buildings/
Materials
Resource
availability
Wellbeing
Stage 4
Interpretation
(Output &
Improvement)

5.4. Data Collection for LCA

The second and third phases of LCA, which are the goal and scope definition and inventory analysis, are adapted from the Swiss EcoInvent 3.2 database Life Cycle Inventory (LCI). Furthermore, in the third phase of LCA, which is the assessment stage, ReCipe2016 Life Cycle Impact Assessment (LCIA) with the methodology of midpoint and endpoint assessment is selected. Table 5.1 provides a comprehensive breakdown of the LCA methodology process as suggested by ISO 14040, which will be used in this study.

Table 5.1: Comprehensive LCA methodology by ISO 14040:2006

LCA stage	**Process**
Initial Phase	Creating system boundaries that define the problem and establish an inventory for main parameters.
Inventory Phase	A comprehensive explanation of raw materials and inputs of used energy at all points and emissions, output of overflow and solid waste. For instance, outputs are resource depletion such as material and energy pollutant emissions and discharges of chemical or physical load, for example, substances, heat, and noise.
Impact Assessment Phase	Linking the known inputs and outputs to the environmental impacts (Life Cycle Impact Assessment). It consist of the following components (first three components are mandatory, the others are optional): 1. Choice of impact categories, for example, category indicators and characterisation models. Impact categories are designated and selected with respect to LCA goal and scope. 2. LCI assignment results (Classification). The environmental loads are classified based on impact categories. Furthermore, some environmental loads have more than one impact category. 3. Calculation of category indicator results (Characterisation). The category indicators are demonstrated for different environmental loads that

	led to environmental impacts such as Global Warming Potential. 4. Data Quality Analysis, which recognises the reliability of indicator results. 5. Normalisation. Stating category indicators comparative to a standard, for example, tonne of CO2 or similar. 6. Grouping, which sorts and ranks possible influence categories. 7. Weighting, which states the (subjective) importance of the impact category which rarely the categories are sorted by damage or theme category. Note: points no. 5, 6 and 7 are optional as per ISO14040:2006
Interpretation (Improvement Phase)	Using data attained in analysis to enhance the overall environmental performance.

The research will adapt the first three main steps from the LCIA phase based on Figure 5.2. Moreover, ISO 14040 states that the rest of the LCIA steps, which are normalisation, grouping and weighting, are optional. In this study, normalisation, grouping and weighting will be ignored.

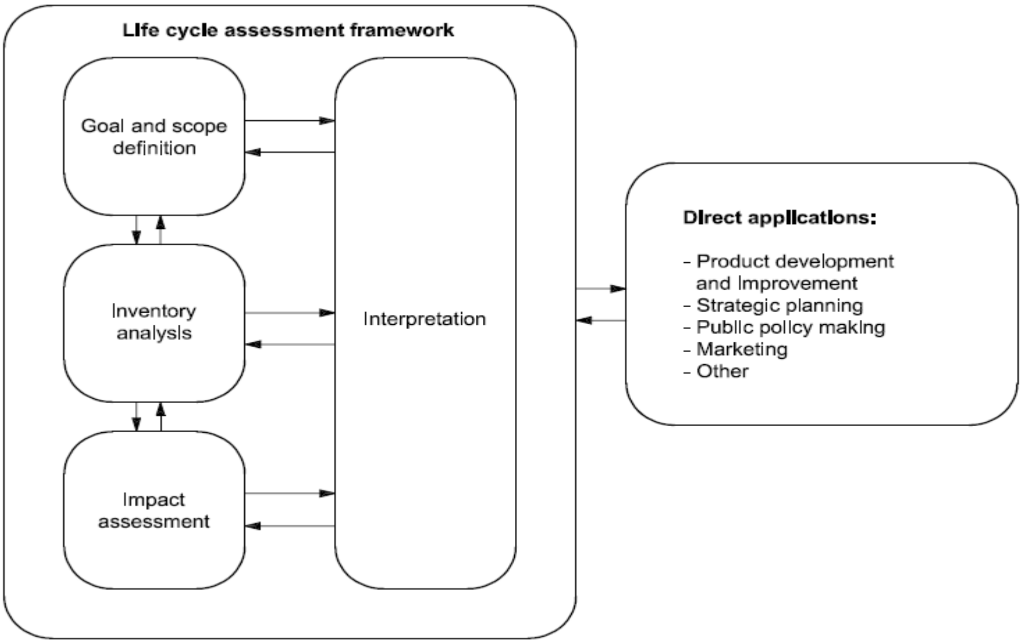


Figure 5.2: Life cycle assessment framework (International Organization for Standardization ISO 14040:2006, p. 14)

5.4.1. First LCA Phase: Life Cycle Goal and Scope Definition

Figure 5.3 shows the system boundaries of the LCA of concrete waste as a disposal option, which is transportation of concrete waste from C&D sites, concrete ready-mix plant and concrete block factories. The system boundary of the recycling option for concrete waste is recycling of unreinforced concrete waste at the landfill itself by using stationary recycling equipment.

The objective of the LCA in this research is to investigate the environmental impact of two waste management options for concrete waste. In addition, data functional unit is 1 tonne for concrete waste, 1 tonne for recycling concrete waste, and 1 km per tonne for transportation method and inventory requirements, by EcoInvent database.

Moreover, the whole research inventory will consider using the Swiss EcoInvent inventory database, although Figure 5.3 illustrates the waste management options made within the system boundaries for this study as recommended by ISO 14040 as at the first phase.

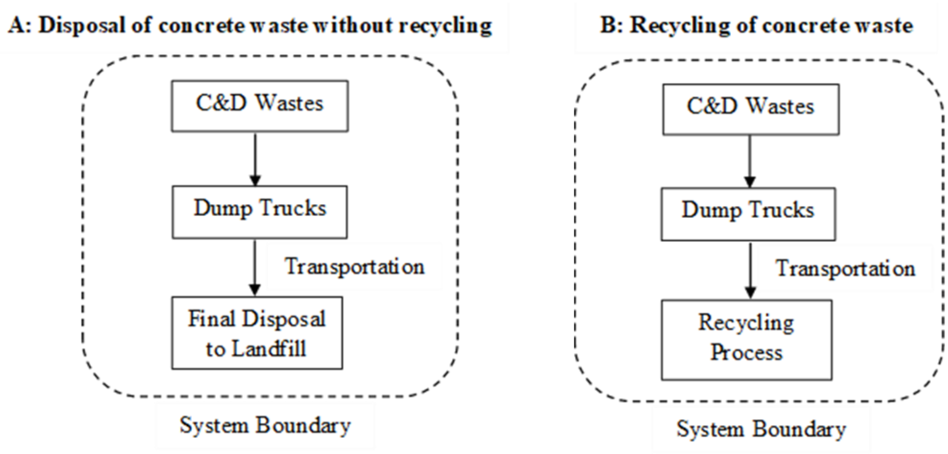


Figure 5.3: System boundaries of the research

Based on the above system boundaries and research aim and objectives, the focus of this research is only on emissions from two waste management options with the transportation process impact, which are the impact of disposal of unreinforced concrete waste to landfill and the impact of recycling unreinforced concrete waste as the scenarios proposed in this study. The inventory data for both management options is taken from the EcoInvent database, which includes equipment used, fuel, generation of electricity and transportation process. In addition, concrete waste sources are from C&D sites, ready-mix plants, those producing fresh concrete, and concrete block factories. On the other hand, there are some outputs and inputs are ignored in this research, which is cost. The reason for not including cost is because it is beyond the research scope.

5.4.2. Second LCA Phase: Life Cycle Inventory (LCI)

Life cycle inventory plays an important role in the LCA second phase. Furthermore, to complete the LCA assessment, the Swiss EcoInvent inventory database is selected for this research because the methodology used for developing the database includes input and output features suitable for the condition of global countries such as the UAE, although EcoInvent is a Swiss centre for Life Cycle Inventories (LCI). Essentially, to develop the database for the inventory process, the inventory will contain details of inputs and outputs related to system boundaries, as illustrated in Figure 5.3, while the inventory model of concrete waste management options are shown in Table 5.2.

Table 5.2: Waste management options in the city of Dubai and its waste quantity (Public Source, 2019)

Type of waste management	Description	Waste Quantity in 2018
Direct disposal to landfill	Concrete waste is collected from the sites in waste skips and transported to landfill	Normal Concrete = 5,400,000 t
Direct recycling	Concrete waste is collected from the sites in waste skips and transported to recycling plant	0

The inventory phase is one of the important stages in LCA. To begin with the inventory process, the EcoInvent 3.2 database was selected as the inventory for all processes. Moreover, as per ISO14040:2006, the inventory is the second phase in LCA. Firstly, the inventory stage processes in this research are transportation of concrete waste to the landfill by lorry, quantity of concrete waste in the landfill, and quantity of concrete waste if it is recycled.

The first inventory input is transportation method, which is given as (*Transport, freight, lorry 16-32 metric ton, EURO3 {GLO}| market for | APOS, S*). In addition, the selected inventory is applicable for the city of Dubai since it is a global inventory and it is used in the market. Although, the emissions from this inventory are provided, such as input from nature and outputs such as emissions to air, emissions to water and emissions to soil. In addition, all types of emissions will be identified in the results analysis in Chapter 6. Furthermore, the transportation inventory is selected to be 40 km as a distance, which is based on concrete waste sources from C&D sites, ready-mix plants, concrete block factories and concrete testing laboratories in the city of Dubai to Al Bayada C&D Landfill. Moreover, different areas are identified in Figure 5.4, which shows the sources of generating concrete waste and distance to transport the waste to Al Bayada C&D Landfill.

Figure 5.4: Sources of generating concrete waste and its transporting distance to Al Bayada C&D Landfill (Google Maps, 2019)

The second inventory output is (*Waste concrete, not reinforced {RoW}| treatment of waste concrete, not reinforced, collection for final disposal | APOS, S*). In addition, this method includes the environmental impact of unreinforced concrete waste sent to landfill. Although, it is applicable for the Rest of World (ROW). In addition, the emissions from this inventory are provided, such as input from nature and outputs such as emissions to air, emissions to water and emissions to soil. In addition, all types of emissions will be identified in the results analysis in Chapter 6.

The third inventory output is (*Waste concrete, not reinforced {RoW}| treatment of waste concrete, not reinforced, recycling | APOS, S*). In addition, this method includes the environmental impact of recycling unreinforced concrete waste, which is applicable for the Rest of World (ROW). Although, the emissions for this inventory are provided, such as input from nature and outputs such as emissions to air, emissions to water and emissions to soil. In addition, all types of emissions will be identified in the results analysis in Chapter 6. Moreover, it is assumed that mobile or stationary recycling equipment is used at the landfill itself.

All the above three inventories are global and used in the rest of the world and applicable for use in most countries such as the United Arab Emirates. By identifying the mandatory inventories to be calculated in the LCA, the third phase is to calculate the Life Cycle Impact Assessment (LCIA). For this research, the ReCiPe2016 impact assessment method was selected, because it is a method used worldwide, which has midpoint and endpoint assessment based on 18 midpoint impact categories, nine damage pathways and three endpoint areas of protection. Furthermore, all the above-mentioned inventories are selected to be Allocation at Point Of Substitution (APOS) and signified as unit processes. For instance, the type of unit process consists only of emissions and source inputs from one process, and references to input from other unit processes, while system process signifies the outcome inventory of the whole LCA with no perception of inputs and outputs of the particular supply chain treated in a system of production, as clarified by Goedkoop et al., 2016). Moreover, Table 5.3 shows the differences between unit process and system process in SimaPro software, which shows the reason behind selecting the unit process of Allocation at Point Of Substitution (APOS) for this study, which is defined in Table 5.4 (EcoInvent website 2018, p. 1).

Table 5.3: Differences between unit process and system process in SimaPro software (SimaPro introduction, p. 20)

Unit process	System process
Transparent and vast, in which the process tree permits the involvement of all single unit processes to be traced	Simple process tree
Includes information of uncertainty, which permits statistical analysis such as Monte Carlo to be conducted	No uncertainty calculation
Relatively slow calculation	Fast calculation

Converted EcoInvent 3.4 data as system and unit processes (results) with covers other processes, which data collected in April 2018. Moreover, uncertainty data is not included (EcoInvent website 2018, p. 1). Monte Carlo analysis is not applicable with the EcoInvent consequential database. Moreover, the EcoInvent v3 database has Life Cycle Inventory (LCI) data for numerous segments, for example, energy production, transport, building materials and production of chemicals, metal production (EcoInvent website 2018, p. 1). The total database contains more than 10,000 added datasets, each one of which defines a life cycle inventory based on process level. On the other hand, SimaPro delivers six libraries, which contain all the processes included in the EcoInvent database, which depend on different system models such as unit or system processes (EcoInvent website 2018, p. 1). However, three systems of EcoInvent models are included, as shown in Table 5.5, which are Allocation at Point of Substitution, Cut-off by Classification and Consequential (EcoInvent website 2018, p. 1).

Table 5.4: System Model EcoInvent models (EcoInvent website 2018, p. 1)

System Model			
Undefined	**Allocation, cut-off by classification**	**Allocation at the point of substitution**	**Substitution, consequential, long term**
Unlinked multi-product activity datasets related to the basis of all other system models. Datasets attained and calculated by data providers. These datasets' activities are important for exploring environmental impacts of a detailed activity (gate-to-gate), without concern for related impacts upstream or downstream.	System model splits into multi-product activities by allocation, which are related to physical properties, economic, mass or other properties. By-products of waste treatment processes are called cut-off, in which all by-products are categorised as recyclable. Markets in this model contain all activities in share with current production volume.	System model splits into multi-output activities by physical properties, economic, mass or other properties' allocation. By-products of treatment processes are measured as a part of the waste-producing system and allocated together. Markets in this model contain all activities in share with current production volume. This system model is called "Allocation, default" in EcoInvent versions 3.01 and 3.1.	System model uses substitution, which is called "system expansion", to substitute by-product outputs. It contains only activities to the extent that they are predictable to change in the long term as concern of changes in demand at a small scale, following a consequential approach, which calculates both constrained markets and technology constraints.

The system model called allocation, recycled content or cut-off is based on the method that main manufacture of materials is continuously allocated to the main user of a material. In addition, if a material is recycled, then the main producer will not benefit from providing any recyclable materials. Moreover, consequence denotes that recyclable materials are available burden-free to recycling processes and secondary recycled materials approve only the impacts of the recycling processes. However, waste producers will not benefit from the recycling or reuse of products, which causing out of any waste treatment. This approach was included in EoInvent v1 and v2, while SimaPro provides Allocation at Point of Substitution (APOS), which is included in EoInvent version 3.4 (EcoInvent website 2018, p. 1).

The system model of Allocation at Point of Substitution (APOS) covers two methodological choices. Firstly, it uses average product supply, as designated in market activity datasets. Secondly, it uses partitioning (allocation) to transform multi-product datasets to single-product datasets. In addition, flows are allocated with correlation of their exact value, which signifies corrected economic revenue for some marketplace limitations and variations (EcoInvent website 2018, p. 1).

The system model of Consequential has two different methodological choices. Firstly, it uses unconstrained supply of products, which correlates with market activity datasets and provides information on technology level. Secondly, it uses substitution (system expansion) to transform multi-product datasets to single-product datasets. Furthermore, the consequential model is a system model projected to present values of small-scale and long-term decisions by calculating limits that are included at scale and time horizon. Finally, the consequential model calculates long-term changes and rules for the technology level of unaffected suppliers based on market tendency (EcoInvent website 2018, p. 1).

The option of choosing consequential allocation gives specialists a substitute for the attributional approach. Both attributional (default) and consequential databases are included in

SimaPro by using unit and system processes. In addition, unit processes contain links to other unit processes, in which inventory flows can be calculated by SimaPro. In contrast, system processes contain provided calculated inventory flows and do not contain links to other processes. However, it is recommended to choose system processes to increase calculation speed when using processes from the EoInvent database as background processes. Moreover, unit processes are used for complete interpretation and uncertainty analysis by using Monte Carlo except for the consequential unit library. Furthermore, looped structure analysis of unit processes may take some time during analysis on a PC. However, both unit and system processes show the same results, although minor differences may infrequently occur due to rounding errors. Finally, it is applicable to switch between unit and system libraries when defining calculation setup (EcoInvent website 2018, p. 1).

5.4.3. Third LCA Phase: Life Cycle Impact Assessment (LCIA)

Determination of environmental impact indicators is critical. On the other hand, LCIA is used to assess a material based on LCI, which helps to recognise its environmental importance and provide data for the analysis phase (UNEP, 2003). The importance of the LCIA step is very significant because the recent and present global influence of the concrete waste will be used based on impact results. Based on the impact indicator selection, ISO14040 standards permit impact category indicators to be used between the inventory results, such as midpoint and endpoint emissions. Moreover, midpoint analysis reduces the estimation in the predicting and modelling effect assimilated with LCIA, which minimises the modelling complexity and simplifies the communication. Life Cycle Assessment (LCA) calculates the ecological impacts of whole product life cycle. Moreover, the life cycle of a product is associated with an enormous amount of emissions and extraction of related resources, which significantly differ in their ecological application. In addition, LCIA correlates the studies of

LCA interpretation by decoding these resource extractions and emissions numbers into limited scores related to environmental impact (Hauschild & Huijbregts, 2015). Furthermore, those scores concluded by characterisation factors, which specify the ecological impact per unit, for example, kilogram of related emission or resource. Moreover, there are two techniques for originating the characterisation factors, which are midpoint and endpoint methods. The midpoint method is positioned at the level of somewhere along the impact pathway approach, which is at the point where the environmental mechanism is equally allocated to that specific impact category (Goedkoop et al. 2009). On the other hand, characterisation factors of the endpoint level relate to three areas, which are human health, quality of ecosystem and lack of resources. In addition, the two methods are complementary; the midpoint method has a robust relation to ecological flows and comparatively less uncertainty, while the endpoint method provides superior information on ecological flows but is more uncertain than the midpoint method (Hauschild & Huijbregts, 2015). Furthermore, ReCiPe2016 was developed in 2008 by cooperation between RIVM, Radboud University Nijmegen, Leiden University and Pré Consultants. Goedkoop et al. (2009) developed the ReCipe2008 LCIA method, which delivers coherent factors of characterisation such as the midpoint and endpoint levels.

The LCIA of this research selected ReCipe2016 to estimate the ecological impacts of concrete waste in the city of Dubai for three methods, landfilling, recycling and transportation. The ReCiPe2016 impact assessment method was chosen because it is global and updated compared to other impact assessment methods. In addition, all the data and methodologies used in ReCiPe2016 are up to date and based on current scientific knowledge approaches. However, as there is some complexity in this assessment, SimaPro 8.5.2 LCA software is used as a tool to correlate between the methodology of LCIA and EcoInvent LCI database. Clauses 5.5 and 5.5 show the processes of calculations for the midpoint and endpoint of ReCipe2016 LCIA inside the SimaPro software by using the EcoInvent database, which is selected in this study based on the research aim and objectives.

5.5. LCIA by ReCipe2016

Table 5.5: Method of characterisation process for ReCiPe2016 (RIVM Report 2016-0104, p. 16-18)

Effect	Description
Climate change	- Egalitarian time horizon was taken as 1,000 years, which is the longest time horizon reported in the literature for CO2 functions. - A considerably greater set of greenhouse gas emissions (207 GHGs in total) is involved on the basis of the newest IPCC report. - Damage factors for human health and terrestrial ecosystems are included. - The damage to freshwater (river) ecosystems is included.
Stratospheric ozone depletion	- New semi-empirical ODPs were included with complete specification among numerous chlorofluorocarbons (CFCs). - A preliminary ODP for N2O is included. - Three time horizons were steadily instigated: 20 years (Individualist), 100 years (Hierarchist) and infinite (Egalitarian).
Ionising radiation	- Three time horizons were steadily implemented: 20 years (Individualist), 100 years (Hierarchist) and 100,000 years (Egalitarian). - Dose and dose rate effectiveness factors (DDREFs) were identified per cultural perspective. - Updated DALYs per fatal cancer occurrence are applied.
Fine particulate matter formation	- The European factor was replaced by a world average factor. - Lung cancer and cardiovascular mortality were included as serious influence. - Value choices is included. - World-region specific characterisation factors are provided.
Photochemical ozone formation	- The European factor was replaced by a world average factor. - Respiratory mortality is provided. - Characterisation factors for individual VOCs, the most recent photochemical ozone formation potentials (POCPs) reported in the literature, were included. - Damage to terrestrial ecosystems is included - World-region specific characterisation factors are provided.
Terrestrial acidification	- The European factor was replaced by a world average factor, based on grid-specific factors. - Soil sensitivity is based on H+ concentration instead of base saturation. - Effects of all vascular plant species are provided. - Country-specific characterisation factors are included.

Freshwater eutrophication	- Fate factors are derived with a state-of-the-art global fate model for phosphorus, instead of a European fate model. - Effect factors are provided based on a global analysis, instead of using information from the Netherlands only. - Country-specific characterisation factors are included.
Marine eutrophication	- Fate factors were derived with a state-of-the-art global fate model for nitrogen, instead of a European fate model. - Endpoint characterisation factors are included by defining influence and damage factors based on a global analysis. - Continent-specific characterisation factors are included.
Toxicity	- Characterisation factors for human cancer and non-cancer influence are separately included. - Fate and exposure for dissociating organics is modelled. - The USEtox organic and inorganic database is implemented (3094 substances). - A time horizon of 20 years is provided for the Individualist perspective. - Only linear effect factors are provided for reasons of simplicity. - Effects on agricultural and urban soil are excluded to avoid double calculating with the land use impact category.
Water use	- Consumption/extraction ratios are included. - Characterisation factors at an endpoint level for human health, terrestrial and aquatic ecosystems are provided. - Country-specific characterisation factors are included.
Land use	- Characterisation factors are based on worldwide scale data. - The local impact of land use is included.
Mineral resource scarcity	- Cumulative grade-tonnage relationships and cumulative cost-tonnage relationships are used based on mine-specific cost and production data. - Estimation of future production is included in the modelling without future discounting.
Fossil resource scarcity	- Cumulative grade-tonnage relationships and cumulative cost-tonnage relationships are used based on mine-specific cost and production data. - Estimation of future production is included in the modelling without future discounting.

5.5.1. Impact Pathways and Areas of Protection

There are three areas of protection, called the endpoint, are human health, ecosystem quality and resource scarcity, as elaborated on in Table 5.6. Moreover, human health is relevant to DALYs (disability adjusted life years), which identifies the lost years or when a person is incapacitated due to accident or disease. Ecosystem quality is relevant to local species loss integrated over time (species year), and resource scarcity represents the extra costs in dollars

($), which relates to future mineral and fossil resource extraction (RIVM Report 2016-0104, p.18).

Table 5.6: Endpoint categories, indicators and characterisation factors (RIVM Report 2016-0104, p. 19)

Area of protection	**Endpoint**	**Abbr**	**Name**	**Unit**
Human health	Damage to human health	HH	Disability adjusted loss of life years	Year
Natural environment	Damage to ecosystem quality	ED	Time integrated species loss	Species * yr
Resource scarcity	Damage to resource availability	RA	Surplus cost	Dollar

Figure 5.5 shows the impact categories included in the ReCiPe2016 methodology. It consists of 18 midpoints impacts category, nine damages pathways and three endpoint areas of protection (RIVM Report, 2016-0104).

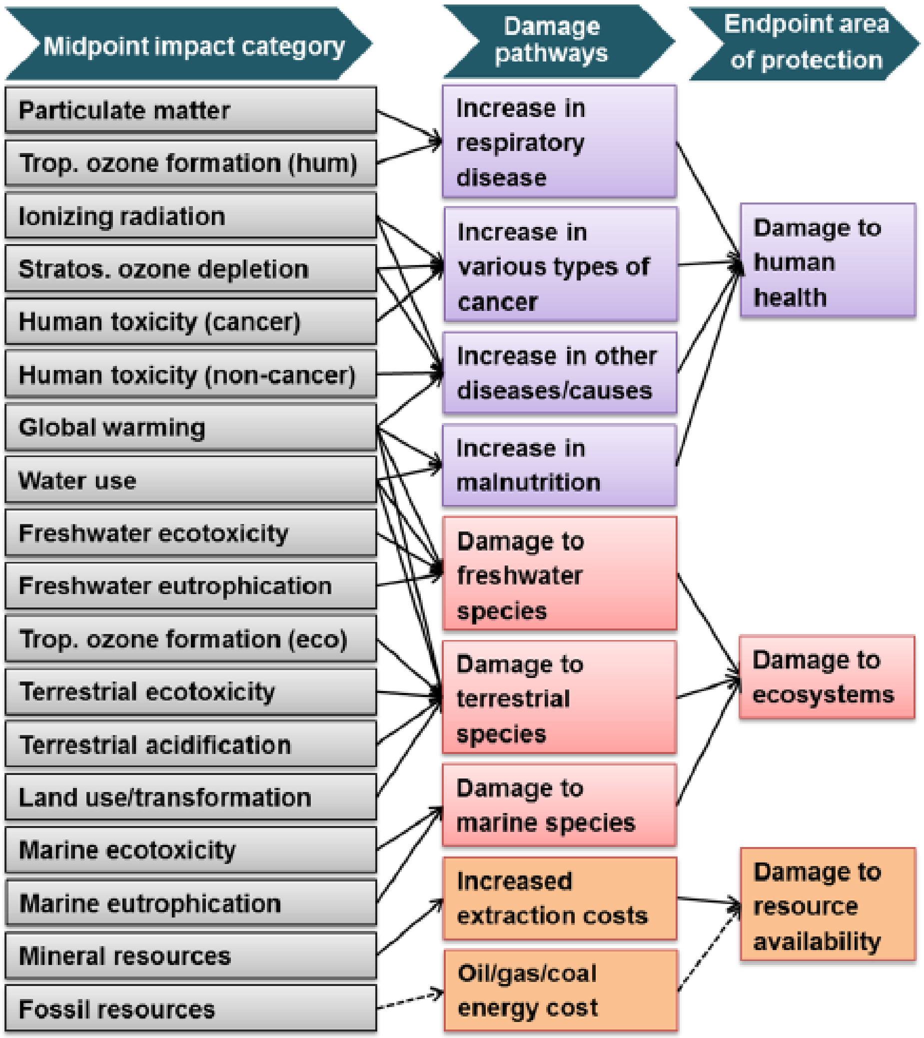


Figure 5.5: ReCiPe2016 impact assessment methodology (RIVM Report 2016-0104, p. 19)

5.5.2. Value Choices

Different choices and different sources of uncertainty are grouped into a certain number of perspectives or scenarios, which are based on "Cultural Theory" (Thompson et al. 1990). Moreover, the perspectives do not represent archetypes of human behaviour; they are simply used to group the same types of assumptions and choices (RIVM Report, 2016-0104).

There are three perspectives included in ReCiPe2016, which are individualistic perspective, hierarchist perspective and egalitarian perspective. Firstly, the individualistic perspective, which has a short time frame in which influence types are undisputed and technological optimism with concern to human adaptation. Secondly, the hierarchist perspective, which has a medium time frame and is based on scientific consensus with concern for the time frame and plausibility of influence mechanisms. Thirdly, the egalitarian perspective, which has a long time frame and it is the most preventative perspective in which all influential data pathways are available (RIVM Report 2016-0104, p. 20). Table 5.7 shows the value choices in the derivation of characterisation factors as provided in ReCiPe2016.

Table 5.7: Value choices in the derivation of characterisation factors as per ReCiPe2016 (RIVM Report 2016-0104, p. 20-22)

	Individualist	**Hierarchist**	**Egalitarian**
Climate change			
Time horizon[1]	20 years	100 years	1,000 years
Climate-carbon feedbacks non-CO2	No	Yes	No
Future socio-economic developments[2]	Optimistic	Baseline	Pessimistic
Adaptation potential[2]	Adaptive	Controlling	Comprehensive
Ozone depletion			
Time horizon[1]	20 years	100 years	Infinite
Included effects[2]	Skin cancer	Skin cancer	Skin cancer and cataract
Ionising radiation			
Time horizon[1]	20 years	100 years	100,000 years
Dose and dose rate effectiveness factor (DDREF)[2]	10	6	2

Included effects[2]	-Thyroid, bone marrow, lung and breast cancer -Hereditary disease	-Thyroid, bone marrow, lung, breast, bladder, colon, ovary, skin, liver, oesophagus and stomach cancer -Hereditary disease	-Thyroid, bone marrow, lung, breast, bladder, colon, ovary, skin, liver, oesophagus, stomach, bone surface and remaining cancer -Hereditary disease
Fine particulate matter formation			
Included effects[2]	Primary aerosols	Primary aerosols, secondary aerosols from SO2, NH3 and NOx	Primary aerosols, secondary aerosols from SO2, NH3 and NOx
Toxicity			
Time horizon[1]	20 years	100 years	Infinite
Exposure routes for human toxicity[1]	Organics: all exposure routes. Metals: drinking water and air only	All exposure routes for all chemicals	All exposure routes for all chemicals
Environmental compartments for marine ecotoxicity[1]	Sea + ocean for organics and non-essential metals. For essential metals, the sea compartment is included only, excluding the oceanic compartments.	Sea + ocean for all chemicals	Sea + ocean for all chemicals
Carcinogenity[1]	Only chemicals with carcinogenicity classified as 1, 2A, 2B by IARC	All chemicals with reported carcinogenic effects	All chemicals with reported carcinogenic effects
Minimum number of tested species for ecotoxicity[1]	4	1	1
Water use			
Regulation of stream flow[2]	High	Standard	Standard
Water requirement for food production[2]	1000 m3/yr/capita	1350 m3/yr/capita	1350 m3/yr/capita

Impacts on terrestrial ecosystems considered[2]	No	Yes	Yes
Mineral resource scarcity			
Future production	Reserves	Ultimate recoverable resource	Ultimate recoverable resource

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5.5.3. Characterisation Factors of the Midpoint Level

Each category has some differences in the unit and the unit of midpoint characterisation factor (CFm), as elaborated in Table 5.8. This is due to the introduction of a reference substance for the characterisation factor, which is a dimensionless number that states the strength of the substance amount related to the reference substance. The unit kg is the reference substance for one specific environmental compartment emission, which is based on impact categories and resource scarcity. Concerning land use, it is the area and time integrated for one kind of land use (RIVM Report, 2016-0104)

Table 5.8: Indicators and categories of midpoint level (RIVM Report 2016-0104, p. 23-24)

Impact category	Indicator	Unit	CFm	Abbr.	Unit
Climate Change	Infra-red radiative forcing increase	W*yr/m2	Global Warming Potential	GWP	kg CO2 to air
Ozone Depletion	stratospheric ozone decrease	ppt*yr	Ozone Depletion Potential	ODP	kg CFC-11 to air

Ionising Radiation	absorbed dose increase	man*Sv	Ionising Radiation Potential	IRP	kBq Co-60 to air
Fine Particulate Matter Formation	PM2.5 population intake increase	kg	Particulate Matter Formation Potential	PMFP	kg PM2.5 to air
Photochemical Oxidant Formation: Ecosystem Quality	tropospheric ozone increase (AOT40)	ppb.yr	Photochemical Oxidant Formation Potential: Ecosystems	EOFP	kg NOx to air
Photochemical Oxidant Formation: Human Health	tropospheric ozone population intake increase (M6M)	kg	Photochemical Oxidant Formation Potential: Humans	HOFP	kg NOx to air
Terrestrial Acidification	proton increase in natural soils	$yr*m^2*mol/l$	Terrestrial Acidification Potential	TAP	kg SO2 to air
Freshwater Eutrophication	phosphorus increase in freshwater	yr*m3	Freshwater Eutrophication Potential	FEP	kg P to fresh water
Marine Eutrophicati On	dissolved inorganic nitrogen increase in marine water	yr.kgO2/kgN	Marine Eutrophicati On Potential	MEP	Kg N to marine water
Human Toxicity: Cancer	risk increase of cancer disease incidence	-	Human Toxicity Potential	HTPc	kg 1,4-DCB to urban air
Human Toxicity: Non-Cancer	risk increase of non-cancer disease incidence	-	Human Toxicity Potential	HTPnc	kg 1,4-DCB to urban air
Terrestrial Ecotoxicity	Hazard weighted increase in natural soils	$yr*m^2$	Terrestrial Ecotoxicity Potential	TETP	kg 1,4-DCB to industrial soil
Freshwater Ecotoxicity	Hazard weighted increase in	$yr*m^3$	Freshwater Ecotoxicity Potential	FETP	kg 1,4-DCB to fresh

	freshwater				water
Marine Ecotoxicity	Hazard weighted increase in marine water	yr*m^3	Marine Ecotoxicity Potential	METP	kg 1,4-DCB to marine water
Land Use	occupation and timeintegrated transformation	yr*m^2	Agricultural Land Occupation Potential	LOP	m^2*yr annual crop land
Water Use	increase of water consumed	m^3	Water Consumption Potential	WCP	m^3 water consumed
Mineral Resource Scarcity	ore grade decrease	kg	Surplus Ore Potential	SOP	kg Cu
Fossil Resource Scarcity	upper heating value	MJ	Fossil Fuel Potential	FFP	kg oil

5.5.4. From Midpoint to Endpoint

To calculate the results from midpoint to endpoint, the characterisation factors (CFe) of the endpoint are derived from CFm, which means that the constant midpoint to endpoint factor per impact category equation is (RIVM Report, 2016-0104):

$$CFe_{x,c,a} = CFm_{x,c} \times F_{M\rightarrow,E,c,a}$$ Equation 5-1

Where, c signifies the cultural perspective, a signifies the area of protection (human health, terrestrial ecosystems, freshwater ecosystems, marine ecosystems or resource scarcity), and x signifies the stressor of concern (RIVM Report, 2016-0104).

$$F_{M\rightarrow,E,c,a}$$ Equation 5-2

The conversion of the midpoint to endpoint factor for cultural perspective c and area of protection a.

The midpoint to endpoint factors are constant per category, since the environmental mechanisms are measured to be indistinguishable for all cause and effect pathways after the stressors of midpoint and endpoint impact location (RIVM Report, 2016-0104).

Table 5.9 elaborates the midpoint-to-endpoint factors for human health damage, terrestrial ecosystem damage, freshwater ecosystem damage, marine ecosystem damage, and resource scarcity, which consist of three cultural perspectives: Individualist (I), Hierarchist (H) and Egalitarian (E). In addition, all impact categories have constant global midpoint to endpoint factors, excluding fossil resource scarcity, because there is less data about the full cause and effect pathway (RIVM Report, 2016-0104). Details about the derivation of these factors are included later in this chapter.

Table 5.9: Midpoint-to-endpoint factors (RIVM Report 2016-0104, p. 25)

Impacts Category	**Unit**[1,2]	**I**	**H**	**E**
Human Health				
Climate Change	yr/kg CO2 to air	8.1E-08	9.3E-07	1.3E-05
Ozone Depletion	yr/kg CFC11 to air	2.4E-04	5.3E-04	1.3E-03
Ionising Radiation	yr//kBq Co-60 to air	6.8E-09	8.5E-09	1.4E-08
Fine Particulate Matter Formation	yr/kg PM2.5 to air	6.3E-04	6.3E-04	6.3E-04
Photochemical Ozone Formation	yr/kg NOx to air	9.1E-07	9.1E-07	9.1E-07
Cancer Toxicity	yr/kg 1,4-DCB to air	3.3E-06	3.3E-06	3.3E-06

Non-Cancer Toxicity	yr/kg 1,4-DCB to air	6.7E-09	6.7E-09	6.7E-09
Water Use	yr/m^3 water	3.1E-06	2.2E-06	2.2E-06
Ecosystem Quality: Terrestrial				
Climate Change	species.yr/kg CO_2 to air	5.3E-10	2.8E-09	2.5E-08
Photochemical Ozone Formation	species.yr/kg NO_x to air	1.3E-07	1.3E-07	1.3E-07
Acidification	species.yr/kg SO_2 to air	2.1E-07	2.1E-07	2.1E-07
Toxicity	species.yr/kg 1,4-DCB to industrial soil	5.4E-08	5.4E-08	5.4E-08
Water Use	species.yr/m^3 water consumed	0	1.4E-08	1.4E-08
Land Use	species/m^2 annual crop land	8.9E-09	8.9E-09	8.9E-09
Ecosystem Quality: Freshwater				
Climate Change	species.yr/kg CO_2	1.5E-14	7.7E-14	6.8E-13
Eutrophication	species.yr/kg P to freshwater	6.1E-07	6.1E-07	6.1E-07
Toxicity	species.yr/kg 1,4-	7.0E-10	7.0E-10	7.0E-10
Water Use	DCB to freshwater species.yr/m^3 water consumed	6.0E-13	6.0E-13	6.0E-13
Ecosystem Quality: Marine				
Toxicity	species.yr/kg 1,4-DCB	1.1E-10	1.1E-10	1.1E-10
Eutrophication	species.yr/kg N to marine water	1.7E-09	1.7E-09	1.7E-09
Resource Scarcity				

Minerals	US2013 $/kg Cu	0.16	0.23	0.23
Fossils[3]	US2013 $/kg crude oil	0.46	0.46	0.46
	US2013 $/kg hard coal	0.03	0.03	0.03
	US2013 $/Nm3 natural gas	0.30	0.30	0.30

1 Denotes the unit for damage to human health, which signifies disability adjusted life years (DALY) lost in the human population.

2 Denotes the units for damage to ecosystems, which represents the number of species lost integrated over time.

3 Denotes fossil resource scarcity, which is the only midpoint category and excludes the constant of midpoint to endpoint factor.

5.6. ReCipe2016 Midpoint and Endpoint Impacts

LCIA by ReCipe 2016 has 18 midpoint impacts and three endpoint impacts assessments, which are going to be explained in detail with related equations and calculations. Moreover, the ReCipe2016 method was selected for this research because it is a global method and has a more updated and recent LCIA compared with the other methods.

5.6.1. Climate Change

The damage modelling of the impact category of climate change is segmented into several steps. Figure 5.6 elaborates that the emission of greenhouse gases in kg results in increased atmospheric concentration of these greenhouse gases (ppb), which also leads to increased radiative forcing capacity (w/m^2), and results in increasing the global mean temperature (°C). The effect of increasing the temperature ultimately leads to damage to human health and ecosystems. Furthermore, the damage estimations for human health, terrestrial ecosystems and freshwater ecosystems are included (RIVM Report, 2016-0104). Figure 5.6 elaborates the process of the cause-and-effect chain starting from emissions of greenhouse gases (GHG) to the damage to human health and the loss of species in terrestrial and freshwater ecosystems (RIVM Report, 2016-0104).

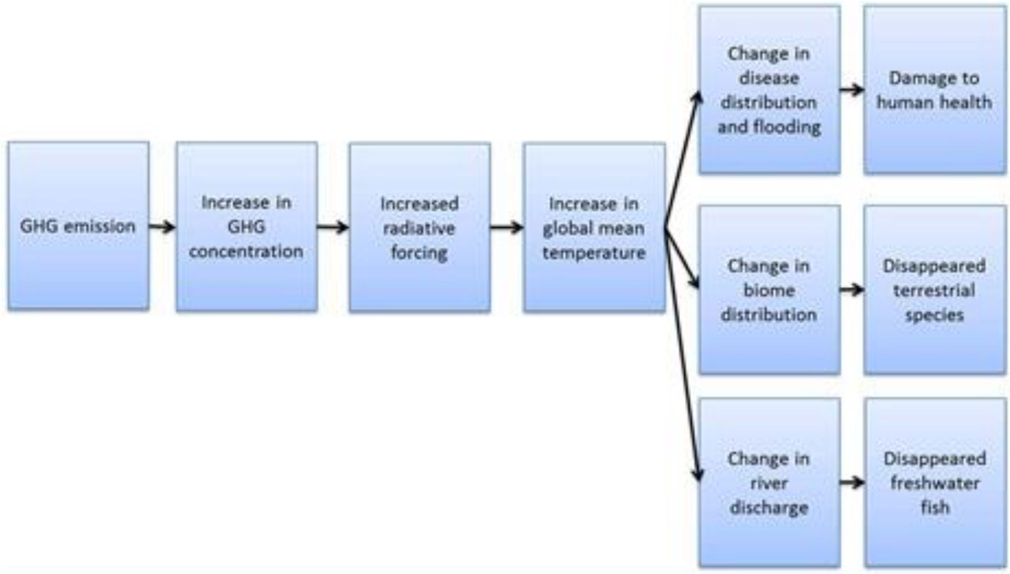


Figure 5.6: Process of cause-and-effect chain of climate change impact (RIVM Report 2016-0104, p. 27)

5.6.1.1. Value Choices

Joos et al. (2013) indicate that the influence of value choice on time horizon can impact on both midpoint and endpoint modelling of climate change. In addition, nemours GHGs result in diverse atmospheric lifetimes with relation to time horizon-dependent characterisation factors. On the other hand, the feedbacks of climate carbon for non-CO2 GHGs influence the important relation of CO2 decision whether to included or not. Furthermore, climate-carbon feedbacks are always included. Moreover, the feedback mechanism increases uncertainty, but it delivers a more reliable midpoint CF, although some other value choice measures are applicable only for damage assessment by including adaptation potential and future socio-economic improvement of human society. Finally, the categorisation of value choices is based on three cultural perspectives. Table 5.10 shows the value choices in the modelling of the influence of GHGs (De Schryver et al. 2009).

Table 5.10: Value choices in the modelling of the influence of GHGs (RIVM Report 2016-0104, p. 28)

Choice category	Individualist	Hierarchist	Egalitarian
Time horizon	20 years	100 years	1,000 years
Climate-carbon feedbacks included for non-CO2 GHGs	No	Yes	No[1]
Future socio-economic developments	Optimistic	Optimistic	Pessimistic
Adaptation potential	Adaptive	Controlling	Comprehensive

[1]It is preferred to include Climate Carbon feedbacks in the perspective, since the feedbacks of GWPs Climate Carbon are not included for the 1000 years' time horizon.

5.6.1.2. Characterisation Factors at Midpoint Level

Global Warming Potential (GWP) is the midpoint characterisation factor for climate change. In addition, GWP states the total of additional radiative forcing integrated over time, for example, 20, 100 and 1,000 years, calculated by emission of 1 kg of GHG; it is called Absolute Global Warming Potential (AGWP) and the unit is expressed as W m^{-2} yr kg^{1}. The calculation of midpoint characterisation factors such as GHG (x) and time horizon (TH) is expressed as follows:

$$GWP_{x,TH} = \frac{AGWP_{x,TH,}}{AGWP_{CO_2,TH}} \qquad \text{Equation 5-3}$$

Obviously, shows that yield a time-horizon-specific GWP with unit kg CO2 eq/kg GHG. In addition, GWPs for 20 and 100 years are included in the latest IPCC report (IPCC, 2013). With regard to the values, it is reported that those < 1 are rounded up to 0 or 1, which is linked to the stated AGWP of the substance and CO2.

Moreover, the AGWP for CO2 is a 1,000-year time horizon (=5.48·10−13 yr·W·m−2·kg-1), as stated by Joos et al. (2013). Moreover, the equation calculation of AGWP for a 1,000-year time horizon is linked to other GHGs (RIVM Report, 2016-0104), which is:

$$AGWP_{x,TH} = RF_x \, cv_x \, LT_x \, (1 - e^{-\frac{TH}{LT_x}}) \qquad \text{Equation 5-4}$$

For more clarification, RF is the stand of radiative efficiency (W m-2/ppb), and cv is the substance-specific mass to concentration conversion factor (ppb/kg); LT is the lifetime in years of substance x; TH is the stand of time horizon in years of assessment, as 1,000 years. In addition, RF and LT are included in the fifth IPCC assessment report (IPCC 2013), although the values of cv are not provided independently in this report, so they are calculated based on AGWPs reported by IPCC (2013), as shown in detail in Appendix 8.

5.6.1.3. From Midpoint to Endpoint

The calculation of endpoint characterisation factors (CFe) for Climate Change (CC) for GHG (x) is (RIVM Report, 2016-0104):

$$CFe_{x,c,a} = GWP_{x,c} \times F_{M\rightarrow,E,CC,c,a} \qquad \text{Equation 5-5}$$

Where, c refers to cultural perspective, a refers to area of protection (human health, terrestrial ecosystems or freshwater ecosystems), and $GWP_{x,c}$ is the midpoint characterisation factor (RIVM Report, 2016-0104).

$$CFe_{x,c,a} = GWP_{x,c} \times F_{M\rightarrow,E,CC,c,a} \qquad \text{Equation 5-6}$$

Midpoint to endpoint conversion factor for cultural perspective c and area of protection a (RIVM Report, 2016-0104).

$$F_{M\rightarrow,E,CC,c,a}$$ Equation 5-7

The calculation factor of midpoint to endpoint for human health is (RIVM Report, 2016-0104):

$$F_{M\rightarrow,E,CC,c,HH} = AGWP_{CO_2,c} \times TF_{CO_2,c} \sum_r \sum_h \Delta RR_{r,h,c} \times DALY_{r,h} = IAGTP_{CO_2,c} \times \sum_r \sum_h \Delta RR_{r,h,c} \times DALY_{r,h}$$ Equation 5-8

The calculation of midpoint to endpoint factor for terrestrial ecosystems is (RIVM Report, 2016-0104):

$$F_{M\rightarrow,E,CC,c,TERR} = IAGTP_{CO_2,c} \times A_{terr} \times EF_{terr} \times SD_{terr}$$ Equation 5-9

The calculation of midpoint to endpoint factor for freshwater ecosystems is (RIVM Report, 2016-0104):

Equation 5-10

$$F_{M\rightarrow,E,CC,c,fw} = IAGTP_{CO_2,c} \times SD_{fw} \sum_i EF_{fw,i} \times V_i$$

Table 5.11: Different areas of protection and cultural perspectives for midpoint to endpoint characterisation factors (RIVM Report 2016-0104, p. 37)

Area of protection	**Unit**	**Individualist**	**Hierarchist**	**Egalitarian**
Human Health	DALY/kg CO2eq	8.12·10-8	9.28·10-7	1.25·10-5
Terrestrial Ecosystems	Species.year/kg CO2eq	5.32·10-10	2.80·10-9	2.50·10-8
Aquatic Ecosystems	Species.year/kg CO2eq	1.45·10-14	7.65·10-14	6.82·10-13

5.6.2. Stratospheric Ozone Depletion

The emissions of ozone depleting substances (ODSs) which cause damage to human health are presented in steps in the cause-and-effect chain in Figure 5.7. Furthermore, ODSs lead to increases in UVB-radiation, which results in damage to human health, as obviously shown in Figure 5.7. In addition, the depleted chemicals from ozone are presented and do not have bromine groups in their molecules, which react with ozone particularly in the stratosphere. Furthermore, the concentrations of tropospheric in all ODSs increase after a certain period in parallel with the concentration of ODS. When ozone increases it leads to a decrease in atmospheric ozone concentration, which influence in larger radiation portion of UVB that hit the earth. More radiation has a negative impact on human health, which leads to an increase in the occurrence of skin cancer and cataracts (RIVM Report, 2016-0104).

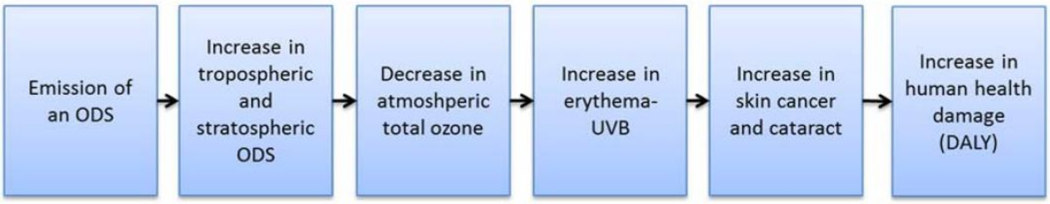


Figure 5.7: Process of cause-and-effect chain of stratospheric ozone depletion impact (RIVM Report 2016-0104, p. 39)

5.6.2.1. Value Choices

The results of uncertainty for environmental pressure and the choice of the time horizon are elaborated clearly, as stated by Schryver et al. (2011), who suggested using different cultural perspectives in the updating of the characterisation factors. Furthermore, in Table 5.12, as described by Struijs et al. (2009), the correlation is slightly uncertain between UVB and the development of cataracts, in which cataracts are included only in the Egalitarian perspective (RIVM Report, 2016-0104).

Table 5.12: Effect of ozone depleting substances by value choices in the modelling (RIVM Report 2016-0104, p. 40)

Choice category	**Individualist**	**Hierarchist**	**Egalitarian**
Time horizon	20 yr	100 yr	Infinite
Included effects	Skin cancer	Skin cancer	Skin cancer Cataract

5.6.2.2. Characterisation Factors at Midpoint Level

The expression of Ozone Depleting Potential (ODP) is unit of kg CFC-11 equivalents, which is used at the midpoint level as a characterisation factor. Although, ODP measures the quantity of ozone as a substance that can reduce in relationship with CFC-11 as a specific time horizon. In addition, mostly linked with ODS molecular structure and specifically in number of chlorine and in the molecule of bromine groups and with atmospheric lifetime of the chemical. The World Meteorological Organisation calculates the ODPs and the latest update was published in 2010 (WMO, 2011). The change in EESC and the ODP calculation is (RIVM Report, 2016-0104):

$$ODP_{inf,x} = \frac{\Delta EESC_x}{\Delta EESC_{CFC-11}}$$ Equation 5-11

$ODP_{inf,x}$ is the ODP for an infinite time horizon for ODS *x*, $\Delta EESC_x$ while $\Delta EESC_{cfc-11}$ are the changes in EESC caused by the emission of 1 kg of ODS *x* and 1 kg of CFC-11, respectively. Moreover, the reader in WMO report refers the procedure and exact modelling of EESC, which includes atmospheric models. In addition, the WMO only provides ODPs for the infinite time horizon, while calculations of all ODSs compared to CFC-11 are included here in order to provide ODPs for different time horizons. On the other hand, calculation of fraction of damage at any time horizon is stated by De Schryver et al.’s (2011) equation:

$$F_t = 1 - e^{(-t-3)\cdot k}$$ Equation 5-12

Ft is the fraction of the total damage caused by an ODS during the first t years, while k is the removal rate of ODS (yr-1), which corresponds to its opposite atmospheric lifetime (WMO). The below formula specifies in years the time lag between emissions to the troposphere and transportation to the stratosphere. The calculation of ODP for another time horizon is (RIVM Report, 2016-0104):

$$ODP_{t,x} = ODP_{inf,x} \cdot \frac{F_{t,x}}{F_{t,CFC-11}} \qquad \text{Equation 5-13}$$

$ODP_{t,x}$ is the ODP at time horizon (*t*) for substance *x*, $ODP_{inf,x}$ is the infinite ODP of substance *x* as provided by the WMO, $F_{t,x}$ is the fraction of damage caused by substance *x* in time t and $F_{t,CFC-11}$ is the fraction of damage caused by CFC-11 at that same time *t*. Moreover, the formula for calculation of ODPs is for the 20 years' time horizon (Individualist) and 100 years' time horizon (Hierarchist). The adoption of ODPs for an infinite time horizon is from the WMO (RIVM Report, 2016-0104). More details about the midpoint characterisation factors in unit (kg CFC-11 equivalents/kg) for 21 ODSs related to three perspectives are available in Appendix 9.

5.6.2.3. From Midpoint to Endpoint

The calculation of endpoint characterisation factors (CFe) for human health damage is (RIVM Report, 2016-0104):

$$CFe_{x,c} = ODP_{x,c} \times F_{M \to E,OD,c} \qquad \text{Equation 5-14}$$

ODPx,c is the ozone depletion potential of substance *x* (in CFC11-eq/kg).

The below equation is the midpoint to endpoint factor for ozone depletion at unit of (DALY/kg CFC11-eq) for cultural perspective c (RIVM Report, 2016-0104).

$F_{M \to E,OD,c}$ Equation 5-15

Moreover, the concept of DALY unit applicable for different effects with the sum them into common unit and the procedure can be summarised as:

$$F_{M \to E,OD,c} = \Delta EESC_{CFC-11} \times \sum_i \sum_q \sum_j \Delta UVB_{i,q} \times EF_{i,q,j,c} \times DF_j \qquad \text{Equation 5-16}$$

$\Delta UVB_{i,q}$ is the increase in UVB radiation (kJ/m2) of bandwidth *q* in region *i*, $EF_{i,q,j,c}$ defines the extra incidence of disease *j* in region *i* caused by UVB radiation of bandwidth *q* for cultural perspective *c* and DF defines the damage to human health affected by incidence of disease *j*, as elaborated in detail by Hayashi et al. (2006). The midpoint to endpoint (DALY/kg CFC-11 eq) factors are different in each perspective due to the presence of different impacts and the difference in time horizon per every perspective, as obvious in Table 5.13.

Table 5.13: Difference in time horizon per every perspective (RIVM Report 2016-0104, p. 43)

Midpoint to endpoint factor	Individualist	Hierarchist	Egalitarian
Human health	2.37E-04	5.31E-04	1.34E-03

5.6.3. Ionising Radiation

This begins at anthropogenic emission of a radionuclide into the environment; the environmental cause-and-effect chain pathway is divided into four sequential steps, as elaborated in Figure 5.8, from airborne or waterborne emission of a radionuclide to human health damage (RIVM Report, 2016-0104).

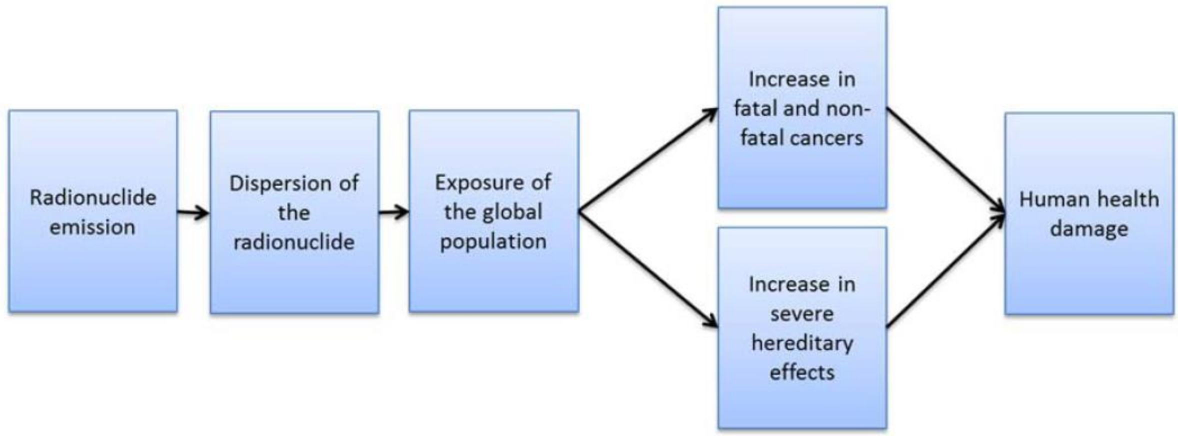


Figure 5.8: Process of cause-and-effect chain of stratospheric ionising radiation impact (RIVM Report 2016-0104, p. 41)

The main generation of anthropogenic emissions of radionuclides is via the fuel cycle (mining, processing and waste disposal), although they are generated during other human activities, for instance, burning of coal and the extraction of phosphate rock. First, the model of dispersion of the radionuclide throughout the environment is included. In addition, an exposure model follows the step which radiation amount such as effective collective dose is received by determination of entire population. On the other hand, exposure to the ionising radiation can be affected by radionuclides, which lead to damaged DNA molecules. Incidence of non-fatal cancers and the incidence of fatal cancers are affected during the analysis step and distinguished from impacts of severe hereditary factors and it is weighted in order to calculate the damage to human health in disability adjusted life years (DALY) as a final step. Currently, quantifying the damage to ecosystems by ionising radiation is not applicable due to the unavailability of impact assessment methodologies (RIVM Report, 2016-0104).

5.6.3.1. Value Choices

There is uncertainty due to different cultural perspectives about ionising radiation, and the choices and different opinions influence the damage modelling. This approach was updated by Schryver et al. (2011) from the original approach used by Frischknecht et al. 2000. Moreover, the consistent of other impact categories are implemented in ReCiPe such as time horizon of assessment, extrapolation from high dose exposure to low dose exposure and cancer types that might affected by ionising radiation, as detailed in Table 5.14. It is obvious that the value choice of time horizon is important for long-lived radionuclides (De Schryver et al. 2011). In addition, some radionuclides have longevity and they use a longer time horizon such as 100,000 years. Moreover, the time horizon can also be short with half-lives of the longest-lived radionuclides, for instance, uranium-235 (half-life 7.1·108 years).

Table 5.14: The effects of substances that emit ionising radiation value choices in the modelling (RIVM Report 2016-0104, p. 46)

Choice category	Individualist	Hierarchist	Egalitarian
Time horizon	20 years	100 years	100,000 years
Dose and dose rate effectiveness factor (DDREF)	10	6	2
Included effects	-Thyroid, bone marrow, lung and breast cancer -Hereditary disease	-Thyroid, bone marrow, lung, breast, bladder, colon, ovary, skin, liver, oesophagus and stomach cancer -Hereditary	-Thyroid, bone marrow, lung, breast, bladder, colon, ovary, skin, liver, oesophagus, stomach, bone surface and remaining cancer

		disease	-Hereditary disease

5.6.3.2. Characterisation Factors at Midpoint Level

The collective dose that is affected by the emission of a radionuclide is the point of the derived characterisation factor at midpoint level. On the other hand, the midpoint characterisation factor, called Ionising Radiation Potential (IRP), is related to the emission of reference substance Cobalt-60 to air, and yields a midpoint factor in Co-60 to air equivalents as per the following equation (RIVM Report, 2016-0104):

$$IRP_{x,i} = \frac{CD_{x,i}}{CD_{Co-60,air}} \qquad \text{Equation 5-17}$$

5.6.3.3. From Midpoint to Endpoint

The calculation of endpoint characterisation factors (CFe) for damage to human health is (RIVM Report, 2016-0104):

$$CFe_{x,i,c} = IRP_{x,i,c} \times F_{M\rightarrow,E,IR,c} \qquad \text{Equation 5-18}$$

IRP_x is the ionising radiation potential of substance *x* to emission compartment *i* (in Co-60 to air eq/kg), while the midpoint to endpoint factor for ionising radiation (DALY/kg Co-60 to air eq.) and for cultural perspective *c* is (RIVM Report, 2016-0104):

$$F_{M\rightarrow,E,IR,c} \qquad \text{Equation 5-19}$$

The influence of receiving a collective dose of radiation has resulted from studies conducted on occupational exposure and from long-term impact studies conducted on the citizens of Hiroshima and Nagasaki, as well as on other people exposed to medium and high doses of radiation. In addition, the influence of different cancer types was assessed by Frischknecht et al. (2000), taking fatal and non-fatal cancer incidence per cancer type.

Furthermore, corresponding midpoint to endpoint factors in (DALY/kBq Co-60 to air equivalents) are calculated for three different cultural perspectives, as elaborated in detail in Table 5.15, and the equation for it is (RIVM Report, 2016-0104):

$$F_{M\rightarrow,E,IR,c} = CD_{Co-60,air} \times \sum_{j} EF_{j,c} \times DF_{j} \qquad \text{Equation 5-20}$$

$EF_{j,c}$ is the modelled extra incidence per disease type *j*, while for perspective *c* and DF is the corresponding damage factor (DALY/incidence) and for disease type *j*. Furthermore, Table 5.15 elaborates the midpoint to endpoint factors for the Individualist, Hierarchist and Egalitarian perspectives in (DALY/kBq Co-60 emitted to air equivalents).

Table 5.15: Midpoint to endpoint factors for three perspectives (RIVM Report 2016-0104, p. 49)

Midpoint to endpoint	**Individualist**	**Hierarchist**	**Egalitarian**
Human health	6.8E-09	8.5E-09	1.4E-08

5.6.4. Fine Particulate Matter Formation

Primary and secondary aerosols can cause air pollution in the atmosphere that has a considerable negative influence on human health extending from respiratory symptoms to hospital admissions and death (WHO, 2006; Friedrich et al. 2011; Burnett et al. 2014; Lelieveld et al. 2015). On the other hand, less than 2.5 μm of Fine Particulate Matter included in represents a complex mixture of organic and inorganic substances, although human intake of PM2.5 influences human health and causes problems as it is inhaled when it reaches the upper part of the airways and lungs. Secondary PM2.5 aerosols, as clarifies by the WHO (2003) form in air from emissions of sulphur dioxide (SO2), ammonia (NH3) and nitrogen oxides (NOx) between other elements. Furthermore, it is possible to attribute mortality effects of chronic PM exposure to PM2.5 rather than to coarser particles of PM, as clarified by WHO studies. In addition, particles with a size of 2.5–10 μm (PM2.5–10) are correlated to respiratory morbidity (WHO, 2006). Figure 5.9 shows the cause and effect of fine dust forming emissions to human health damage (RIVM Report, 2016-0104).

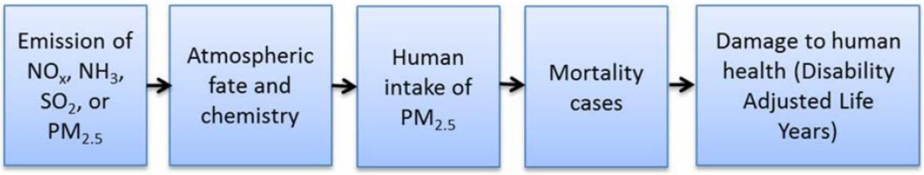


Figure 5.9: Cause and effect of fine dust forming emissions to human health damage (RIVM Report 2016-0104, p. 51)

The damage to human health occurs due to fine dust and, since only short-lived substances are included, the time horizon is not important. In addition, a number of secondary substances are provided as per De Schryver et al. (2011). Moreover, the level of knowledge has a correlation with inclusion and exclusion, because the impacts or exposure are assumed for every perspective, although all secondary data about particulates is contradictory and uncertain in recent studies that describe the impacts of these substances (Lelieveld et al. 2015; Lippmann et al. 2013; Tuomisto et al. 2008). On the other hand, the magnitude of impacts compared to the influences of primary PM is questioned. Moreover, there is no distinction between particles and the effects, so all the influences of secondary particles are included in the hierarchist perspective, although the impacts of secondary particulates from SO2, NH3 and NOx are not included in the individualist perspective (RIVM Report, 2016-0104). Table 5.16 illustrates the value choices in modelling the impacts of fine particulate matter formation (RIVM Report, 2016-0104).

Table 5.16: Value choices in modelling the impacts of fine particulate matter formation (RIVM Report 2016-0104, p. 52)

Choice category	Individualist	Hierarchist	Egalitarian
Included effects	Primary aerosols	Primary aerosols, secondary aerosols from SO2, NH3 and NOx	Primary aerosols, secondary aerosols from SO2, NH3 and NOx

5.6.4.2. Characterisation Factors at Midpoint Level

Damage to human health that correlated with midpoint characterisation due to PM2.5. Although, the pollutant intake is important and the effect and damage have independent precursor substance. In addition, intake fraction (iF) of fine particulate is critical due to emission in region i which is determined by precursor *x* ($iF_{x,i}$). Moreover, the equation calculations of particulate matter formation potential (PMFP) are articulated in primary PM2.5-equivalents by dividing $iF_{x,i}$ with the emission weighted as world average iF of PM2.5 (RIVM Report, 2016-0104):

$$PMFP_{x,i} = \frac{iF_{x,i}}{iF_{PM2.5,world}} \qquad \text{Equation 5-21}$$

$$iF_{x,i} = \frac{\sum_j dC_j \cdot N_j \cdot BR}{dM_{x,i}} \qquad \text{Equation 5-22}$$

On the other hand, Table 5.17 illustrates the average midpoint factors of PM2.5 worldwide and as described as average particulate matter formation potentials of emitted substance x.

Table 5.17: Average midpoint factors PM2.5 worldwide (RIVM Report 2016-0104, p. 53)

Source region	**Continent**	**Emitted substance**			
		PM2.5	**NH3**	**NOx**	**SO2**
World Weighted Average	World	1	0.24	0.11	0.29
Gulf states	Asia	1.22	0.32	0.09	0.36

Table 5.18 shows midpoint characterisation factors for human health damage due to fine dust formation in (kg primary PM2.5-equivalents/kg) as per supporting document for region-specific factors (RIVM Report, 2016-0104).

Table 5.18: Midpoint characterisation factors for human health damage due to fine dust formation (RIVM Report 2016-0104, p. 129)

Pollutant	**Emitted substance**	**Individualist**	**Hierarchist**	**Egalitarian**
Particulate Matter Formation Potential ($PM_{2.5}$-eq/kg)				
PM2.5	NH3	-	0.24	0.24
	NOx	-	0.11	0.11
	SO2	-	0.29	0.29
	PM2.5	1	1	1

Conversion mole mass can be applied, which helps to derive PMFPs-related substances; for instance, for SO and SO3 the CFs will be 0.39 and 0.23 accordingly while NO, NO2, and NO3 are the factors at 0.17, 0.11 and 0.08 accordingly.

5.6.4.3. Characterisation Factors at Endpoint Level

The characterisation of world average endpoint factors is (CFe,x,i), which is for the damage to human health related to particulate matter formation of precursor x and its calculation defined as (RIVM Report, 2016-0104):

$$CF_{e,x,world} = PMFP_{x,world} \times F_{M \rightarrow E,PM2.5} \qquad \text{Equation 5-23}$$

$$F_{M \rightarrow E,PM2.5} \qquad \text{Equation 5-24}$$

The characterisation factors for the midpoint to endpoint factor for the formation of world average particulate matter is unit of (yr/kg PM2.5-eq). In addition, to ensure the consistency of the midpoint to endpoint factor, it is equivalent to the impacts of the world average endpoint characterisation factor for particulate matter with formulation of the equation PM2.5 emissions can be calculated as per the equation (RIVM Report, 2016-0104):

$$F_{M \rightarrow E,PM_{2.5}} = CF_{e,PM2.5,world} = \sum_i \left(\sum_j \left(\left(iF_{PM2.5,i \rightarrow j}\right) \cdot \sum_h \left(EF_{h,j} \cdot DF_{h,j}\right) \right) \cdot Em_{NOx,i} \right) / \sum_i Em_{PM2.5,i} \qquad \text{Equation 5-25}$$

The equation $iF_{PM2.5,i \rightarrow j}$ is the dimensionless population intake fraction of particulate matter in receptor region *j* (in kg/year) ensuing emission change of primary PM2.5 in source region *i* (in kg/year). On the other hand, factors from midpoint to endpoint for PM2.5 emissions are associated with damage to human health in conversion factor of (yr/kg PM2.5-eq), as elaborated in Table 5.20 that region-specific endpoint characterisation factors merging region-specific fate and effect factors, as indicated by Van Zelm et al. (2016) in supporting information in Table 5.19.

Table 5.19: Association of PM2.5 emissions with damage to human health (RIVM Report 2016-0104, p. 54)

Pollutant	Individualist	Hierarchist	Egalitarian
PM2.5	6.29×10^{-4}	6.29×10^{-4}	10^{-4}

Table 5.20 shows region-specific endpoint characterisation factors for the damage to human health due to fine dust formation in (yr·kton-1) as provided by Van Zelm et al. (2016).

Table 5.20: Region-specific endpoint characterisation factors for the damage to human health (RIVM Report 2016-0104, p. 131)

Source region	Continent	Emitted substance			
		PM2.5	**NH3**	**NOx**	**SO2**
World Weighted Average	World	629.2	149.2	70.1	183.2
Gulf states	Asia	5.6E+02	1.4E+02	4.7E+01	2.1E+02

5.6.5. Photochemical Ozone Formation

There are some causes of primary and secondary aerosols in the atmosphere which are related to air pollution and can result in substantial negative effects on human health, ranging from respiratory symptoms to hospital admissions and death, as clarified by a number of authors (Bell et al., 2005; WHO, 2006; Friedrich et al., 2011; Jerrett et al., 2009; Lelieveld et al., 2015). Furthermore, ozone is emitted directly into the atmosphere directly; however, it is formed as an outcome of photochemical reactions of NOx and Non-Methane Volatile Organic Compounds (NMVOCs), which are very intense in the summer season. In addition, ozone is a health hazard to humans since it causes damage to lungs and inflames airways. Moreover, the concentration of ozone results in increased frequency and severity of respiratory distress in humans, for example, asthma and chronic obstructive pulmonary disease (COPD). Ozone also has negative effects on vegetation; for instance, as stated by Ashmore (2005) and Gerosa et al. (2015), it causes reduction of growth and seed production, acceleration of leaf senescence and reduced ability to withstand stressors. Finally, ozone formation, as stated by the European Environment Agency (2005), is non-linear process which is based on meteorological conditions and background concentrations of NOx and NMVOCs (RIVM Report, 2016-0104). Figure 5.10 illustrates the cause-and-effect chain from ozone formatting emissions to damage to human health and ecosystems.

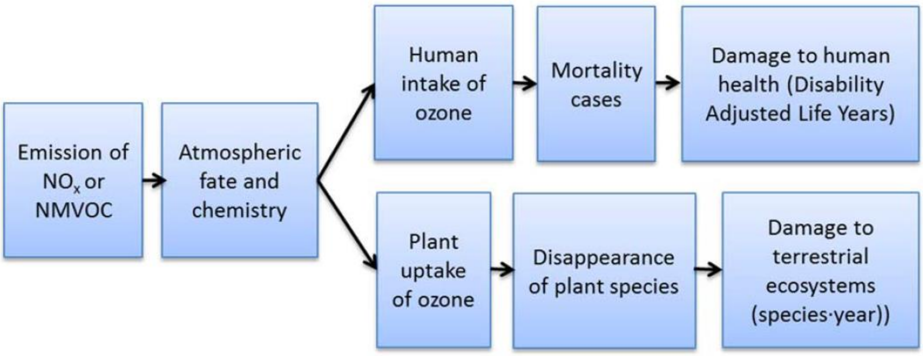


Figure 5.10: Cause and-effect chain from ozone formatting emissions (RIVM Report 2016-0104, p. 56)

There are five consecutive modelling steps from emission to damage, as illustrated in Figure 5.10. The process starts from the emission of NOx or NMVOC, which is followed by atmospheric fate and chemistry in the air. Moreover, NOx and NMVOCs are transformed in air to ozone, which can be inhaled by the human population or taken up by plants, which leads to an increase in mortality among humans and detrimental impacts on plant species. In addition, the end process is the final damage to human health and ecosystems. The effects calculation of ozone impacts is assumed, while the equation of life years affected by respiratory health damage due to exposure to ozone, Disability Adjusted Life Years (DALY), is included as a measurement. Moreover, damage to terrestrial species affected because of the exposure to ozone, species·yr, is included as a measurement (RIVM Report, 2016-0104).

5.6.5.1. Value Choices

The time horizon is not mandatory since short living of damage to ozone substances are included. In addition, there are no other values recognised for this impact category (RIVM Report, 2016-0104).

5.6.5.2. Human Health Damage (midpoint level)

Impacts to ozone of midpoint characterisation factors for human health damage, related to the intake of pollutants, which is critical due to the impacts on damage of precursor substance as it is independent. Although, intake fraction (iF) ozone occurs due to emissions in region i which included per precursor *x* ($iF_{x,i}$). On the other hand, the equation of human health ozone formation potential (HOFP) is calculated by NO_x equivalents and dividing $iF_{x,i}$ by the emission-weighted world average iF of NOx, as elaborated in the following equation (RIVM Report, 2016-0104):

$$HOFP_{x,i} = \frac{iF_{x,i}}{iF_{NOx,world}}$$ Equation 5-26

The definition of region-specific intake fraction is calculated as the sum in the change in the ozone intake rate in every receiving region *j*, which is related to a change in emission of a precursor substance in region *i* ($dM_{x,i}$). Moreover, the calculation of intake rate is started by multiplying the change in concentration of ozone in every receptor region ($dC_{k,j}$) by the population (Nj) in the receptor region *i* and the average breathing rate per person (BR), as is obvious in the below equation (RIVM Report, 2016-0104):

$$iF_{x,i} = \frac{\sum_j dC_j \cdot N_j \cdot BR}{dM_{x,i}}$$ Equation 5-27

On the other hand, Table 5.21 illustrates the world average human health ozone formation potentials (HOFPs) with unit of NOx-eq/kg of emitted substance x.

Table 5.21: World average human health ozone formation potentials (HOFPs) (RIVM Report 2016-0104, p. 57)

Pollutant	Emitted substance	Individualist	Hierarchist	Egalitarian
Ozone	NOx	1	1	1
	NMVOC	0.18	0.18	0.18

Table 5.22 shows the region-specific damage from Human Health Ozone Formation Potentials (HOFPs) in (kg NO_x-eq·kg^{-1}) and damage from Ecosystem Ozone Formation Potentials (EOFP) in (kg NO_x equivalents/ kg).

Table 5.22: Region-specific damage from Human Health Ozone Formation Potentials (HOFP) (RIVM Report 2016-0104, p. 133)

Source region	**Continent**	**HOFP (kg NOx-eq·kg^{-1})**		**EOFP (kg NOx-eq·kg^{-1})**	
		Emitted substance			
		NOx	**NMVOC**	**NOx**	**NMVOC**
World Weighted Average	World	1	0.18	1	0.29
Gulf states	Asia	1.17	0.16	3.04	0.30

The calculation equation for HOFP for a specific hydrocarbon is (RIVM Report, 2016-0104):

$$HOFP_x = \frac{POCP_x}{POCP_{NMVOC}} \times HOFP_{NMVOC} \qquad \text{Equation 5-28}$$

Moreover, supporting information illustrates midpoint factors for individual NMVOCs, which are equal for every perspective, in Appendix 11, which shows the midpoint characterisation factors for individual NMVOCs and signifies Human Health Ozone Formation Potentials (HOFPs) in (NO_x-equivalents/kg) for tropospheric ozone formation (RIVM Report, 2016-0104).

5.6.5.3. Terrestrial Ecosystem Damage (midpoint)

Ecosystem Ozone Formation Potential (EOFP) is calculated in kg NO_x, by dividing $FF_{x,i}$ by the emission-weighted world average FF of NO_x for the following equation (RIVM Report, 2016-0104):

$$EOFP_{x,i} = \frac{FF_{x,i}}{FF_{NOx,world\ average}} \qquad \text{Equation 5-29}$$

Fate factor signifies the sum in change of AOT40 in every receiving grid *g*, which relates to change in the emission of precursor x in region *I*, as clarified by Van Goethem et al. (2013b) in the below equation:

$$\mathrm{FF}_{x,i \rightarrow g} = \sum_{g} \frac{\Delta AOT40_{g}}{\Delta M_{x,i}} \qquad \text{Equation 5-30}$$

Table 5.23: World average ecosystem damage ozone formation potentials (EOFPs) (RIVM Report 2016-0104, p. 58)

Pollutant	Emitted substance	Individualist	Hierarchist	Egalitarian
Ozone	NOx	1	1	1
	NMVOC	0.29	0.29	0.29

To calculate the results of EOFPs for the same substance, mole mass conversion can be included; for example, NO, NO2, and NO3 factors will be 1.53, 1 and 0.74 respectively (RIVM Report, 2016-0104), while the equation calculation of EOFP for a specific hydrocarbon is expressed as:

$$EOFP_x = \frac{POCP_x}{POCP_{NMVOC}} \times EOFP_{NMVOC}$$ Equation 5-31

5.6.5.4. Damage to Human Health (endpoint level)

The calculation of world average endpoint characterisation factors ($CF_{e,x,i}$) for the damage to human health which is related ozone formation of precursor x is expressed as the following equations (RIVM Report, 2016-0104):

$$CF_{e,x,world} = HOFP_{x,world} \times F_{M \to E,HO3}$$ Equation 5-32

$$F_{M \to E,HO3}$$ Equation 5-33

To possess a consistent midpoint to endpoint factor, the factor shall be equal to the emission-weighted world average endpoint characterisation factor for ozone effects related to NOx, as classified in the following equation (RIVM Report, 2016-0104):

$$F_{M \to E,NO_x} = CF_{e,NO_x,world} = \sum_i \left(\sum_j \left((iF_{NOx,i \to j}) \cdot \sum_h (EF_{h,j} \cdot DF_{h,j}) \right) \cdot Em_{NOx,i} \right) / \sum_i Em_{NOx,i}$$ Equation 5-34

The conversation factors of midpoint to endpoint for ozone emissions are associated with damage to human health (yr/kg NOx-eq), as illustrated in Table 5.24, which shows the characterisation factors of region-specific endpoint merging with region-specific fate and effect factors (Van Zelm et al. 2016). Supporting information is available in Table 5.25.

Table 5.24: Characterisation factors of region-specific endpoint merging with region-specific fate and effect factor, (RIVM Report 2016-0104, p. 59)

Pollutant	Individualist	Hierarchist	Egalitarian
Ozone	$9.1x10^{-7}$	$9.1x10^{-7}$	$9.1x10^{-7}$

Table 5.25 shows region-specific characterisation factors of endpoint for human health damage (Van Zelm et al., 2016) and ecosystem damage due to ozone formation.

Table 5.25: Region-specific characterisation factors of endpoint (RIVM Report 2016-0104, p. 139)

Source region	Continent	Human health damage (yr·kton-1)		Ecosystem damage (species·yr/kg)	
		Emitted substance			
		NO_x	NMVOC	NO_x	NMVOC
World Weighted Average	World	9.1·10-1	1.6·10-1	1.29·10-7	3.68·10-8
Gulf states	Asia	9.7E-01	1.6E-01	2.64E-07	3.02E-08

Endpoint characterisation factors (CFe) for terrestrial ecosystem damage are calculated by (RIVM Report, 2016-0104):

$$CF_{e,x,i} = EOFP_{x,i} \times F_{M\rightarrow E,O3}$$ Equation 5-35

$$F_{M\rightarrow E,O3}$$ Equation 5-36

The equation of midpoint to endpoint factor for terrestrial ecosystem damage is (species·year/kg NOx-eq). Moreover, the midpoint to endpoint factor equals the emission-weighted world average endpoint characterisation factor for NOx and it is (RIVM Report, 2016-0104):

$$F_{M\rightarrow E,EO3} = CF_{e,NO_x,world} = SD_{terr} \cdot \sum_i \left(\sum_g \sum_n \left(FF_{NOx,i\rightarrow g} \cdot EF_{n,g} \right) \cdot Em_{NOx,i} \right) / \sum_i Em_{NOx,i}$$ Equation 5-37

Although it is clarified by Van Zelm et al. (2016) that region-specific endpoint characterisation factors, merge with region-specific fate and effect factors. Supporting data is available in Table 5.31.

Table 5.26: Conversion factor of midpoint to endpoint for damage to the ecosystem (RIVM Report 2016-0104, p. 60)

	Individualist	**Hierarchist**	**Egalitarian**
Midpoint to endpoint factor	$1.29 \cdot 10^{-7}$	$1.29 \cdot 10^{-7}$	$1.29 \cdot 10^{-7}$

5.6.6. Terrestrial Acidification

There are some substances associated with atmospheric deposition of inorganic matter that can result in a change in soil acidity, for example, sulphates, nitrates and phosphates. In addition, the definitions of the optimum level of acidity for most plant species are provided. Moreover, the optimum level of acidity is harmful for certain species and is denoted as acidification, and it has been clarified by Goedkoop et al. (1999) and Hayashi et al. (2004) that variations in levels of acidity can result in changes in species' occurrence. The main acidifying emissions are NO_x, NH_3 and SO_2 (Van Zelm et al. 2015). There is a calculation for the characterisation factors for acidification related to vascular plant species in biomes worldwide. In addition, as described by Roy et al. (2012a,b) and Van Zelm et al. (2007b) fate factors, accounting for the environmental persistence of an acidifying substance, can be calculated with an atmospheric deposition model correlated with a geochemical soil acidification model. Moreover, effect factors can be calculated for the ecosystem damage that occurs because of acidification and expressed with dose-response curves of the potential occurrence of plant species as it is resulting from logistic regression functions (Azevedo et al. 2013c). Furthermore, acidification is calculated by dividing endpoint modelling from the emission to the damage with six sequential steps, as elaborated in Figure 5.11. Moreover, the emission of NO_x, NH_3 or SO_2 should be correlated with atmospheric fate before it lands on the soil and leaches into it, which changes the soil solution H+ concentration. The change in acidity can influence plant species living in the soil and lead them to disappear. Figure 5.11 elaborates the cause-and-effect chain starting from acidifying emissions to relative species loss in terrestrial ecosystems (RIVM Report, 2016-0104).

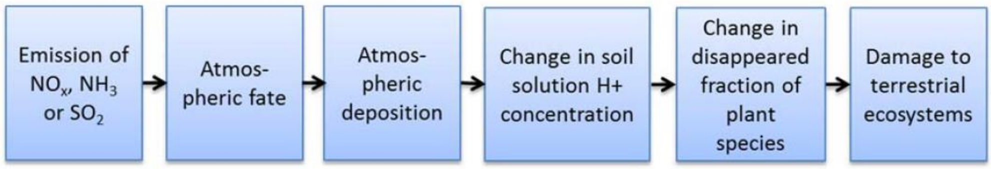


Figure 5.11: Cause-and-effect chain of terrestrial acidification (RIVM Report 2016-0104, p. 61)

5.6.6.1. Value Choices

No value choices have been provided for acidification (RIVM Report, 2016-0104).

5.6.6.2. Characterisation Factors at Midpoint Level

The factors of midpoint characterisation for the damage to terrestrial ecosystems occur due to acidifying emissions, while it is important to know the fate of a pollutant in the atmosphere and soil as the influence of the precursor substance is critical. In addition, the expression for the equation of fate factor (FF) for acidification is correlated to emissions, while in the grid i is determined per precursor *x* ($FF_{x,i}$). Although, Acidification Potential (AP), expressed equivalents in kg SO_2 and it is calculated by dividing $FF_{x,i}$ by the emission-weighted world average FF of SO_2 (RIVM Report, 2016-0104).

$$AP_{x,i} = \frac{FF_{x,i}}{FF_{SO2,world\ average}}$$ Equation 5-38

$$FF_{x,i} = \sum_{j} FF_{air,x,i \to j} \times FF_{soil,x,j}$$ Equation 5-39

Table 5.27 shows the results of world average APs as terrestrial acidification potential emissions of NO_x, NH_3 and SO_2 to air in unit of (kg SO_2-equivalents/kg). The supporting information on country-specific factors is provided in Table 5.28, which shows country-specific terrestrial acidification potentials for the damage to terrestrial ecosystems due to acidifying emissions in (kg SO2-equivalents/kg).

Table 5.27: World average APs for terrestrial acidification potential emissions (RIVM Report 2016-0104, p. 151)

	NOx	**NH3**	**SO2**
World Weighted Average	0.36	1.96	1
United Arab Emirates	0.51	2.10	1.12

Country-specific and world average factors are emission-weighted as shown in Table 5.28 using grid-level data based on year 2005 as per Roy et al. (2012b).

Table 5.28: Country-specific terrestrial acidification potentials for the damage to terrestrial ecosystems (RIVM Report 2016-0104, p. 63)

Substance	**Individualist**	**Hierarchist**	**Egalitarian**
NO_x	0.36	0.36	0.36
NH_3	1.96	1.96	1.96
SO_2	1.00	1.00	1.00

To express the results of CF substances, a conversion based on mole mass can be conducted; for example SO, SO_3 and H_2SO_4 of CFs will be 1.33, 0.8, and 0.65 accordingly

Furthermore, NO, NO_2, and NO_3 will be 0.55, 0.36 and 0.27 accordingly (RIVM Report, 2016-0104).

5.6.6.3. From Midpoint to Endpoint

Endpoint characterisation factors (CFe) for the damage to terrestrial ecosystems are expressed as (RIVM Report, 2016-0104):

$$CF_{e,x,i} = AP_{x,i} \times F_{M\rightarrow E,ACI}$$ Equation 5-40

$$F_{M\rightarrow E,ACI}$$ Equation 5-41

The terrestrial acidification equals the emission-weighted world average endpoint characterisation factor for SO2 as expressed in the below equation (RIVM Report, 2016-0104):

$$F_{M\rightarrow E,ACI} = CF_{e,SO_2,world} = SD_{terr} \cdot \sum_i \left(\sum_j \left(FF_{air,SO2,i\rightarrow j} \cdot FF_{soil,SO2,j} \cdot EF_j \right) \cdot Em_{SO2,i} \right) / \sum_i Em_{SO2,i}$$ Equation 5-42

Table 5.29 illustrates the conversion factors for acidification from midpoint to endpoint (species·yr/kg SO_2-eq), while characterisation factors of the country-specific endpoint are available in supporting data in Table 5.36 (RIVM Report, 2016-0104).

Table 5.29: Conversion factors for acidification from midpoint to endpoint (RIVM Report 2016-0104, p. 64)

	Individualist	**Hierarchist**	**Egalitarian**
Midpoint to endpoint factor	$2.12 \cdot 10^{-7}$	$2.12 \cdot 10^{-7}$	$2.12 \cdot 10^{-7}$

The expression of CF substances' results is conducted by applying conversion based on mole mass; for example, SO and SO_3, so CFs will represent 1.33 and 0.8 respectively (RIVM Report, 2016-0104).

Table 5.30 shows country-specific characterisation factors of the endpoint for the damage to terrestrial ecosystems due to acidifying emissions (species·yr/kg) as stated by Roy et al. (2014).

Table 5.30: Country-specific characterisation factors of the endpoint for the damage to terrestrial ecosystems (RIVM Report 2016-0104, p. 155)

	NOx	**NH3**	**SO2**
World Weighted Average	7.70·10-8	4.14·10-7	2.12·10-7
United Arab Emirates	1.04E-07	3.98E-07	3.54E-07

5.6.7. Freshwater Eutrophication

The occurrence of freshwater eutrophication is due to the discharge of nutrients into the soil or freshwater bodies and the subsequent rise in nutrient levels, such as phosphorus and nitrogen. Furthermore, ecological influence is correlated with freshwater eutrophication and they are various. Moreover, it is by following structure environmental impacts offset by increasing nutrient emissions into freshwater, which results in increasing nutrient uptake by autotrophic organisms such as cyanobacteria and algae, and heterotrophic species such as fish and invertebrates. In addition, those effects lead to relative loss of species. Moreover, emission impacts freshwater are based on the transfer of phosphorus from the soil to freshwater bodies, which residence time in freshwater systems and on Potentially Disappeared Fraction (PDF) by increase in phosphorus concentrations in freshwater (RIVM Report, 2016-0104). Figure 5.12 shows the chain of cause and effect for phosphorus emissions that causes loss in freshwater species' richness

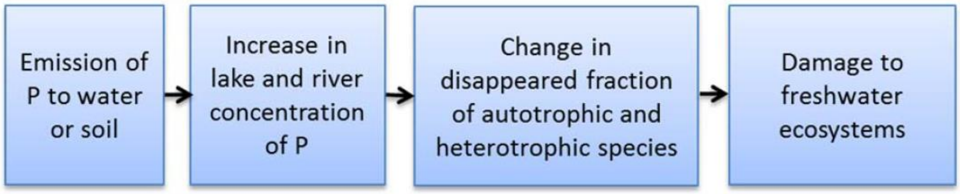


Figure 5.12: Cause and effect of freshwater eutrophication (RIVM Report 2016-0104, p. 66)

5.6.7.1 Value Choices

No value choices were provided for modelling of fate and effects of P emissions (RIVM Report, 2016-0104).

5.6.7.2. Characterisation Factors at Midpoint Level

The expression equation of world average of freshwater eutrophication midpoint factors is (RIVM Report, 2016-0104):

$$FEP_{x,c,i} = \frac{FF_{x,c,i}}{FF_{P,fw,world\ average}} \qquad \text{Equation 5-43}$$

In addition, Table 5.31 shows the world average of FEPs. Moreover, using the same methodology for phosphoric acid and phosphorus pentoxide leads to FEP results for emissions to freshwater of 0.32 and 0.22 kg P-eq to freshwater/kg, accordingly. Country-specific factors are provided as supporting data in Table 5.31 Furthermore, Table 5.32 shows the potentials of freshwater eutrophication for phosphorus and phosphate to freshwater, agricultural soil and seawater (in kg P to freshwater equivalents/ kg), which are equivalent for all perspectives (RIVM Report, 2016-0104).

Table 5.31: Potentials of freshwater eutrophication for phosphorus and phosphate (RIVM Report 2016-0104, p. 67)

Substance	Emission compartment	FEP (kg P-eq to freshwater/kg)
Phosphorus (P)	freshwater	1.00
	agricultural soil	0.10
	seawater	0
Phosphate ($PO4^{3-}$)	freshwater	0.33
	agricultural soil	0.033
	seawater	0

Table 5.32 shows country-specific freshwater eutrophication potentials for the damage to freshwater ecosystems (kg P-equivalents/kg). However, data for the United Arab Emirates is not provided (RIVM Report, 2016-0104).

Table 5.32: Country-specific freshwater eutrophication potentials for the damage to freshwater ecosystems (RIVM Report 2016-0104, p. 157)

Country	Emitted to freshwater		Emitted to soil	
	P	**$PO4^{3-}$**	**p**	**$PO4^{3-}$**
na	na	na	na	na

5.6.7.3. From Midpoint to Endpoint

Endpoint characterisation factors (CFe) for the damage of freshwater eutrophication ecosystems are expressed by using the equation (RIVM Report, 2016-0104):

$$CFe_{x,i} = FEP_{x,i} \times F_{M\rightarrow E,FE}$$ Equation 5-44

$$F_{M\rightarrow E,FE}$$ Equation 5-45

Equation 5-45 is the conversion factor of midpoint to endpoint for freshwater eutrophication (species·year/kg P to freshwater-eq). To ensure the consistency of the midpoint to endpoint conversion factor, freshwater eutrophication equivalent to emission-weighted world average endpoint characterisation factor for P emitted to freshwater by following the equation (RIVM Report, 2016-0104):

$$F_{M\rightarrow E,FW} = SD_{fw} \times \sum_i \left(\sum_j \left(FF_{e,i,j} \cdot \overline{EF_j} \right) \cdot Em_{P,i} \right) / \sum_i Em_{P,i}$$ Equation 5-46

In addition, the impact factor of species group k and water type w in grid cell *j* is expressed by using the equation (RIVM Report, 2016-0104):

$$EF_{w,k,j} = \frac{0.5}{10^{\alpha_{w,k,j}}}$$ Equation 5-47

In addition, supporting information is available for effect factors and alphas for each climate-water, type-species group combination in tables 5.33 and 5.34 (RIVM Report, 2016-0104).

Table 5.33: Linear effect factors for streams and lakes based on different climate zones (RIVM Report 2016-0104, p. 157)

	Heterotrophic species		Autotrophic species		Combined
	Lake [PDF·m³/kg]	Stream	Lake [PDF·m³/kg]	Stream	Lake Stream [PDF·m³/kg]
tropical	13,458	778	813	2,323	7,135
sub-tropical	13,458	778	813	2,323	1,550
temperate	1,253	674	5,754	766	7,135
cold	18,280	674	8,530	766	1,550
xeric	13,458	778	2,594	2,323	3,504

Table 5.34: Alfa for streams and lakes of different climate zones (RIVM Report 2016-0104, p. 157)

	Heterotrophs		Autotrophs	
	Lake [PDF·L/mg]	Stream	Lake [PDF· L /mg]	Stream
sub-tropical	-1.430	-0.192	-0.211	-0.667
temperate	-0.399	-0.130	-1.061	-0.185
cold	-1.563		-1.232	na
xeric	na		-0.715	na
	na			

The conversion factor of the midpoint to endpoint for freshwater eutrophication for all perspectives (species.yr/kg P-eq) is elaborated in Table 5.35, while the supporting data for emission-weighted country-specific endpoint characterisation factors is provided in Table 5.36 (RIVM Report, 2016-0104).

Table 5.35: Conversion factor of midpoint to endpoint for freshwater eutrophication for all perspectives (RIVM Report 2016-0104, p. 68)

Midpoint to endpoint factor	species.yr/kg P-eq
Freshwater ecosystems	6.7E-7

Table 5.36 shows country-specific endpoint characterisation factors for freshwater eutrophication (in species·yr/kg). However, data for United Arab Emirates is not provided (RIVM Report, 2016-0104).

Table 5.36: Country-specific endpoint characterisation factors for freshwater eutrophication (RIVM Report 2016-0104, p. 162)

Country	Emitted to freshwater		Emitted to soil	
	P	**$PO4^{3-}$**	**p**	**$PO4^{3-}$**
na	na	na	na	na

5.6.8. Marine Eutrophication

Marine eutrophication happens because of the runoff and leaching of plant nutrients from the soil to discharge into riverine or marine systems, with a subsequent rise in nutrient levels, for example, phosphorus and nitrogen (N). Furthermore, N is assumed to be a limiting nutrient in marine waters (Cosme et al., 2015). On the other hand, ecological influence is correlated with marine eutrophication, because nutrient enrichment points to a variety of ecosystem impacts, one of which is the existence of benthic oxygen depletion. In addition, it leads to the start of hypoxic waters if in excess to anoxia as called ‘dead zones’, which is one of the most severe and widespread root causes of disturbance to marine ecosystems. Moreover, impacts on marine water are related to certain circumstances, such as transfer of dissolved inorganic nitrogen (DIN) from soil and freshwater bodies or straight to marine water, which residence time in marine systems on dissolved oxygen (DO) depletion and on potentially disappeared fraction (PDF), as clarified in the model function of DIN emission in Figure 5.13. Figure 5.13 illustrates the cause-and-effect chain process for the emission of dissolved inorganic nitrogen (DIN), which comes from diffuse sources, such as runoff and leachate from soils. Direct emission to rivers and coastal waters leads to loss of marine species’ richness (RIVM Report, 2016-0104).

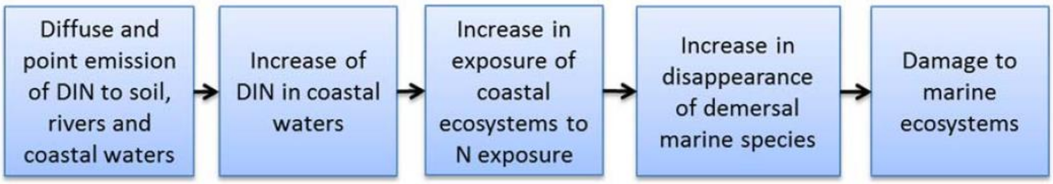


Figure 5.13: Cause-and-effect of marine eutrophication (RIVM Report 2016-0104, p. 69)

5.6.8.1. Value Choices

No value choices were provided for modelling of the fate and effects of N emissions (RIVM Report, 2016-0104).

5.6.8.2. Characterisation Factors at Midpoint Level

The marine eutrophication potential of substance *x* for emission to compartment *c* (MEPx,c) is signified as emission (E)-weighted combined fate factor and exposure factor, which is scaled to the world average of N emitted to marine water and the equation is (RIVM Report, 2016-0104):

$$MEP_{x,c} = \frac{\sum\left(FF_{x,c,LME} \cdot XF_{x,c,LME} \cdot E_{x,c,LME}\right)}{\sum E_{x,c,LME}} \Bigg/ \frac{\sum\left(FF_{N,mw,LME} \cdot XF_{N,mw,LME} \cdot E_{N,mw,LME}\right)}{\sum E_{N,mw,LME}} \qquad \text{Equation 5-48}$$

Table 5.37 shows the expression results of world average Marine Eutrophication Potentials (MEPs) for Dissolved Inorganic Nitrogen (DIN) emissions to freshwater (rivers), soil and coastal waters. The unit is given as (kg N to marine water-equivalents/kg), which is applicable for all perspectives. In addition, supporting data is available for continent-specific factors in Table 5.38 (RIVM Report, 2016-0104).

Table 5.37: Expression results of world average Marine Eutrophication Potentials (MEPs) (RIVM Report 2016-0104, p. 70)

Substance	**Emission compartment**	**MEP (kg N-eq to marine water/kg)**
Nitrogen (N)	Freshwater (rivers)	0.30
	Soil	0.13
	Coastal waters	1.00
Ammonia ($NH4^+$)	Freshwater (rivers)	0.23
	Soil	0.10
	Coastal waters	0.78
Nitrogen dioxide (NO_2)	Freshwater (rivers)	0.09
	Soil	0.04
	Coastal waters	0.30

Table 5.38: Continent-specific characterisation factors of marine eutrophication potentials for marine ecosystem damage (kg N-equivalents/kg) (RIVM Report 2016-0104, p. 168)

	Emitted to rivers			**Emitted to soil**			**Emitted to coastal waters**		
Country	**N**	**NH_4^+**	**NO_2**	**N**	**NH_4^+**	**NO_2**	**N**	**NH_4^+**	**NO_2**
North Asia	0.48	0.37	0.15	0.08	0.07	0.03	1.38	1.07	0.42
South Asia	0.33	0.25	0.10	0.18	0.14	0.06	1.26	0.98	0.38

5.6.8.3. From Midpoint to Endpoint

The characterisation factors for the endpoint (CFe) of marine eutrophication ecosystem damage are expressed by using (RIVM Report, 2016-0104):

$$CFe_{x,i} = MEP_{x,i} \times F_{M\rightarrow E,ME} \qquad \text{Equation 5-49}$$

$$F_{M\rightarrow E,ME} \qquad \text{Equation 5-50}$$

In addition, to ensure the consistency of the conversion factor from midpoint to endpoint, the midpoint to endpoint factor for marine eutrophication equals the emission-weighted world average endpoint characterisation factor for N emitted to marine water by using (RIVM Report, 2016-0104):

$$\mathrm{F}_{\mathrm{M}\rightarrow\mathrm{E,ME}} = \frac{\sum\left(\mathrm{FF}_{\mathrm{N,mw,LME}} \cdot \mathrm{XF}_{\mathrm{N,mw,LME}} \cdot \mathrm{EF}_{\mathrm{N,mw,LME}} \cdot \mathrm{SD}_{\mathrm{mw,LME}}\right) \cdot \mathrm{E}_{\mathrm{N,mw,LME}}}{\sum \mathrm{E}_{\mathrm{N,mw,LME}}} \quad \text{Equation 5-51}$$

Moreover, the effect factor signifies a change in the potentially affected fraction of species (PAF) which, because the change is in Dissolved Oxygen (DO), the expression equation is:

$$EF_{N,mw,LME} = \frac{\Delta PAF_{LME}}{\Delta DO_{LME}} = \frac{0.5}{HC50_{LME}} \quad \text{Equation 5-52}$$

The conversion from midpoint to endpoint factor that relates to marine eutrophication is illustrated in Table 5.39. Supporting data for climate zone-specific EFs is available in Table 5.40, which ranges from 218 to 306 (PAF)·m^3·kgO_2^{-1}).

Table 5.39: Conversion factor from midpoint to endpoint of marine eutrophication, (RIVM Report 2016-0104, p. 72)

Table 5.40 shows the climate zone-specific HC50 and effect factors (Cosme & Hauschild, 2016) as supporting data for marine eutrophication.

Midpoint to endpoint factor	**species.yr/kg N-eq**
Marine ecosystems	1.7E-9

Table 5.40: Climate zone-specific HC50 and effect factors (RIVM Report 2016-0104, p. 167)

Climate zone	**$HC50_{LOEC}$ [mgO_2 L^{-1}]**	**EF [(PAF) m^3 kgO_2^{-1}]**
	2.29	218
Subpolar	2.07	242
Temperate	1.80	278
Subtropical	1.82	275
Tropical	1.64	306
Global average	1.89	264

Table 5.41 shows continent-specific characterisation factors of the endpoint for the damage to marine eutrophication ecosystems (species.yr/ kg) as provided by Cosme and Hauschild (2017).

Table 5.41: Continent-specific characterisation factors of the endpoint for the damage to marine eutrophication ecosystems (RIVM Report 2016-0104, p. 168)

	Emitted to rivers			**Emitted to soil**			**Emitted to coastal waters**		
Country	**N**	**NH_4^+**	**NO_2**	**N**	**NH_4^+**	**NO_2**	**N**	**NH_4^+**	**NO_2**
North Asia	1.5E-10	1.2E-10	4.6E-11	3.2E-11	2.5E-11	9.7E-12	4.1E-10	3.2E-10	1.2E-10
South Asia	5.2E-10	4.0E-10	1.6E-10	2.4E-10	1.9E-10	7.3E-11	2.1E-09	1.6E-09	6.4E-10

5.6.9. Toxicity

The characterisation factor of human toxicity and ecotoxicity is associated with environmental persistence (fate), accumulation in human food chain (exposure) and toxicity effect of chemicals. Figure 5.14 elaborates the cause-effect chain pathway, which starts from emission to environment through fate and exposure, in which impacts are related to species and disease incidences, which leads to damage to ecosystems and human health (RIVM Report, 2016-0104).

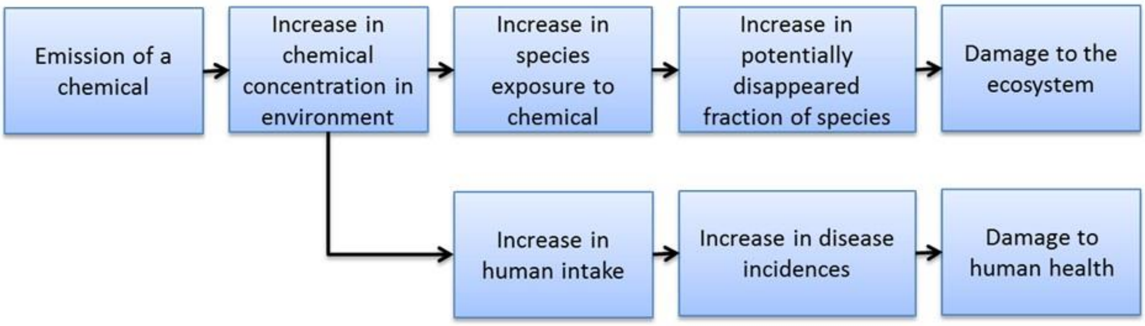


Figure 5.14: Cause-effect chain of toxicity effect (RIVM Report 2016-0104, p. 73)

The calculation of fate and exposure factors is as denoted by (evaluative) multimedia fate and exposure models. Moreover, the effect factors are calculated from toxicity data on humans and laboratory animals. Multimedia fate is applied frequently, while the exposure and effects model is related to the Uniform System for the Evaluation of Substances adapted for LCA (USES-LCA) (Van Zelm et al. 2009), which is a global multimedia fate, exposure and effects model, and has been updated to version 3.0. Moreover, the calculation is provided for environmental fate and exposure factors in multiple compartments and human intake fractions for inhalation and oral intake in which there are 10 emission compartments. On the other hand, the fate part has dependent time, which is calculated by USES-LCA 3.0 by applying the model for numerous time horizons, for example, 20-100 years, and a steady-state option. The most recent updates of USES-LCA 3.0 are available in the supporting data in Appendix 12 (RIVM Report, 2016-0104).

5.6.9.1. Value Choices

Uncertainty calculation for value choices is provided through different cultural perspectives. In addition, most of the choices depend on different modelling opinions of effect and damage, while the choices are correlated with three perspectives, as shown in Table 5.42 (RIVM Report, 2016-0104).

5.6.9.2. Time Horizon

Huijbregts et al. (2001) state that the impact of metals mainly depends on the interest of time horizon. In addition, the egalitarian scenario is counted as infinite for the time horizon, the hierarchic scenario counts for 100 years for the time horizon and the individualist scenario counts for 20 years for the time horizon (RIVM Report, 2016-0104).

5.6.9.3. Exposure Routes

The concept of bioconcentration can be applied for organic pollutants but does not hold for inorganics. In addition, Hendriks et al. (2001) state that concentrations of internal body for metals increase less than proportionally with increases in concentration in the environment. Moreover, the calculation of sensitivity of human population intake fractions for metals is predicted in the egalitarian and hierarchic scenarios that human exposure occurs through all intake routes, for example, air, drinking water and food. In contrast, the individualistic scenario predicted that human exposure occurs through only air and drinking water (RIVM Report, 2016-0104).

5.6.9.4. Marine Ecotoxicity

The potential impact on the marine environment is closely related to accounts of additional inputs for essential metals to oceans, which lead to toxic effects. The egalitarian and hierarchic scenarios provided the calculation of sea and oceanic compartments of marine ecotoxicological impacts, while the individualistic scenario included the calculation of just the sea compartment for essential metals. Essential metals are cobalt, copper, manganese, molybdenum and zinc (RIVM Report, 2016-0104).

5.6.9.5. Carcinogenicity

Not all substances with a carcinogenic substance ED50 are important for carcinogenics to humans. The International Agency for Research on Cancer (IARC) is part of the World Health Organisation (WHO), and assessed the carcinogenic risk of 844 substances (mixtures) to humans by assigning a carcinogenicity class to every substance (IARC, 2004). In addition, classes indicate the strength of evidence for carcinogenicity, which is expressed from studies in humans and in experiments on animals and other significant data. Furthermore, the egalitarian and hierarchic scenarios include all 844 substances (IARC-category 1, 2A, 2B, 3 or no classification), while the individualistic scenario includes only the substances with robust evidence of carcinogenicity (IARC-category 1, 2A and 2B).

5.6.9.6. Minimum Number of Species Tested for Ecotoxicity

The uncertainty aspect is highly related to the effect of ecotoxicity factors if only a low number of tested species are included, mainly lower than four species (Van Zelm et al. 2007a; Van Zelm et al. 2009). Moreover, there is a minimum number of four tested species for the individualistic scenario, whilst the hierarchic egalitarian scenarios have no minimum requirements. On the other hand, Schryver et al. (2011) state three perspectives, as shown in Table 5.42.

Table 5.42: Perspectives regarding the minimum number of species tested for ecotoxicity (RIVM Report 2016-0104, p. 75)

Choice category	**Individualist**	**Hierarchist**	**Egalitarian**
Time Horizon	20 years	100 years	Infinite
Exposure Routes for Human Toxicity	Organics: all exposure routes. Metals: drinking water and air only	All exposure routes for all chemicals	All exposure routes for all chemicals
Environmental Compartments for Marine Ecotoxicity	Sea + ocean for organics and nonessential metals. For essential metals, sea compartment included only, excluding oceanic compartments	Sea + ocean for all chemicals	Sea + ocean for all chemicals
Carcinogenicity	Only chemicals with TD50 classified as 1, 2A, 2B by IARC	All chemicals with reported TD50	All chemicals with reported TD50
Minimum Number of Species Tested for Ecotoxicity	4	1	1

5.6.9.7. Characterisation Factors at Midpoint Level

Table 5.43 recaps the emission compartments, environment receptors and human exposure routes that are available in the human TP calculations (RIVM Report, 2016-0104).

Table 5.43: Review of the emission compartments, environment receptors and human exposure routes (RIVM Report 2016-0104, p. 76)

Emission compartments	**Environmental receptors**	**Human exposure routes**
Urban air	Terrestrial environment (excl. agri and urban land)	Inhalation
Rural air	Freshwater environment	Ingestion via root crops
Freshwater	Marine environment	Ingestion via leaf crops
Seawater		Ingestion via meat products
Agricultural soil		Ingestion via dairy products
Industrial soil		Ingestion via eggs
		Ingestion via freshwater fish
		Ingestion via marine fish
		Ingestion via drinking water

The expression equation for the compartment-specific ecotoxicological midpoint characterisation factor consists of the fate factor (FF), which is an effect factor (EF):

$$\mathrm{ETP}_{x,i,j,c} = \sum_{g} \frac{\mathrm{FF}_{x,i,j,g,c} \times \mathrm{EF}_{x,j,c}}{\mathrm{FF}_{\mathrm{DCB},ref,j,g,c} \times \mathrm{EF}_{\mathrm{DCB},j,c}} \qquad \text{Equation 5-53}$$

In addition, all the mentioned toxicity potentials are grouped into an overall human population characterisation factor of substance x that is emitted to compartment i with use of equation (RIVM Report, 2016-0104):

$$\mathrm{HTP}_{i,x,c/nc,c} = \sum_{r} \sum_{g} \frac{\mathrm{iF}_{x,i,r,g,c} \times \mathrm{EF}_{x,r,c/\,nc,c}}{\mathrm{iF}_{\mathrm{DCB},ua,r,g,c} \times \mathrm{EF}_{\mathrm{DCB},r,c/nc,c}} \qquad \text{Equation 5-54}$$

On the other hand, Appendix 13 shows three cultural perspectives and four emission compartments of ETPs and HTPs as midpoint characterisation factors (1,4-DCB eq/kg) for 1,4-DCB and nickel (RIVM Report, 2016-0104).

5.6.9.8. From Midpoint to Endpoint

The ecotoxicological endpoints consist of freshwater, marine and terrestrial ecotoxicity. Moreover, the characterisation factors of endpoint (CF_{eco}) for ecotoxicity are expressed by using the equation (RIVM Report, 2016-0104):

$$CFeco_{x,i,j,c} = ETP_{x,i,j,c} \times F_{M\rightarrow,E,ETOX,j,c} \qquad \text{Equation 5-55}$$

$ETP_{x,i,j,c}$ denote the ecotoxicity potential for environmental endpoint j (freshwater, marine, terrestrial) of substance x to emission compartment i, which is associated with the cultural perspective c by using unit (in 1,4DCB-eq/kg).

$$F_{M\rightarrow,E,ETOX,j,c} \qquad \text{Equation 5-56}$$

In addition, midpoint to endpoint factors for ecotoxicity equal endpoint characterisation factors for 1,4DCB that are emitted correspondingly to freshwater, such as freshwater ecotoxicity, seawater, for instance, marine ecotoxicity, and industrial soil, for example, terrestrial ecotoxicity, which includes species concentrations by using the equation (RIVM Report, 2016-0104):

$$F_{M\rightarrow,E,ETOX,j,c} = \sum_{g} SD_j \times FF_{DCB,ref,j,g,c} \times EF_{DCB,j,c} \qquad \text{Equation 5-57}$$

Furthermore, for human health damage, endpoint characterisation factors for carcinogenic and non-carcinogenic (CFhum) factors are expressed by using the equation (RIVM Report, 2016-0104):

$$CFhum_{x,i,c/nc,c} = HTP_{x,i,c/nc,c} \times F_{M\rightarrow,E,HTOX,c/nc,c} \qquad \text{Equation 5-58}$$

$HTP_{x,i,nc/c,c}$ signifies the human toxicity potential for carcinogenic or non-carcinogenic effects of substance x to emission compartment i, which is associated with the cultural perspective c in units of (1,4DCB-eq/kg).

$$F_{M\rightarrow,E,HTOX,c/nc,c} \qquad \text{Equation 5-59}$$

The midpoint to endpoint factor for human carcinogenic or non-carcinogenic toxicity is applicable for cultural perspective c, while the midpoint to endpoint factor for human carcinogenic or non-carcinogenic toxicity equals the endpoint characterisation factor for 1,4DCB emitted to urban air as an equation expressed as (RIVM Report, 2016-0104):

$$F_{M\rightarrow E,HTOX,c/nc,c} = \sum_{r}\sum_{g} iF_{1,4-DCB,r,g,ua} \times EF_{1,4-DCB,r,c/nc} \times DF_{c/nc} \qquad \text{Equation 5-60}$$

$DF_{c/nc}$ signifies the damage factor for carcinogenic or non-carcinogenic effects, which equals 11.5 and 2.7 disability adjusted life years (DALYs) per incidence case respectively. On the other hand, Table 5.44 shows the midpoint to endpoint conversion factors for all perspectives (RIVM Report, 2016-0104).

Table 5.44: Midpoint to endpoint conversion factors for all perspectives (RIVM Report 2016-0104, p. 80)

Midpoint to endpoint	**Unit**	**Value**
Freshwater ecotoxicity	species·yr/kg 1,4-DCB eq	6.95E-10
Marine ecotoxicity	species·yr/kg 1,4-DCB eq	1.05E-10
Terrestrial ecotoxicity	species*yr/kg 1,4-DCB eq	1.14E-11
Human toxicity (cancer)	DALY/kg 1,4-DCB eq	3.32E-06
Human toxicity (non-cancer)	DALY/kg 1,4-DCB eq	2.28E-07

5.6.10. Water Use

Figure 5.15 shows relevant impact pathways, in which all used water-related impacts are correlated with water consumption. In addition, water consumption is defined as the use of water in a way that it evaporates, is integrated into products, transferred to other watersheds or disposed of into the sea (Falkenmark et al. 2004). Furthermore, consumed water is no longer available in watersheds for humans and ecosystems (RIVM Report, 2016-0104).

Modelling of water use starts with availability quantification of freshwater reduction. With regard to humans, availability reduction in freshwater leads to challenges between different water uses. Moreover, a small percentage of irrigation leads to a decrease in the production and an increase in malnutrition between local populations. However, the relation of people's vulnerability to malnutrition is increasing compared to lower human development indexes (HDI) in a range of industrial countries (HDI>0.88) that has enough evidence in order to buy food and avoid malnutrition and which results in no damage arising to human health. On the other hand, the modelling of terrestrial ecosystems' impacts is through potential decline in vegetation and plant variety. Moreover, a decrease in blue water, for example, water in lakes, rivers, aquifers and precipitation, leads to a decrease in the available green water in soil moisture and causes a reduction in plant species. Furthermore, a fraction of freshwater fish evaporate based on estimation of water consumption that is related to species discharge as associated with river entrances (RIVM Report, 2016-0104). Figure 5.15 shows the cause-and-effect chain of water consumption, which leads to impacts on human health and ecosystem quality, both terrestrial and freshwater. The evaporation of freshwater fish species was highlighted by Hanafiah et al. (2011), while the other two relate to work by Pfister et al. (2009).

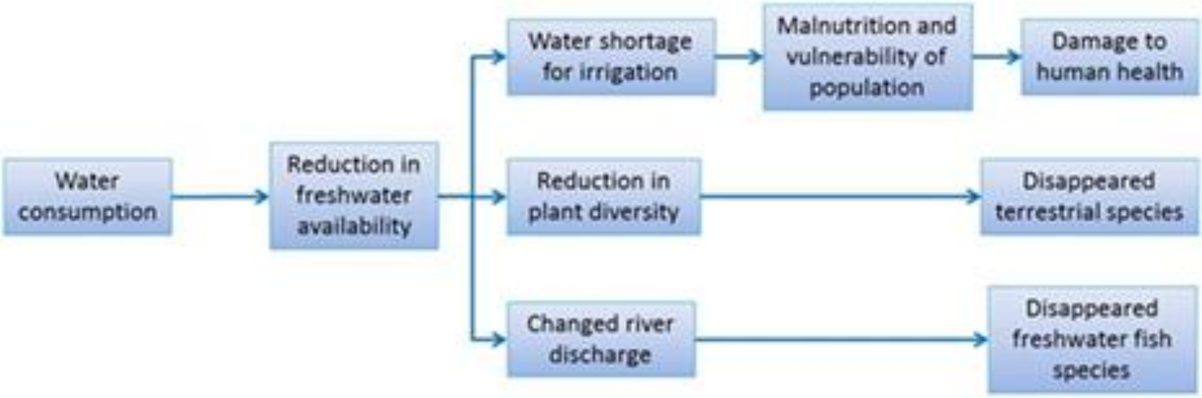


Figure 5.15: Cause-and-effect chain of water consumption use (RIVM Report 2016-0104, p. 82)

5.6.10.1. Value Choices

Correlated value choices of water consumption impacts on human health are based on moderately intensely on management of watersheds that results in change variation factors and management of agricultural practices in reducing the quantity of water. In addition, both require the production of a certain quantity of food. Table 5.45 shows the three cultural perspectives and respective value choices for modelling the impacts of water consumption, which are from De Schryver et al. (2011), ignoring age weighting.

Table 5.45: Cultural perspectives and respective value choices for modelling the impacts of water consumption (RIVM Report 2016-0104, p. 82)

Choice category	**Individualist**	**Hierarchist**	**Egalitarian**
Human health Regulation of stream flow	High	Standard	Standard
Water requirement for food production	1,000 m^3/yr·capita (efficient management)	1,350 m^3/yr·capita (standard management)	1,350 m^3/yr·capita (standard management)
Terrestrial ecosystems	Zero (too uncertain)	Default value	Default value

5.6.10.2. Characterisation Factors at Midpoint Level

The calculation uses total amount of water withdrawn, irrespective of return flows to water bodies or efficiencies of water use, while water consumption is defined as the quantity of water that the original watershed is losing, and the equation expression is (RIVM Report, 2016-0104):

$$CF_{mid} = \begin{cases} 1 & if\ inventory\ in\ m^3\ consumed \\ water\ requirement\ ratio & if\ inventory\ in\ m^3\ withdrawn \end{cases} \qquad \text{Equation 5-61}$$

Figure 5.16 illustrates the global average value, which is 0.44, and standard deviation of 0.14, and it shows the water requirement ratio for converting agricultural water extraction to agricultural water consumption (RIVM Report, 2016-0104).

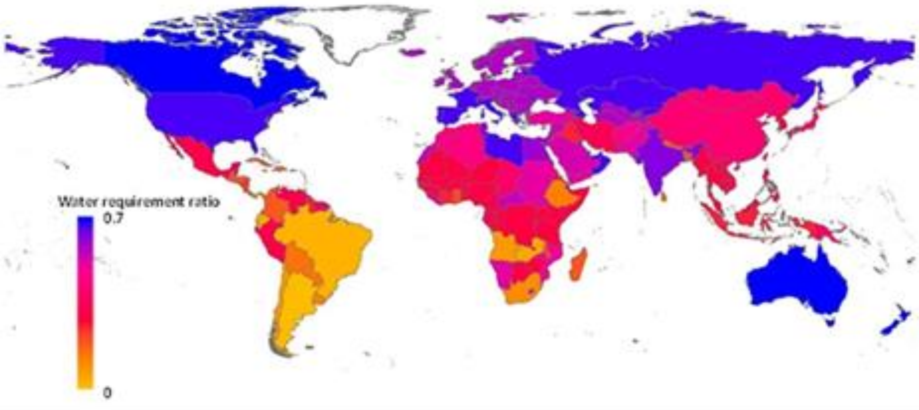


Figure 5.16: Global average value of water-use efficiency (RIVM Report 2016-0104, p. 83)

Table 5.46 shows the water requirement ratio for agriculture for the United Arab Emirates (Döll & Siebert, 2002; AQUASTAT FAO, 2012); Verones et al., 2013) as supporting data for water stress.

Table 5.46: Water requirement ratio for agriculture in the UAE (RIVM Report 2016-0104, p. 175)

Country	**Water requirement ratio**
United Arab Emirates	0.6

Table 5.47 provides a summary of the recommended ratios for water requirements, which are converted to water consumption.

Table 5.47: Ratios for water requirements (RIVM Report 2016-0104, p. 84)

	Surface water	**Groundwater**
Agriculture	0.44	0.44
Industry	0.1	1
Domestic	0.1	1

5.6.10.3. From Midpoint to Endpoint (Human Health)

Characterisation factors for the endpoint related to impacts of water consumption on human health are signified as water stress index (WSI) as part of the modelling pattern. Moreover, damage calculation is denoted as in disability adjusted life years (DALYs) for every watershed or country by using the water stress index (WSI) established by Pfister et al. (2009). Furthermore, the WSI signifies the ratio between the sum of freshwater withdrawals, which is not considered as consumption for different sectors j (WU), and hydrological availability in the watershed i (WA) expressed by using the equation (RIVM Report, 2016-0104):

$$WTA_i = \frac{\sum_j WU_{ij}}{WA_i} \qquad \text{Equation 5-62}$$

The WSI is calculated for every watershed and every country independently. It is a logistic function scales range of water stress between 0.01 to 1. Moreover, it starts at 0.01 and not at 0, because every water extraction has at least a marginal local impact, as clarified by Pfister et al. (2009). On the other hand, results are expressed by using the equation:

$$WSI = \frac{1}{1 + e^{-6.4 \cdot WTA*} \cdot \left(\frac{1}{0.01} - 1\right)} \qquad \text{Equation 5-63}$$

Furthermore, the expression equation of WTA* (RIVM Report, 2016-0104) is:

$$WTA *= \begin{cases} \sqrt{VF} \cdot WTA & for\ SRF \\ VF \cdot WTA & for\ non - SRF \end{cases} \qquad \text{Equation 5-64}$$

Variation factor (VF) is the compulsory correction factor that enables WTA to be distinguished between watersheds, which is strongly regulated flows (SRF) and those with no strongly regulated flows.

$$VF = e^{\sqrt{ln(s^{*}_{month})^{2} + ln(s^{*}_{year})^{2}}}$$ Equation 5-65

On the other hand, grid-cell is related to VF (subscript k) combined with watershed level *i*, in order to express the equation of WTA* per watershed *I* (RIVM Report, 2016-0104):

$$VF_{i=}\frac{1}{\sum P_k}\sum_{k=1}^{n} VF_k \cdot P_k$$ Equation 5-66

De Schryver et al. (2011) provide a list with country-based WSI for the three different perspectives, which is available as supporting data in Table 5.48.

Table 5.48: Water Stress Index (WSI) for the three different perspectives (RIVM Report 2016-0104, p. 85)

	Individualist	**Hierarchist**	**Egalitarian**
WSI	0.698	0.657	0.657

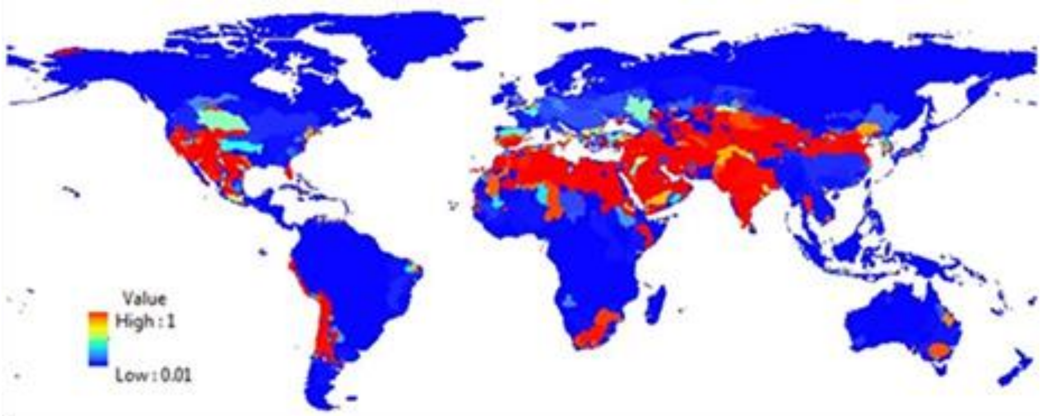


Figure 5.17: Watershed level of Water Stress Index (WSI) (RIVM Report 2016-0104, p. 85)

The calculation of human health damage correlated with malnutrition potentially triggered by water consumption and the equation is (RIVM Report, 2016-0104):

$$CF_{end} = \begin{cases} 1 \cdot CF_{malnutrition} & if\ inventory\ in\ m^3 consumed \\ water\ req.ratio \cdot CF_{malnutrition} & if\ inventory\ in\ m^3\ withdrawn \end{cases}$$ Equation 5-67

The calculation of human health damage in disability adjusted life years (DALYs) due to water-scarcity-related malnutrition for every watershed (or country) *i* and expression equation (RIVM Report, 2016-0104) i:

$$CF_{malnutrition,i} = \underbrace{WSI_i \cdot WU_{\%agriculture,i}}_{\substack{\text{Fate factor} \\ \text{(Water deprivation factor)}}} \cdot \underbrace{\frac{HDF_{malnutrition,i}}{WR_{malnutrition}}}_{\text{Effect factor}} \cdot \underbrace{DF_{malnutrition}}_{\text{Damage factor}}$$ Equation 5-68

Moreover, to express the results of HDF, it is based on (RIVM Report, 2016-0104):

$$HDF_{malnutrition} = \begin{cases} 1 & for\ HDI < 0.30 \\ 2.03 \cdot HDI^2 - 4.09 \cdot HDI + 2.04 & for\ 0.30 \leq HDI \leq 0.88 \\ 0 & for\ HDI > 0.88 \end{cases} \quad \text{Equation 5-69}$$

Country values are available as supporting data in Table 5.49 (RIVM Report, 2016-0104).

Table 5.49: Global average consumption-weighted human health CFs for three different perspectives (RIVM Report 2016-0104, p. 86)

	Individualist	**Hierarchist**	**Egalitarian**
CF (DALY/m^3)	3.10E-06	2.22E-06	2.22E-06

Table 5.50 shows country averages for the Water Stress Index (WSI) and characterisation factors (CFs) for Human Health (HH) in three cultural perspectives (De Schryver et al., 2011; Pfister et al., 2009).

Table 5.50: Country averages for the Water Stress Index (WSI) and characterisation factors (CFs) for Human health (HH) in three cultural perspectives (RIVM Report 2016-0104, p. 179)

Country	**WSI Egalitarian** (-)	**WSI Hierarchist** (-)	**WSI Individualist** (-)	**CF Egalitarian [DALY/m^3]**	**CF Hierarchist [DALY/m^3]**	**CF Individualist [DALY/m^3]**
United Arab Emirates	4.47E+01	4.47E+01	5.49E+01	1.90E-07	1.90E-07	4.38E-07

5.6.10.4. Terrestrial Ecosystems

Direct influences of water consumption on terrestrial ecosystems are correlated with damage for vascular plant species. In addition, net primary productivity (NPP) is included as a proxy for the ecosystem well-being. Moreover, Pfister et al. (2009) state that the characterisation factor is calculated as the sum of water-limited NPP in every pixel k of watershed or country i, which is divided by the sum of grid-specific precipitation, P, as the weighting factor. Furthermore, the fraction of water-availability-limited NPP signifies the vulnerability of an ecosystem to water shortages and it is denoted as a proxy for potentially disappeared fraction (PDF). On the other hand, the unit of the water-limited NPP is dimensionless as a fraction, since precipitation, P, is in m/year by using the unit equal to ($m^3/m^2 \cdot yr$). Moreover, the CF unit is signified as ($m^3/m^2 \cdot yr$), which can be used as ($PDF \cdot m^3/m2 \cdot yr.$). In addition, Pfister et al. (2009) state that PDF omits non-SI units and merely specifies the presence of a fraction, in which the equation expression is (RIVM Report, 2016-0104):

$$CF_i = \frac{\sum_{k=1}^{n} NPP_{water-limited,k}}{\sum_{k=1}^{n} P_k} \qquad \text{Equation 5-70}$$

The equation of water-limited NPP is expressed as:

$$NPP_{water-limited} = ICC_{water} \cdot \left(1 - \frac{ICC_{temperature} + ICC_{radiation}}{2}\right) \qquad \text{Equation 5-71}$$

Supporting information is available for country values in Table 5.52. Moreover, global values used for the area-weighted approach are illustrated in Table 5.51, which shows three cultural perspectives for globally averaged endpoint characterisation factors as impacts of water consumption on terrestrial ecosystems (RIVM Report, 2016-0104).

Table 5.51: Three cultural perspectives for globally averaged endpoint characterisation factors for water consumption on terrestrial ecosystems (RIVM Report 2016-0104, p. 87)

	Individualist	**Hierarchist**	**Egalitarian**
CF [species·yr/m3]	0	1.35E-08	1.35E-08

Table 5.52: Country averages for terrestrial and aquatic ecosystem quality (EQ) in three cultural perspectives (RIVM Report 2016-0104, p. 183)

Country	**CF terrestrial EQ [species-eq·yr/m3]**			**CF aquatic EQ [species eq·yr/m^3]**
	Individualist	**Hierarchist**	**Egalitarian**	**all perspectives**
United Arab Emirates	0	5.06E-08	5.06E-08	0

5.6.10.5. Aquatic Ecosystems

Calculation of water consumption impacts on freshwater fish species is provided for river basins. However, river basins are excluded from higher latitudes as reported species discharge relationships are not available for river basins above 42° latitude (Hanafiah et al. 2011). In addition, characterisation factors are calculated based on marginal variations in river discharge at mouth (dQmouth) of river, because of marginal modification in consumption (dWC) and marginal change of species lost (dPDF) correlated with decline in discharge, since V is the volume of the river basin (RIVM Report, 2016-0104).

$$CF = \frac{dQ_{mouth}}{dWC} \cdot \left(\frac{dPDF}{dQ_{mouth}} \cdot V\right) \quad \left[m^3 \cdot \frac{PDF \cdot yr}{m^3}\right] \qquad \text{Equation 5-72}$$

Moreover, change in freshwater fish species' richness is calculated based on Hanafiah et al. (2011) by using the equation:

$$\frac{dPDF}{dQ_{mouth}} = \frac{0.4}{Q_{mouth}} \left[\frac{PDF \cdot yr}{m^3}\right] \qquad \text{Equation 5-73}$$

In addition, watersheds values are provided in Figure 5.18 and subject to countries, which are available as supporting data in tables 5.53 and 5.54. Finally, CFs in PDF-units are larger for terrestrial ecosystems and the density of average species has two orders of magnitude, which are smaller for aquatic ecosystems. In addition, Table 5.54 illustrates the globally averaged values for water consumption impacts on freshwater ecosystems in watersheds and area-weighted average based on country level (RIVM Report, 2016-0104).

Table 5.53: Globally averaged values for water consumption based on three cultural perspectives (RIVM Report 2016-0104, p. 88)

	Individualist	**Hierarchist**	**Egalitarian**
CF watershed median (species·yr/m^3)	6.04E-13	6.04E-13	6.04E-13
CF area-weighted country Average (species·yr/m^3)	1.74E-12	1.74E-12	1.74E-12

Figure 5.18 shows the map with coverage of watersheds used (Hanafiah et al., 2011). Watersheds above 42° are not included.

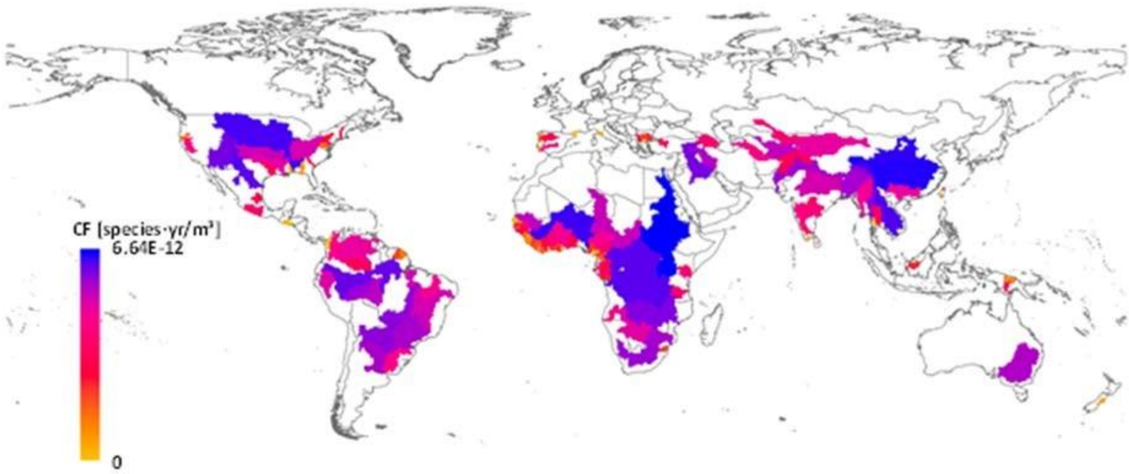


Figure 5.18: Global watershed values (RIVM Report 2016-0104, p. 88)

Moreover, watersheds above 42° latitude from those listed in Table 5.54 are not included. Although, the values are effective for three cultural perspectives. Table 5.54 shows endpoint CFs for impacts of water consumption on aquatic ecosystems, corrected by Hanafiah et al. (2011).

Table 5.54: Endpoint CFs for impacts of water consumption on aquatic ecosystems (RIVM Report 2016-0104, p. 188)

Watershed	**CF [species·yr/m3]**	**Watershed**	**CF [species·yr/m^3]**
NA	NA	NA	NA

5.6.11. Land Use

The impact pathway of land use, as provided in Figure 5.19, contains the direct, local impact of land use on terrestrial species through change of land cover and actual use of new land. Moreover, variation of land impacts consequently on original habitat and original species composition. In addition, land use signifies, agricultural and urban activities, which additionally excludes land as suitable habitat for many species (RIVM Report, 2016-0104).

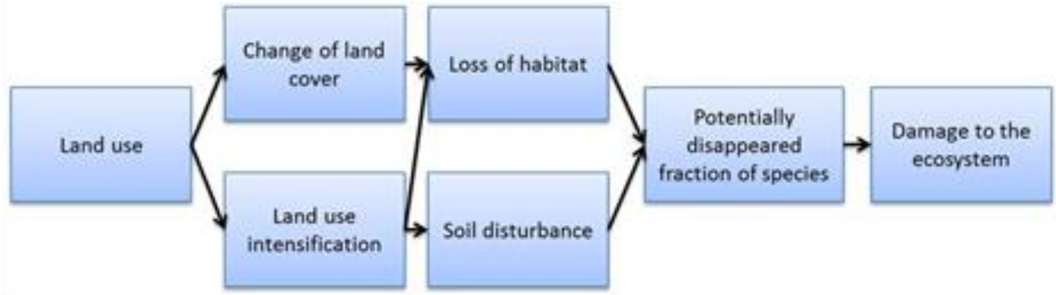


Figure 5.19: Cause-and-effect chain of kand use impact (RIVM Report 2016-0104, p. 89)

Figure 5.19 shows the chain of cause-and-effect of land use, leading to correlated species loss in terrestrial ecosystems. However, indirect pathways of relative species loss to land-use-induced climate change are not included. There are three steps that are different in the land use process, as clarified by Milà i Canals et al. (2007). Firstly, through the conversion phase, the land is made suitable for a new function, such as by removing the original vegetation. Secondly, through the occupation phase, the land is operated for some period. Moreover, both steps are included in CFs for land occupation, which are expressed in Potentially Disappeared Fraction of Species (PDF) per annual crop equivalent. Finally, land is not used, which is a phase of relaxation, when the land may reappear in a semi-natural state. Furthermore, it is predicted that, through the relaxation period, the land has a negative impact on species richness, which does not immediately return to its main habitat or not return to the original habitat in a different state. In addition, CFs for land relaxation are included individually and use a unit of PDF·year per annual crop equivalent. Moreover, Life Cycle Inventory (LCI) data on areas related to land use and duration of land relaxation are multiplied by appropriate CFs and used to express the results of total ecosystem damage (RIVM Report, 2016-0104). Figure 5.20

shows three phases of land use and their impact on land quality, which is provided as species richness.

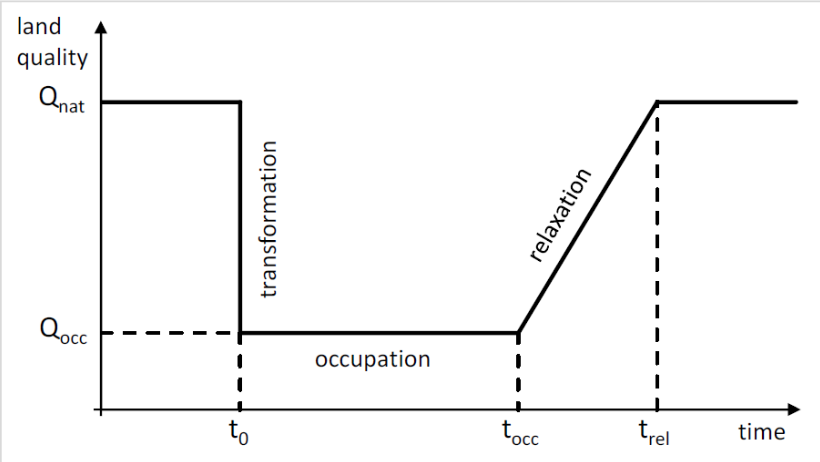


Figure 5.20: Phases of land use and their impact on land quality (RIVM Report 2016-0104, p. 90)

Figure 5.20 shows a schematic approach to the three phases connected with land use and their impact on land quality as stated by Milà i Canals et al. (2007). In addition, land transformation and occupation arises with a range of t_0 and t_{occ}, and relaxation arises with a range of t_{occ} and t_{rel}. Q_{nat} is obviously the original, natural land quality, while Q_{occ} signifies land quality after transformation.

The impact of greenhouse gas emissions on biodiversity is calculated by adopting a methodology available in Climate Change Impact in section 5.6.1 (RIVM Report, 2016-0104).

5.6.11.1. Value Choices

No value choices were provided for calculation of fate CFs for land use (RIVM Report, 2016-0104).

5.6.11.2. Characterisation Factors at Midpoint Level

Characterisation factor of the midpoint (in annual crop equivalents) for land transformation or occupation by using unit of CF_{mocc} signifies the associated species loss, S_{rel}, that is affected by land use type *x* and is proportionate to relative species loss that results from annual crop production by using the following equation (RIVM Report, 2016-0104):

$$CFm_{occ,x} = \frac{S_{rel,x}}{S_{rel,annualcrop}} \qquad \text{Equation 5-74}$$

The calculation of S_{rel} is conducted by comparing field data on local species richness in precise types of natural and human-made land covers, applying a linear relationship described by Köllner et al. (2007) as:

$$S_{rel,x} = 1 - \frac{S_{LU,x,i}}{S_{ref,i}} \qquad \text{Equation 5-75}$$

S_{LU} and S_{ref} are detected species richness (number of species) under land use type x and detected species richness of the reference land cover in region i, respectively. Moreover, outcomes of the above equation are yields ranging from -∞ and +1, since a negative value is defined as a positive influence of land occupation, for example, larger species richness, and a maximum of one signifies 100% loss of species richness. In addition, midpoint characterisation factor is used for land relaxation to semi-natural state CFm_{relax} (in annual crop equivalent·yr), which is correlated to $CF_{mocc,}$ by applying the equation adapted from Köllner et al. (2007).

$$CFm_{relax,x} = CF_{occ,x} \times 0.5 \times t_{rel} \qquad \text{Equation 5-76}$$

Although recovery times are described as independent for land use that changed the natural system, the data collected from Curran et al. (2014), as provided in Table 5.55, shows a global average recovery time calculated as weighted over the total areas of forest and open habitats in the world. In addition, it is assumed as 40% of the global terrestrial area, which is based on forest biomes, and 60% of grassland/shrubland biomes, as elaborated by Olson et al. (2001), in which typical t_{rel} is calculated at 33.9 years.

Table 5.55 clearly shows the midpoint CFs for impact of land transformation/occupation, CFm_{occ}, and land relaxation, CFm_{relax}, on total species richness. Moreover, each CFm_{occ} is correlated with data adapted from De Baan et al. (2013), which is associated with species loss related to different types of land use. Furthermore, recovery time, t_{rel}, is provided in calculation of CFm_{relax} as global average recovery time, which is adapted from Curran et al. (2014).

Table 5.55: Various types of land use based on total species richness (RIVM Report 2016-0104, p. 92)

Land use type	**CFm_{occ} (annual crop eq)**	**CFm_{relax} (annual crop eq·yr)**
Used forest	0.30	5.1
Pasture and meadow	0.55	9.3
Annual crops	1.00	17.0
Permanent crops	0.70	11.9
Mosaic agriculture	0.33	5.6
Artificial areas[1]	0.73	12.4

[1]urban areas, industrial areas, road and rail networks and dumpsites.

Appendix 14 includes supporting data for land use, showing the midpoint CFs for impact of land transformation/occupation (CF_{occ}) by De Baan et al. (2013) and land relaxation (CF_{mrelax}) based on specific species groups. Moreover, recovery times (t_{rel}) are used in the calculation of CF_{mrelax} by Curran et al. (2014).

5.6.11.6. From Midpoint to Endpoint

The calculation of endpoint characterisation factors for transformation/occupation (CFe_{occ}) and relaxation (CFe_{relax}) related to land use type x is provided by using the following equation (RIVM Report, 2016-0104):

$$CFe_{occ,x} = CFm_{occ,x} \times F_{M \to E,LU}$$
$$CFe_{relax,x} = CFm_{relax,x} \times F_{M \to E,LU}$$ Equation 5-77

Since

$$F_{M \to E,LU} = SD_{terr} \times S_{rel,annual\ crops}$$ Equation 5-78

Moreover, FM->E,LU represents the midpoint to endpoint conversion factor (in species/m^2) as shown in Table 5.56. However, SD_{terr} signifies the average species density for terrestrial ecosystems, which is around $1.48 \cdot 10^{-8}$ species/m^2, as per explained by Goedkoop et al. (2009), and shows that $S_{rel,annual}$ crop denotes relative species loss for annual crops, which is 0.60 (annual crop eq^{-1}), which is adapted from De Baan et al. (2013) in Table 5.57. In addition, Table 5.58 includes all endpoint characterisation factors (RIVM Report, 2016-0104).

Table 5.56: Midpoint to endpoint conversion factor (in species/m2) (RIVM Report 2016-0104, p. 94)

	Individualist	**Hierarchist**	**Egalitarian**
Midpoint to endpoint factor	8.88E-09	8.88E-09	8.88E-09

Table 5.57 shows relative species losses (S_{rel}) because of land transformation or occupation, as clarified by De Baan et al. (2013). Moreover, numbers are included for total world averages, although biome or species group-specific S_{rel} are available in the original publication.

Table 5.57: Relative species losses (Srel) of land transformation (RIVM Report 2016-0104, p. 191)

Land Use Type	**S_{rel}**
Pasture and meadow	0.33
Annual crops	0.60
Permanent crops	0.42
Mosaic agriculture	0.2
Artificial areas	0.44

Table 5.58: Endpoint CFs for the impact of land occupation (CFeocc) and land relaxation (CFerelax) on total species richness (RIVM Report 2016-0104, p. 191)

Land Use Type	CFe_{occ} species·/m^2	CFe_{relax} species·yr/ m^2
Used forest	2.66E-09	4.52E-08
Pasture and meadow	4.88E-09	1.51E-07
Annual crops	8.88E-09	8.28E-08
Permanent crops	6.22E-09	1.05E-07
Mosaic agriculture	2.93E-09	4.97E-08
Artificial areas	6.48E-09	1.10E-07

5.6.11.7. Implementation of Land Use in EcoInvent v3.

Net transformation of natural land to anthropogenically correlated to land constitutes, which is one of the main results express of species extinctions. In addition, recently transformed natural land establishes added impact that should be calculated in addition to land occupation effects, which cover the effect when cannot land return to its natural state for an extended period. Moreover, only transformation of natural land is included; land that was converted from one type of anthropogenic use to another is not calculated. Five kinds of natural land are provided in the EcoInvent database, as shown in Table 5.59. Furthermore, converted from kind of natural land constitutes results in impacts on the ecosystem, while conversion to one of land benefits the ecosystem, such as negative CFs. In addition, conversion to primary forest is not applicable (RIVM Report 2016-0104, p. 191). Characterisation factors are calculated by using the equation:

$$CF_{trans,i} = 0.5 * CF_{occ,max} * T_{rec,i} \quad \text{Equation 5-79}$$

The CF of kind of land conversion (in annual crop equivalents) i is expresses the calculation as a function of maximum occupation CF (in 1 annual crop equivalent) and Trec,i denotes recovery time of land type i (in 73.5 years for forest and 7.5 years for open land). Furthermore, the equation establishes the area under curve, which ensues from a linear conversion of anthropogenic land that is occupied back to a normal state. Moreover, conversion to any kind of land is calculated as a negative equivalent of CF conversion to confirm the provided effects of net natural land transformation (RIVM Report, 2016-0104).

Table 5.59: Midpoint CFs for natural land transformation (RIVM Report 2016-0104, p. 192)

Name	**Midpoint CF (annual crop equivalents·yr)**
Transformation, from grassland, natural (non-use)	3.75
Transformation, from forest, primary (non-use)	36.75
Transformation, from forest, secondary (non-use)	36.75
Transformation, from shrub land, sclerophyllous	3.75
Transformation, from wetland, inland (non-use)	3.75
Transformation, to shrub land, sclerophyllous	-3.75
Transformation, to forest, secondary (non-use)	-36.75
Transformation, to wetland, inland (non-use)	-3.75
Transformation, to grassland, natural (non-use)	-3.75

Midpoint indicators are calculated in the same units as midpoints for land occupation, while endpoint impact calculation includes multiplying the same factor of midpoint to endpoint. Furthermore, EcoInvent separates 30 categories of land occupation in a related inventory. Moreover, it is suggested that different categories are applicable to CFs from ReCiPe, as provided in Appendix 15, which shows applicable land occupation categories in EcoInvent with ReCiPe (RIVM Report, 2016-0104).

5.6.12. Mineral Resource Scarcity

The damage modelling for the impact category of mineral resource scarcity is subdivided into a number of steps, as shown in Figure 5.21. In addition, primary extraction of a Mineral Resource (ME) leads to decrease in Ore Grade (OG), which influences concentration, which is related to the resource in global ores, since it increases ore produced per kilogram of Mineral Resource Extracted (OP). Furthermore, joined predicted future extraction of mineral resource leads to average Surplus Ore Potential (SOP), which represents the midpoint indicator for the impact category. Moreover, an increase in Surplus Ore Potential (SOP) leads to surplus cost potential, while both two indicators follow the standard of mining sites with more grades or less costs for SOP and SCP respectively. Moreover, calculation of damage to natural resource scarcity is included (RIVM Report, 2016-0104). Figure 5.21 shows the cause-and-effect chain, from mineral resource extraction to natural resource scarcity.

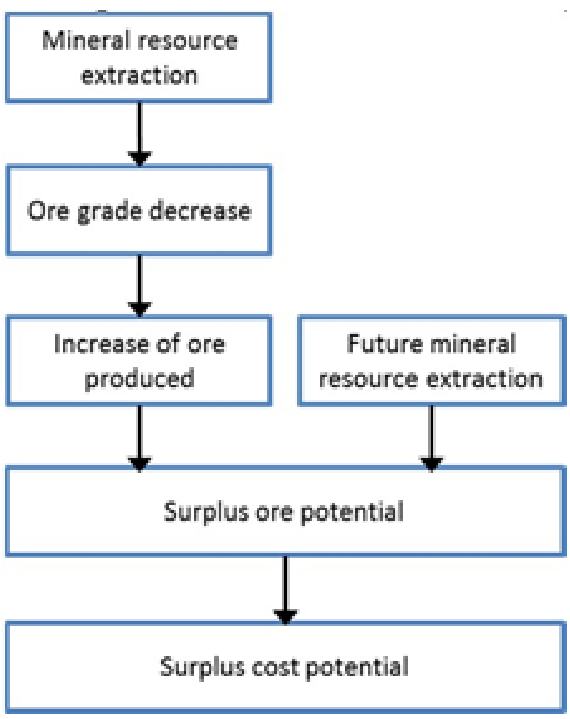


Figure 5.21: Cause-and-effect chain of mineral resource scarcity (RIVM Report 2016-0104, p. 95)

5.6.12.1. Value Choices

Value choice for future extraction of mineral resources influences modelling of the midpoint and endpoint mineral resource that correlates to scarcity (Vieira et al. 2016a,b). In addition, calculation of characterisation factors of two different reserve estimates is applied. The first type is Reserves (R), which signifies the part of a resource which at the time of determination might be economically extracted or produced at that illustrated the definition of current prices and technology state (U.S. Geological Survey, 2014). However, the three different cultural perspectives categorise value choices as modelling of the effect of extracting mineral resources, as illustrated in Table 5.60 (RIVM Report, 2016-0104).

Table 5.60: Cultural perspectives categorisation of value choices for extracting mineral resources (RIVM Report 2016-0104, p. 96)

Choice category	**Individualist**	**Hierarchist**	**Egalitarian**
Future production	Reserves	Ultimate recoverable resource	Ultimate recoverable resource

5.6.12.2. Characterisation Factors at Midpoint Level

The characterisation factor of the midpoint of any mineral resource, x, and any reserve, denoted as Rx, is calculated by using the equation (RIVM Report, 2016-0104):

$$\mathrm{SOP}_{x,\mathrm{R}} = \frac{\mathrm{ASOP}_{x,\mathrm{R}}}{\mathrm{ASOP}_{\mathrm{Cu,R}}} \qquad \text{Equation 5-80}$$

Yield signifies the Future Production Specific (SOP) by using the unit of (kg Cu-eq/kg x).

ASOP consists of two calculation steps, as stated by Vieira et al. (2016b). The first step counts the cumulative grade-tonnage relationship that is derived as elaborated in the copper example in Figure 5.22, while the expression equation of the ORE grade of mineral resource *x* (as per Vieira et al. 2012) is:

$$\mathrm{OG}_x = \exp(\alpha_x) \cdot \exp\left(\beta_x \cdot \ln\left(\frac{\mathrm{A}_x - \mathrm{CME}_x}{\mathrm{CME}_x}\right)\right)$$ Equation 5-81

Moreover, the table in Appendix 16 shows more data about parameters alpha and beta for every covered mineral resource, while the data is used to derive ASOPs adopted from the calculation of SOPs (Vieira et al. 2016b) as supporting data for mineral resource scarcity. Figure 5.22 shows the cumulative grade-tonnage relationship for existing copper mines plotted by using log-logistic regression in logarithmic scale.

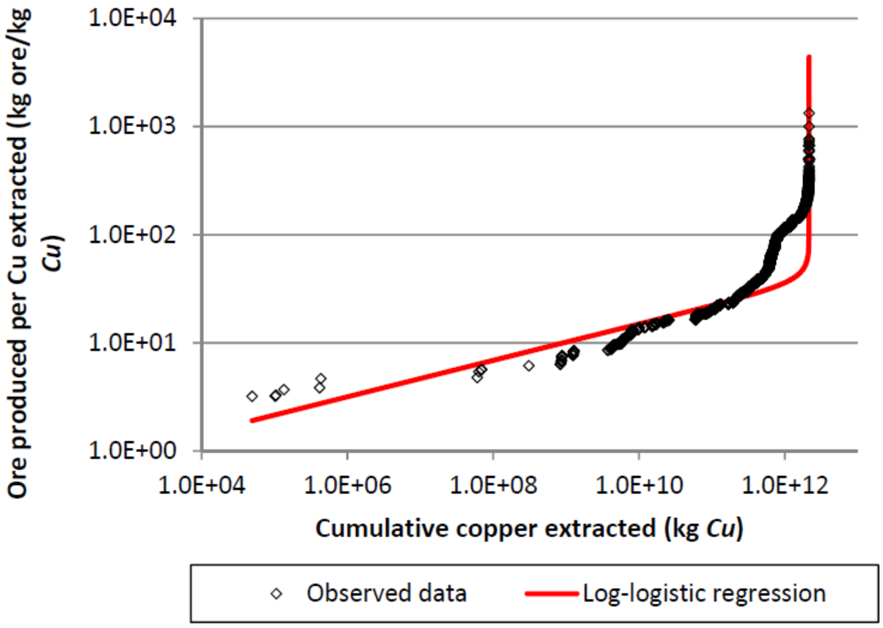


Figure 5.22: Cumulative grade-tonnage relationship for existing copper mines (RIVM Report 2016-0104, p. 97)

Absolute Surplus Ore Potential of mineral *x* $ASOP_x$ by using unit (kg ore/kg x), which signifies the extra quantity of ORE that can be formed in the future per unit of extracted mineral resource *x*, which is calculated by using the equation provided by Vieira et al. (2016b):

$$\mathrm{ASOP}_x = \frac{\int_{\mathrm{CME}_x}^{\mathrm{MME}_x} (\Delta \mathrm{OP}_x)\, d\mathrm{ME}_x}{\mathrm{R}_x} \qquad \text{Equation 5-82}$$

Figure 5.23 elaborates the relationship between average price in 2013 (USD1998/kg x) and absolute surplus ore potential (kg ore/kg x).

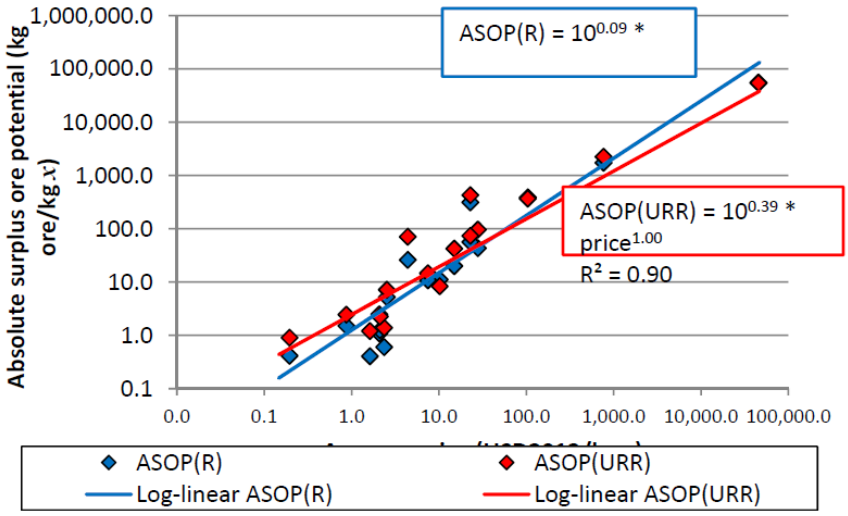


Figure 5.23: Price of a mineral that is measured as good predictor for ASOP with adjusted regression range between 90-91% (RIVM Report 2016-0104, p. 98)

Appendix 17 shows the midpoint characterisation factors' SOPs (kg Cu-eq/kg), which represent three perspectives for 70 mineral resources and for groups of garnets, gemstones, platinum-group metals, rare-earth metals and zirconium minerals (RIVM Report, 2016-0104).

5.6.12.3. From Midpoint to Endpoint

The calculation equation of endpoint characterisation factors (CFe) for extraction of mineral resource x and the cultural perspective c (RIVM Report, 2016-0104):

$$\mathrm{CF}e_{x,c} = \mathrm{SOP}_{x,c} \times \mathrm{F}_{\mathrm{M}\rightarrow\mathrm{E},c} \qquad \text{Equation 5-83}$$

c denotes the cultural perspective, SOPx, and c is the midpoint characterisation factor in kg Cu-eq/kg x.

$$\mathrm{F}_{\mathrm{M}\rightarrow\mathrm{E},c} \qquad \text{Equation 5-84}$$

The equation of the midpoint to endpoint factor for mineral resource scarcity is equivalent to the copper endpoint characterisation factor by using the following equation, as calculated by RIVM Report (2016-0104):

$$\mathrm{F}_{\mathrm{M}\rightarrow\mathrm{E},c} = \mathrm{ASOP}_{Cu,c} \times \frac{\mathrm{ASCP}_{x,c}}{\mathrm{ASOP}_{x,c}} \qquad \text{Equation 5-85}$$

However, the last part of the midpoint to endpoint calculation factor for mineral resource scarcity at every cultural perspective c is expressed by using this equation (RIVM Report, 2016-0104):

$$\frac{\mathrm{ASCP}_{x,c}}{\mathrm{ASOP}_{x,c}} = \frac{10^{a_{\mathrm{ASCP},c}} \times price^{b_{\mathrm{ASCP},c}}}{10^{a_{\mathrm{ASOP},c}} \times price^{b_{\mathrm{ASOP},c}}} \approx \frac{10^{a_{\mathrm{ASCP},c}}}{10^{a_{\mathrm{ASOP},c}}} \qquad \text{Equation 5-86}$$

Data is used to derive the midpoint to endpoint factors by using the equation of Absolute Surplus Cost Potential (ASCP) of mineral resource x calculated by Vieira et al. (2016a):

$$\mathrm{ASCP}_x = \frac{\int_{\mathrm{CME}_x}^{\mathrm{MME}_x} (\Delta \mathrm{C}_x)\, d\mathrm{ME}_x}{\mathrm{R}_x} \qquad \text{Equation 5-87}$$

Finally, log-logistic distribution is applicable for inverse operating costs per extracted mineral resource and extracted cumulative mineral resource, as calculated by Vieira et al. (2016a):

$$\frac{1}{\mathrm{C}_x} = \exp(\alpha_x) \cdot \exp\left(\beta_x \cdot \ln\left(\frac{\mathrm{MME}_x - \mathrm{CME}_x}{\mathrm{CME}_x}\right)\right) \qquad \text{Equation 5-88}$$

aASCP and bASCP are objects and slopes for the log-linear function between ASCP and price of every mineral resource, which is available in supporting information, figures 5.25 and 5.26. Moreover, aASOP and bASOP are objects and slopes for the log-linear function between ASOP and price of every mineral resource, as elaborated in Figure 5.24. In addition, all b slopes range from 0.97-1.08, which results in a factor that does not exceed 4 if the price is not included in the equation (RIVM Report, 2016-0104).

Furthermore, the average price of every mineral resource in 2013 is included. Table 5.61 shows three cultural perspectives of midpoint to endpoint factors for mineral resource scarcity. In addition, midpoint to endpoint factors equal the potential of absolute surplus cost in unit of USD2013/kg Cu, which is expressed for copper based on Vieira et al. (2016a).

Table 5.61: Cultural perspectives of midpoint to endpoint factors for mineral resource scarcity (RIVM Report 2016-0104, p. 101)

	Unit	Individualist	Hierarchist	Egalitarian
Midpoint to endpoint factor	USD2013/kg Cu	0.16	0.23	0.23

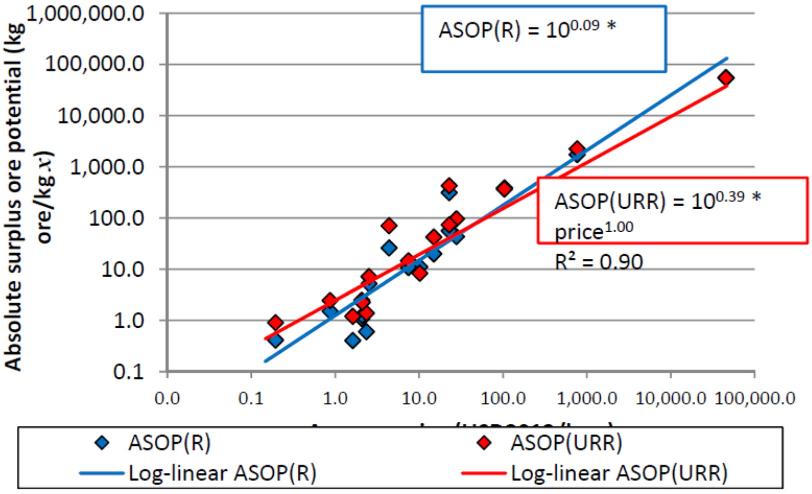


Figure 5.24: Log-linear function between ASOP and price of every mineral resource (RIVM Report 2016-0104, p. 98)

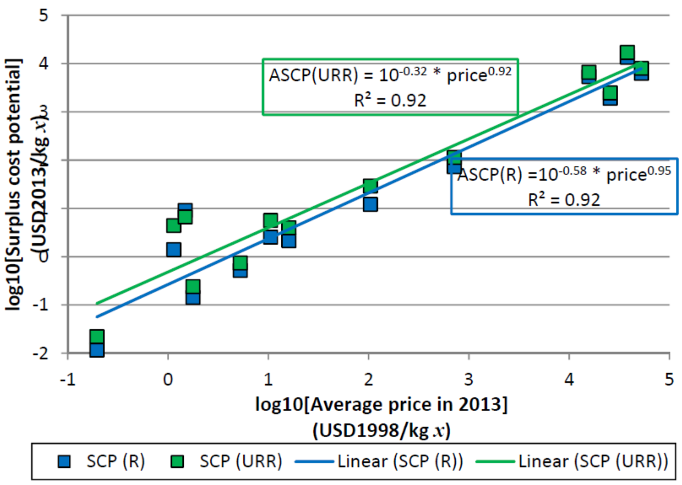


Figure 5.25: Relationship between average price in 2013 (USD1998/kg x) and absolute surplus cost potential (USD2013/kg x) (RIVM Report 2016-0104, p. 195)

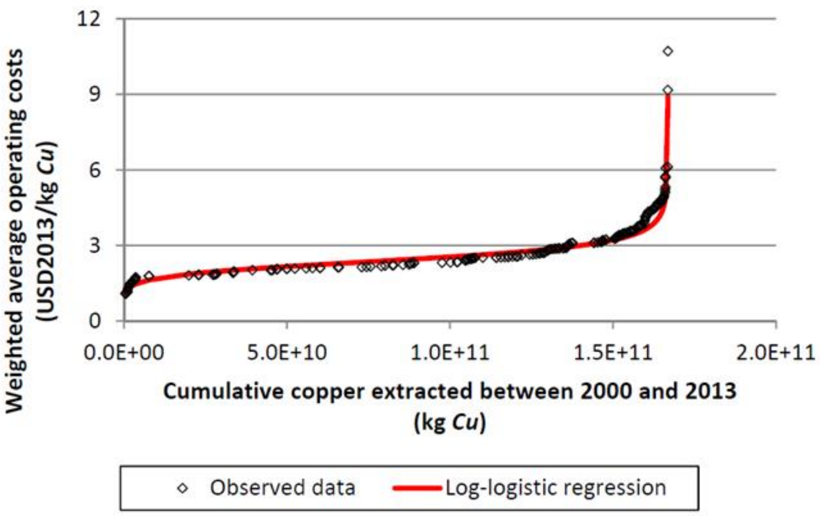


Figure 5.26: Cumulative cost-tonnage relationship for copper plotted using a log-logistic regression in logarithmic scale (RIVM Report 2016-0104, p. 195)

5.6.13. Fossil Resource Scarcity

The impact category of fossil resource scarcity has damage modelling, which is subdivided into a number of steps, as illustrated in Figure 5.27. Moreover, it is predicted in modelling of the endpoint that low-cost fossil fuels are extracted first. In addition, an increase in fossil fuel extraction has an influence on costs due to change in production technique or sourcing from a higher location. For instance, when there is a depletion of all conventional oil, other techniques, for example, enhanced oil recovery, are used or oil is produced in different geographical locations with higher costs, for example, Arctic regions (Ponsioen et al., 2014). On the other hand, combining the expected future extraction of fossil resources results in surplus cost potential (SCP), which represents the endpoint indicator for the impact category, although estimation of damage to natural resource scarcity is included. Moreover, fossil fuel potential with higher heating value is used as a midpoint indicator (RIVM Report, 2016-0104). Figure 5.27 shows the cause-and-effect chain from fossil resource extraction to natural resource scarcity.

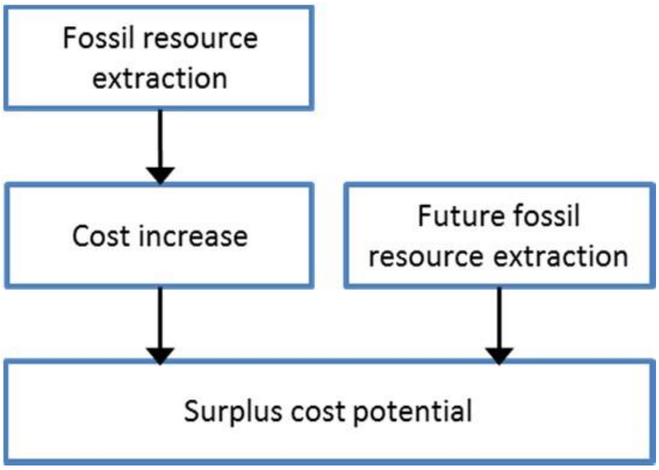


Figure 5.27: Cause-and-effect chain of fossil resource scarcity impact (RIVM Report 2016-0104, p. 103)

5.6.13.1. Value Choices

No value choices are provided for fossil resources at midpoint and endpoint modelling (RIVM Report, 2016-0104).

5.6.13.2. Characterisation Factors at Midpoint Level

The midpoint indicator for fossil resource use, determined as the Fossil Fuel Potential of fossil resource x (kg oil-eq/unit of resource), is defined as the ratio between the energy content of fossil resource x and the energy content of crude oil, which is calculated by (RIVM Report, 2016-0104):

$$\mathrm{FFP}_x = \frac{\mathrm{HHV}_x}{\mathrm{HHV}_{oil}} \qquad \text{Equation 5-89}$$

Fossil fuel potential (FFP) that uses units in kg oil-eq/unit of resource is correlated with higher heating value (HHV) of five fossil resources: crude oil, natural gas, hard coal, brown coal and peat, as elaborated in Table 5.76. In addition, HHV is used similar to one provided in the EcoInvent database (Jungbluth & Frischknecht, 2010) as elaborated in the table in Appendix 18.

The table in Appendix 19 shows the data of cumulative cost-tonnage parameters a and b for every fossil resource, which is the calculated SCPs and supporting data for the derived FFPs (Vieira & Huijbregts, in prep.).

5.6.13.3. Characterisation Factors at Endpoint Level

Endpoint characterisation factors (CFe) for extraction of fossil resource x, signified as Surplus Cost Potential (SCP), by using the equation to express the results, as stated by Vieira & Huijbregts, in prep.):

$$\mathrm{CF}_e = \mathrm{SCP}_x = \frac{\int_{\mathrm{CFE}}^{\mathrm{MFE}} (\Delta \mathrm{C}_x)\, d\mathrm{FE}_x}{\mathrm{R}_x} \qquad \text{Equation 5-90}$$

In addition, cost equation of fossil resource x is calculated based on Vieira et al. (2016b):

$$\mathrm{C}_x = \frac{1}{a_x + b_x \cdot \ln(\mathrm{CFE}_x)} \qquad \text{Equation 5-91}$$

C_x represents production cost of fossil resource *x* in USD/kg or Nm^3 x; CFE_x in kg or Nm^3 x signifies the cumulative quantity of extracted fossil resource *x*; while a_x and b_x are the intercept and slope of the log-linear distribution of cumulative cost-tonnage with correlation of fossil resource *x*, as elaborated in Figure 5.28, in which the cumulative cost-tonnage relationship for crude oil is plotted by using log-linear regression (x-axis in logarithmic scale). Appendix 20 shows the five fossil resources for endpoint characterisation factors in USD2013/unit of resource.

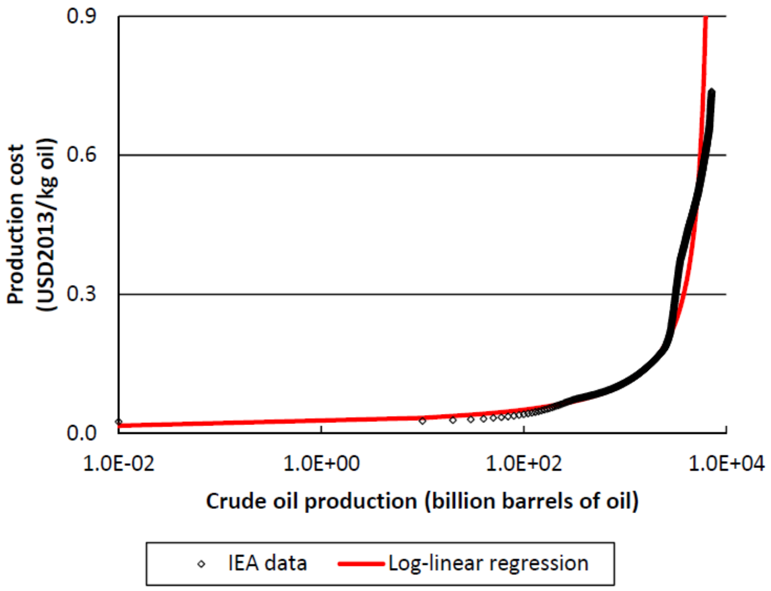


Figure 5.28: Cumulative cost-tonnage relationship for crude oil (RIVM Report 2016-0104, p. 105)

Moreover, the table in Appendix 21 shows the provided substances and weights per every substance group. Additionally, no characterisation factors are included for the group and compartment if no emissions are available for the specific compartment. Although characterisation factors for sub-compartments associated with weights respective compartments as provided in the tables in Appendix 22, which include weighted average factors per substance group and sub-compartment for three perspectives, Individualist, Hierarchist and Egalitarian. In addition, nine midpoint categories of some substance groups have CFs. On the other hand, characterisation factors for emissions to air are not calculated precisely for urban or rural air for midpoint categories of Global Warming, Ozone Depletion, Ionising Radiation and Photochemical Ozone Formation. Moreover, the equivalent factor for unspecified air is included twice in the tables, as detailed in Appendix 22 (RIVM Report, 2016-0104).

5.7. Summary

To conclude this chapter, LCA phases of this study were discussed separately with the data collection process, although the modelling framework shows the flow chart of the research processes in the stages which will be followed. Furthermore, the data calculations and analysis of ReCiPe2016 LCIA was elaborated in detail with calculation of every impact, which consists of 18 midpoint impact categories, nine damages pathways and three endpoint areas of protection. Moreover, the calculations of ReCipe2016 show how it works in SimaPro software to generate the final LCA results. In addition, this chapter provides the phases used as per ISO14040 methodology to conduct LCA on concrete waste management. Furthermore, all equations and data provided in this chapter will be used in SimaPro 8.5.2 software, which helps to correlate all inventories and calculate the inventory data of concrete waste in the software itself to obtain the all LCA results. In Chapter 6, the generated results and analysis of LCIA will be discussed.

CHAPTER 6: ENVIRONMENTAL IMPACT AND DAMAGE COST RESULTS OF CONCRETE WASTE

6.1. Introduction

This chapter presents the third phase of LCA as per ISO14040:2006, which is Life Cycle Impact Assessment (LCIA), as data findings and analysis of the midpoint and endpoint assessment method, which is based on 18 midpoint impact categories, nine damage pathways and three endpoint area of protection. In addition, the results of the environmental impact assessment using ReCipe2016 are for unreinforced concrete waste, which is related to the research aim and objectives on selected waste management options, which are final disposal method and recycling method in parallel with the transportation method for concrete waste. Furthermore, the assessment results cover all ReCipe2016 of LCIA in terms of midpoint and endpoint. Moreover, extraction of inventory data for all relevant impacts such as landfilling, recycling and transportation is from the EcoInvent database. Additionally, LCIA assessment calculations and results are obtained from SimaPro 8.5.2 software, which is used as a tool to correlate and simplify the results. On the other hand, the chapter presents the results of damage cost and its relationship with the LCA results. Moreover, the monetisation of environmental indicators used damage costs and all the environmental prices are based on Handbook Environmental Prices (2017), which includes 15 impact prices that are applicable for the selected LCIA of this study, which is ReCiPe2016. In addition, LCA and damage cost assessment were conducted for 2018, while the results for the years from 2013 to 2017 are available in appendices 4 and 5.

6.2. Environmental Prices

The damage costs were used by referring to Handbook Environmental Prices 2017, which includes 15 prices that are applicable for LCIA by ReCiPe2016. In addition, Table 6.1 illustrates the impact category price in euros per related unit and the reference page number linked to the Handbook Environmental Prices 2017.

Table 6.1: Environmental prices (Handbook Environmental Prices, 2017)

Handbook Environmental Prices 2017				
No.	**Impact category**	**Unit**	**Price in Euros (€)**	**Page**
1	Climate change	€/kg CO2-eq.	0.057	42
2	Ozone layer depletion	€/kg CFC-eq.	30.4	42
3	Acidification	€/kg SO2-eq.	5.4	42
4	Freshwater eutrophication	€/kg P-eq.	1.9	42
5	Marine eutrophication	€/kg N	3.11	42
6	Land use	€/m2a	0.0261	42
7	Terrestrial ecotoxicity	€/kg 1,4 DB-eq.	8.89	42
8	Freshwater ecotoxicity	€/kg 1,4 DB-eq.	0.0369	42
9	Marine ecotoxicity	€/kg 1,4 DB-eq.	0.00756	42
10	Human toxicity	€/kg 1,4 DB-eq.	0.214	42
11	PM2.5	€/kg PM2.5 eq.	79.5	100
12	Nitrogen oxides (Nox) (Human health)	€/kg NOx eq.	18.7	107
13	Nitrogen oxides (Nox) (Terrestrial ecosystems)	€/kg NOx eq.	18.7	107
14	Mineral resource scarcity (Atmospheric)	$/kg Cu eq	4.2	165
15	Mineral resource scarcity (Soil)	$/kg Cu eq	0.239	173

6.3. LCA and Damage Cost

The calculation to link the LCA results with damage cost results is made by referring to the Handbook Environmental Prices 2017, as shown in Table 6.1. Furthermore, the assessment of LCA results begins with the LCIA phase by using the ReCipe2016 method. In addition, the EcoInvent database was used for the second phase of LCA, which is life cycle inventory. However, the third stage of LCA is LCIA, which is to find out the results, and this stage was conducted by using SimaPro software, which helps to simplify and categorise the final result of the complete LCA. In addition, based on Table 6.1, the damage prices were used for each year to find out the damage cost besides the LCIA results and its cost influence on the environment.

Table 6.2 illustrates the quantity of concrete waste in Dubai landfill from 2013 to 2018, which shows that it is increasing yearly. Furthermore, midpoint LCIA and damage cost were conducted for the year 2018. Data for the years 2017 to 2013 is available in appendices 4 and 5, and is based on Table 6.2 data and by referring to the same inventory as calculated for the year 2018.

Table 6.2: Quantity of concrete waste in Dubai landfill in each year separately (Public Source, 2019)

Year	**Normal Concrete Waste from C&DW going into Landfill**	**Tonnes**
2013	"	3,900,000
2014	"	4,150,000
2015	"	4,400,000
2016	"	4,700,000
2017	"	5,000,000
2018	"	5,400,000

6.4. Midpoint LCIA by ReCipe2016

The midpoint method is positioned at the level of somewhere along with impact pathway approach, which is at the point where the environmental mechanism is equal to that allocated to that specific impact category (Goedkoop et al., 2009). In addition, the midpoint method has a robust relation to ecological flows and comparatively less uncertainty.

Figure 6.1 shows the results of the third phase of LCA by using the ReCipe2016 method that includes midpoint assessment. In addition, Figure 6.1 elaborates each midpoint impact result in percentage for three inventories, which are concrete waste in the landfill as final disposal, recycling of concrete waste and transportation process. It indicates that waste concrete in landfill does more damage to the environment compared to the recycling method and transportation process in most of the midpoint impacts, for example, Global Warming, Stratospheric Ozone Depletion, Ionising Radiation, Ozone Formation (Human Health), Fine Particulate Matter Formation, Ozone Formation (Terrestrial Ecosystems), Terrestrial Acidification, Freshwater Eutrophication, Marine Eutrophication, Freshwater Ecotoxicity, Marine Ecotoxicity, Land Use, Mineral Resource Scarcity, Fossil Resource Scarcity and Water Consumption. In contrast, the transportation process of concrete waste had the highest influence on the environment in only some midpoint impacts, which are Human Carcinogenic Toxicity, Human Non-Carcinogenic Toxicity and Terrestrial Ecotoxicity. On the other hand, some midpoint impacts of the recycling method, such as Ozone Formation (Human Health), Fine Particulate Matter Formation and Ozone Formation (Terrestrial Ecosystems), were found to be higher than the transportation process only. In the overall perspective, the landfilling method for concrete waste had the highest impacts in midpoint assessment compared to the recycling method and transportation process.

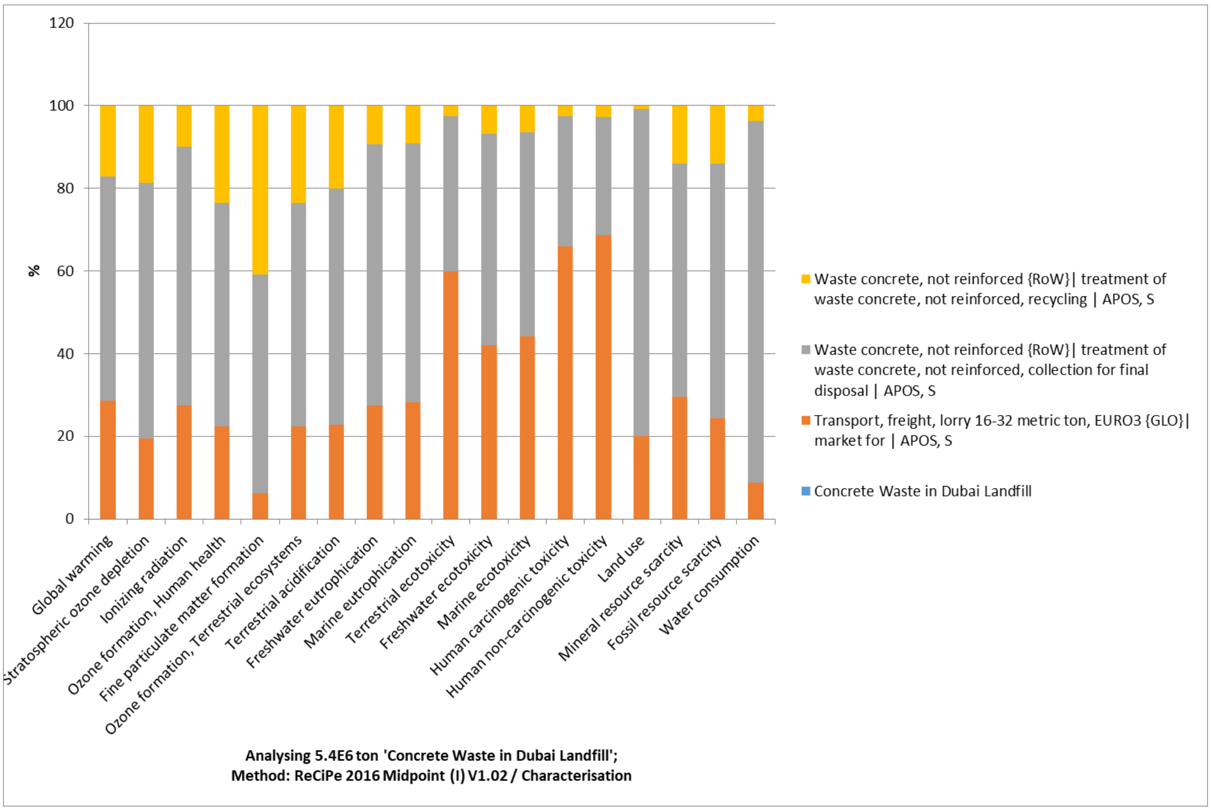


Figure 6.1: Results of each midpoint impact in percentage (SimaPro software, ReCiPe2016 Midpoint Method)

Figure 6.1 shows the results of 15 impacts of midpoint LCIA and related damage cost. From an overall perspective, the landfilling method and transportation process had a high midpoint damage cost compared to the recycling method, especially in Terrestrial Ecotoxicity, Ozone Formation (Human Health), Fine Particulate Matter Formation and Global Warming. Marine Eutrophication and Marine Ecotoxicity had lower damage cost midpoint impacts, which shows that they have less impact compared to the others. Midpoint LCIA and related damage cost results are available for the years 2013 to 2017 in appendices 4 and 5.

6.4.1. Fine Particulate Matter Formation

Airborne Particulate Matter (PM) is a combination of particles such as liquids or solids, which vary in size and composition. The effects of airborne particulates contribute to damage to human health and buildings, and cause visual inconvenience, in the form of haze. In addition, Fine Particulate Matter transport has a large variety of toxic substances, which are directly released into the air passages and lungs. Moreover, the smaller particles are more dangerous, as they can penetrate the deepest into the lungs and can cause both direct and future damage (Handbook Environmental Prices, 2017).

The LCA results in Figure 6.1 show that waste concrete in landfill as final disposal has a 52.9% impact on Fine Particulate Matter Formation compared to transportation and recycling method, which are 6.22% and 40.8% respectively. From a unit perspective, waste concrete in the landfill as final disposal was 148,450.38 (kg PM2.5 eq) compared to transportation and recycling method, which were 17,440.96 and 114,384.59 respectively. The reason that landfilling and recycling methods have more impact on Fine Particulate Matter Formation is due to the dust created during both methods' process and operation. Furthermore, the damage pathway of Tropical Ozone Formation (Human Health) is increased in respiratory disease, which leads to damage to human health as the endpoint area of protection.

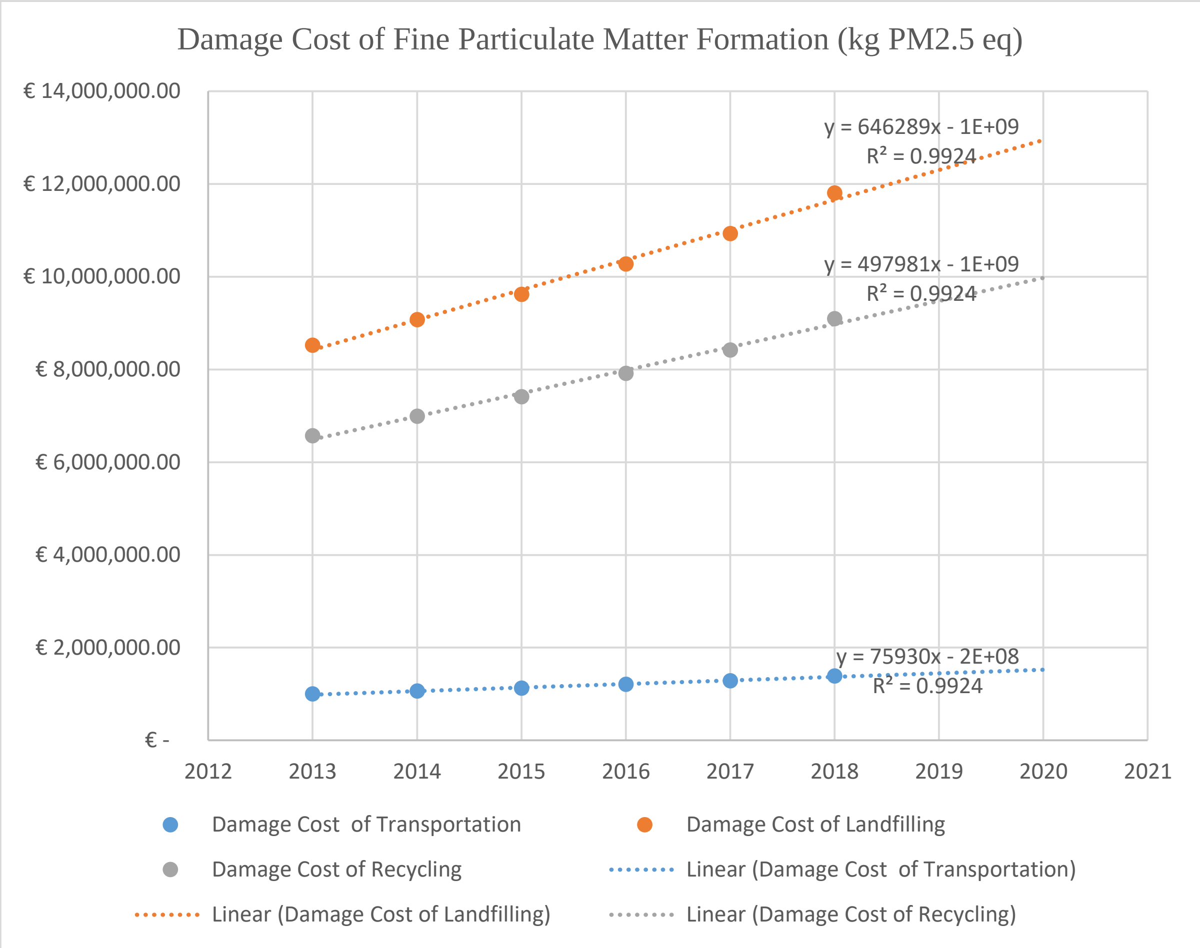


Figure 6.2: Damage Cost of Fine Particulate Matter Formation

Figure 6.2 presents the results of the damage cost for three methods from the year 2013 to 2018, in which R^2 = 0.9924. In addition, the damage cost forecast for the three methods shows that it will keep increasing in the future, almost reaching €13,500,000 in 2020, particularly in regard to the landfilling method. In addition, the damage cost of the three methods in Fine Particulate Matter Formation from the years 2013 to 2018 is increasing by around €60,000 to €100,000 as the yearly average in the transportation of concrete waste by lorry. The damage cost of both methods, landfilling and recycling of concrete waste, has increased by an average €50,000 to €100,000 every year.

6.4.2. Tropical Ozone Formation (Human Health)

Ozone Depleting Substances (ODSs) lead to an increase in UVB-radiation, which results in damage to human health. Although, when ozone increases it leads to a decrease in atmospheric ozone concentration, which results in a larger radiation portion of UVB hitting the Earth, more radiation has a negative impact on human health, which leads to an increase in the occurrence of skin cancer and cataracts (RIVM Report, 2016-0104).

LCA results in Figure 6.1 demonstrate that the landfill method has an influence of 54.1% on Ozone Formation (Human Health), which is more than transportation and recycling method at 22.3% and 23.6% respectively. Moreover, waste concrete in landfill was 601,163.86 (kg NOx eq), while transportation and recycling were 248,111.96 and 262,075.62 respectively. Although, the damage pathway of ReCipe2016 shows that Tropical Ozone Formation (Human Health) increases respiratory disease, which results in final endpoint in damage to human health.

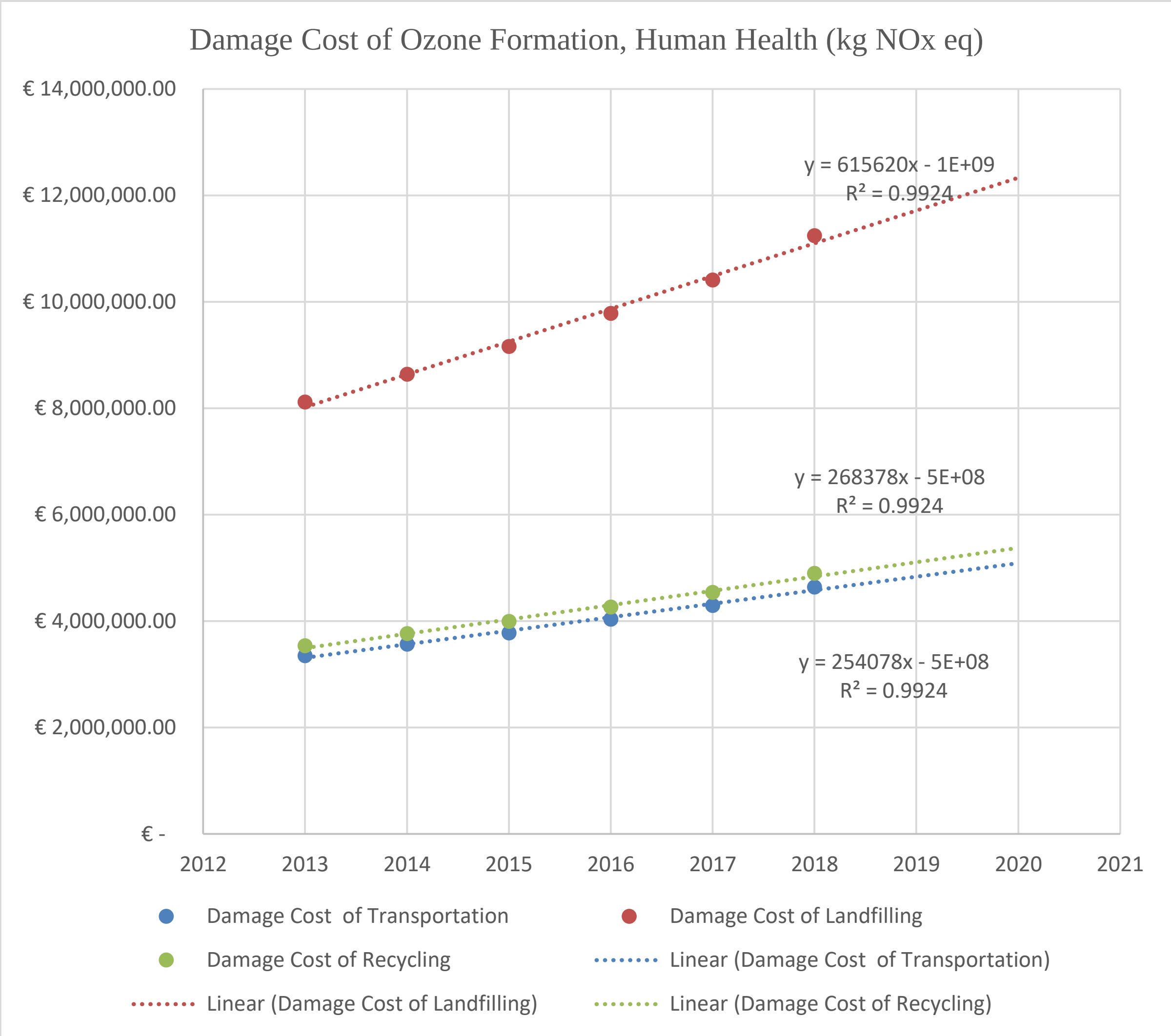


Figure 6.3: Damage Cost of Ozone Formation, Human health

Figure 6.3 illustrates the results from the years 2013 to 2018 for the damage cost for the three methods related to the impact of Ozone Formation (Human Health). The results show an increase on average of €200,000 to €900,000 yearly in all three methods, in which R^2 = 0.992. Moreover, the projection for the three methods of damage cost shows accumulation in future, and mostly in the landfilling method.

6.4.3. Ionising Radiation

The generation of anthropogenic emissions of radionuclides occurs in the fuel cycle (mining, processing and waste disposal), although it is also generated during other human activities, for instance, burning of coal and the extraction of phosphate rock. First, the model of dispersion of the radionuclide throughout the environment is included. In addition, an exposure model follows the step which radiation amount such as effective collective dose is received by determination of entire population. On the other hand, exposure to ionising radiation can be affected by radionuclides, which lead to damaged DNA molecules. Incidence of non-fatal cancers and the incidence of fatal cancers are affected during the analysis step and distinguished from impacts of severe hereditary and it is weighted in order to calculate the damage to human health in disability adjusted life years (DALY) as a final step. Currently, it is not possible to quantify the damage to ecosystems by ionising radiation due to the unavailability of impact assessment methodologies (RIVM Report, 2016-0104).

Figure 6.1 for LCA results proves that accumulated waste concrete in landfill as a final disposal has a 62.5% influence on Ionising Radiation compared to transportation process and recycling method, which are 27.5% and 9.97% respectively. From a unit viewpoint, the landfilling method was 1,3677,16.3 (kBq Co-60 eq), and transportation and recycling method were 602,350.37 and 218,163.46 respectively. Moreover, the Ionising Radiation damage pathway is increased in different types of cancer and growth in other diseases or causes which influence damage to human health.

The damage price of Ionising Radiation is not available in the Handbook Environmental Prices 2017 or other resources, so calculation of damage cost was not conducted.

6.4.4. Stratospheric Ozone Depletion

Ozone Depleting Substances (ODSs) lead to an increase in UVB-radiation, which results in damage to human health. Although, when ozone increases it leads to a decrease in atmospheric ozone concentration, which results in a larger radiation portion of UVB hitting the Earth, more radiation has a negative impact on human health, which leads to an increase in the occurrence of skin cancer and cataracts (RIVM Report, 2016-0104).

Midpoint LCA results in Figure 6.1 show that the waste concrete in landfill was 61.7% for Stratospheric Ozone Depletion, while transportation and recycling method results were at 19.5% and 18.8% respectively. From a unit perspective, the result for waste concrete in landfill is 29.765 (kg CFC11 eq), while transportation and recycling methods are 9.407 and 9.0504113 respectively. The impact of Stratospheric Ozone Depletion increases numerous kinds of cancer and an increase in other diseases or causes which lead to damage to human health as the endpoint area of protection.

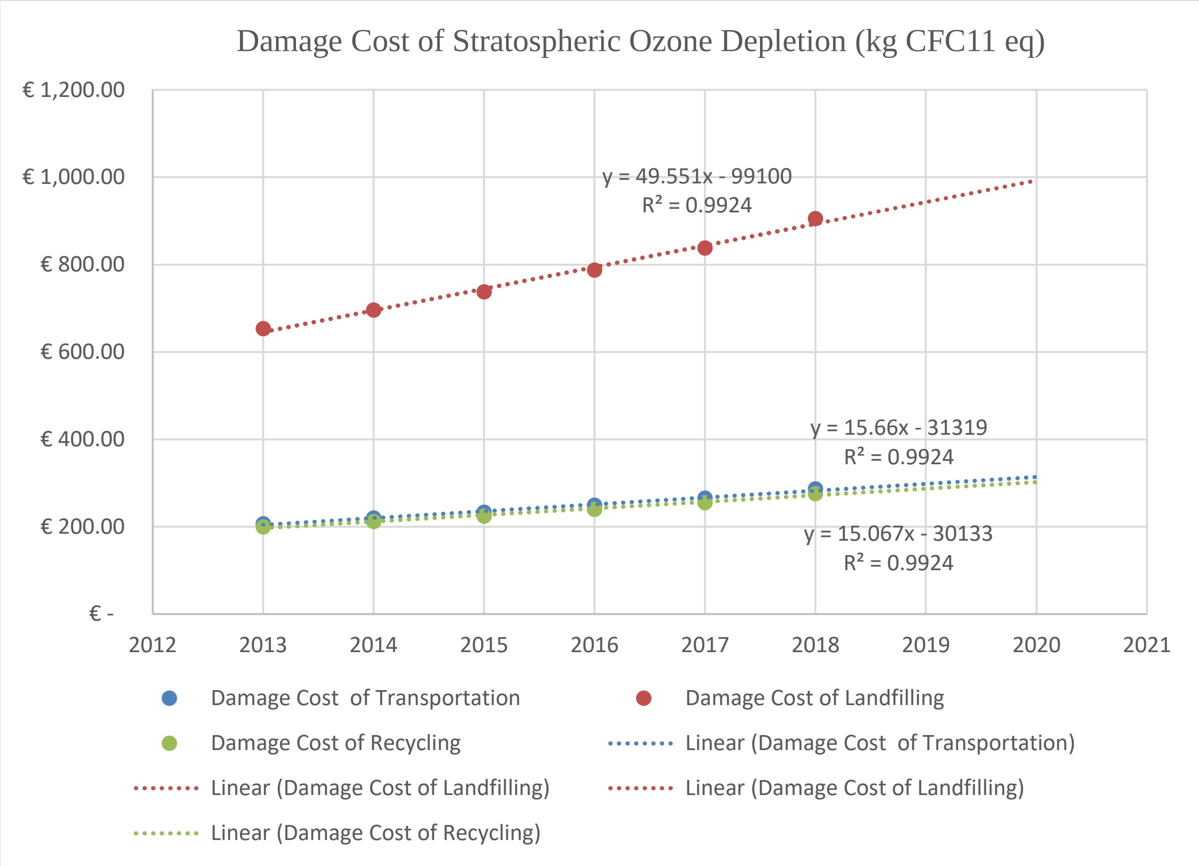


Figure 6.4: Damage Cost of Stratospheric Ozone Depletion

Figure 6.4 shows the damage cost results for the three methods from 2013 to 2018, in which $R^2 = 0.992$. Moreover, the prediction of damage cost for the three methods appears to increase in the future, and mostly in the landfilling method. The damage cost results of Stratospheric Ozone Depletion illustrate that there is a slight increase every year with an average of €20 to €80, and mostly in the landfilling method.

6.4.5. Human Carcinogenic Toxicity (Cancer)

Not all substances with a carcinogenic substance ED50 are important for carcinogenics to humans. The International Agency for Research on Cancer (IARC) is part of the World Health Organisation (WHO), which assessed the carcinogenic risk of 844 substances (mixtures) to humans by assigning a carcinogenicity class to every substance (IARC, 2004). In addition, classes indicate the strength of evidence for carcinogenicity, which is expressed from studies in humans and experiments on animals and other significant data. Furthermore, the definition of the two scenarios used from the data, egalitarian and hierarchic scenarios, includes all 844 substances (IARC-category 1, 2A, 2B, 3 or no classification), while the individualistic scenario includes only the substances with robust evidence of carcinogenicity (IARC-category 1, 2A and 2B).

LCA results for the midpoint in Figure 6.1 present that the landfilling method has a 31.5% impact on Human Carcinogenic Toxicity compared to transportation and recycling method, which were 66 % and 2.47% respectively. From a unit perspective, waste concrete in landfill as final disposal was 21,587.083 (kg 1,4-DCB) compared to transportation and recycling methods, which were 45149 and 1692.438 respectively. Furthermore, the impact of Human Carcinogenic Toxicity (Cancer) is growing in several categories of cancer, which result in damage to human health.

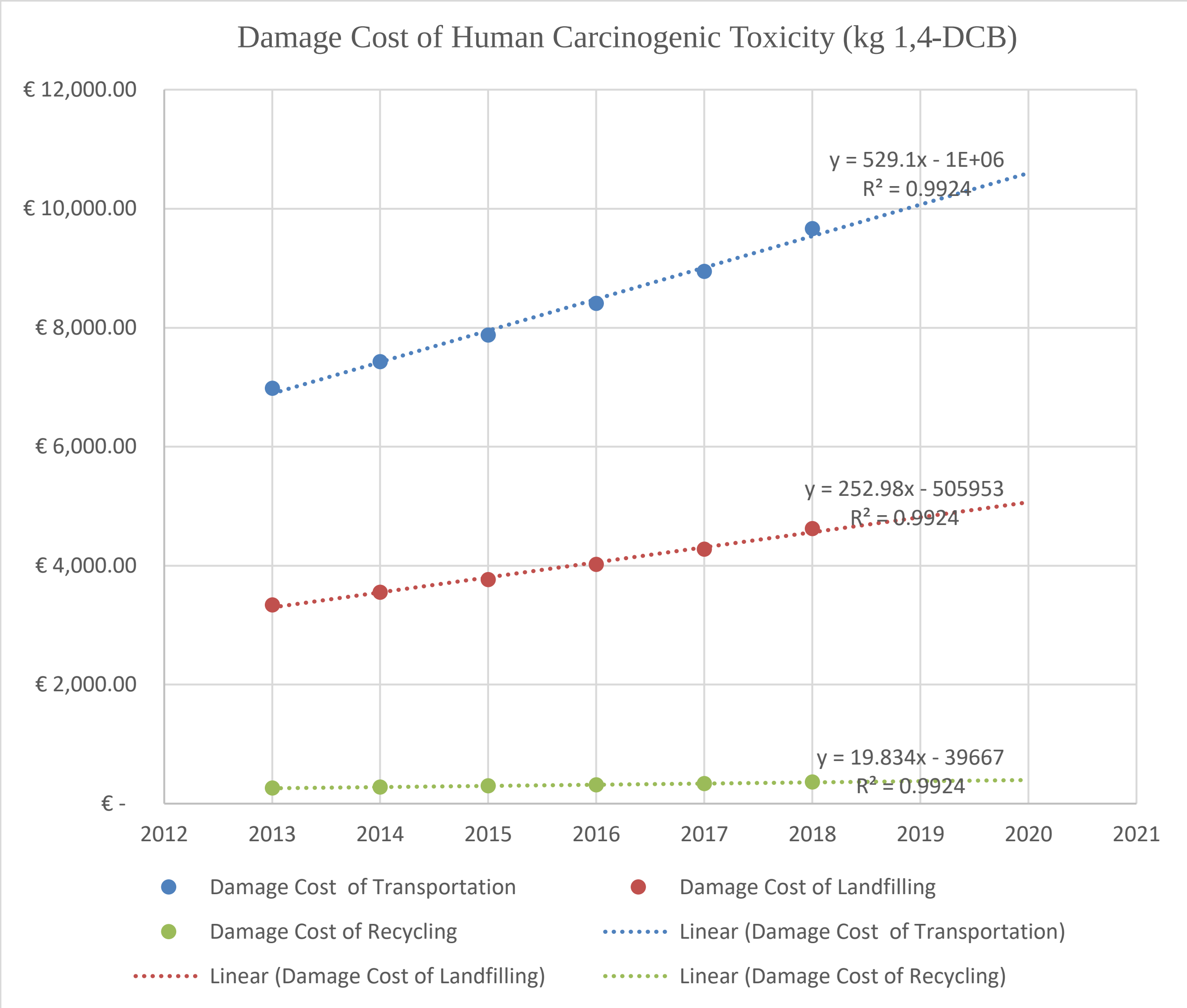


Figure 6.5: Damage Cost of Human Carcinogenic Toxicity

Figure 6.5 demonstrates the damage cost of Human Carcinogenic Toxicity for the three methods from 2013 to 2018, in which $R^2 = 0.992$. Likewise, the damage cost forecast for Human Carcinogenic Toxicity of the three methods seems to show an increase in the future. The results of the damage cost for Human Carcinogenic Toxicity show more cost impact in the transportation process, and little cost impact in both landfilling and recycling methods.

6.4.6. Human Non-Carcinogenic Toxicity (Non-Cancer)

The results of LCA in Figure 6.1 show that the landfilling method has a 28.6% impact on Human Non-Carcinogenic Toxicity, while the transportation process is 68.7% and recycling 2.76%. In terms of the unit, the landfilling method has an impact at 746,082.49 (kg 1,4-DCB), transportation at 1,792,538.1 and recycling method 72,150.367. Although, Human Non-Carcinogenic Toxicity (Non-Cancer) lead to damage to human health as the endpoint area of protection.

The damage price of Human Non-Carcinogenic Toxicity (Non-Cancer) was not calculated to find the damage cost, because it is not available in the Handbook Environmental Prices 2017 or other resources.

6.4.7. Global Warming

The damage modelling of the impact category of climate change is segmented into several steps, such as emission of greenhouse gases in kg results in an increased atmospheric concentration of greenhouse gases (ppb), which also leads to an increase in radiative forcing capacity (w/m2), and results in an increase in the global mean temperature (°C). The influence of increasing the temperature ultimately leads to damage to human health and ecosystems (RIVM Report, 2016-0104).

The LCA outcome in Figure 6.1 shows the results of waste concrete in landfill has an impact of 54.4% for Global Warming compared to transportation and recycling method, which are 28.6% and 17% respectively. From a unit viewpoint, waste concrete in landfill contributes 7,228,4957 (kg CO2 eq) compared to transportation and recycling method, which are 3,793,4796 and 2,2640,946 individually. Moreover, Global Warming increases malnutrition and damage to freshwater and damage to terrestrial species, which lead to damage to human health and ecosystems.

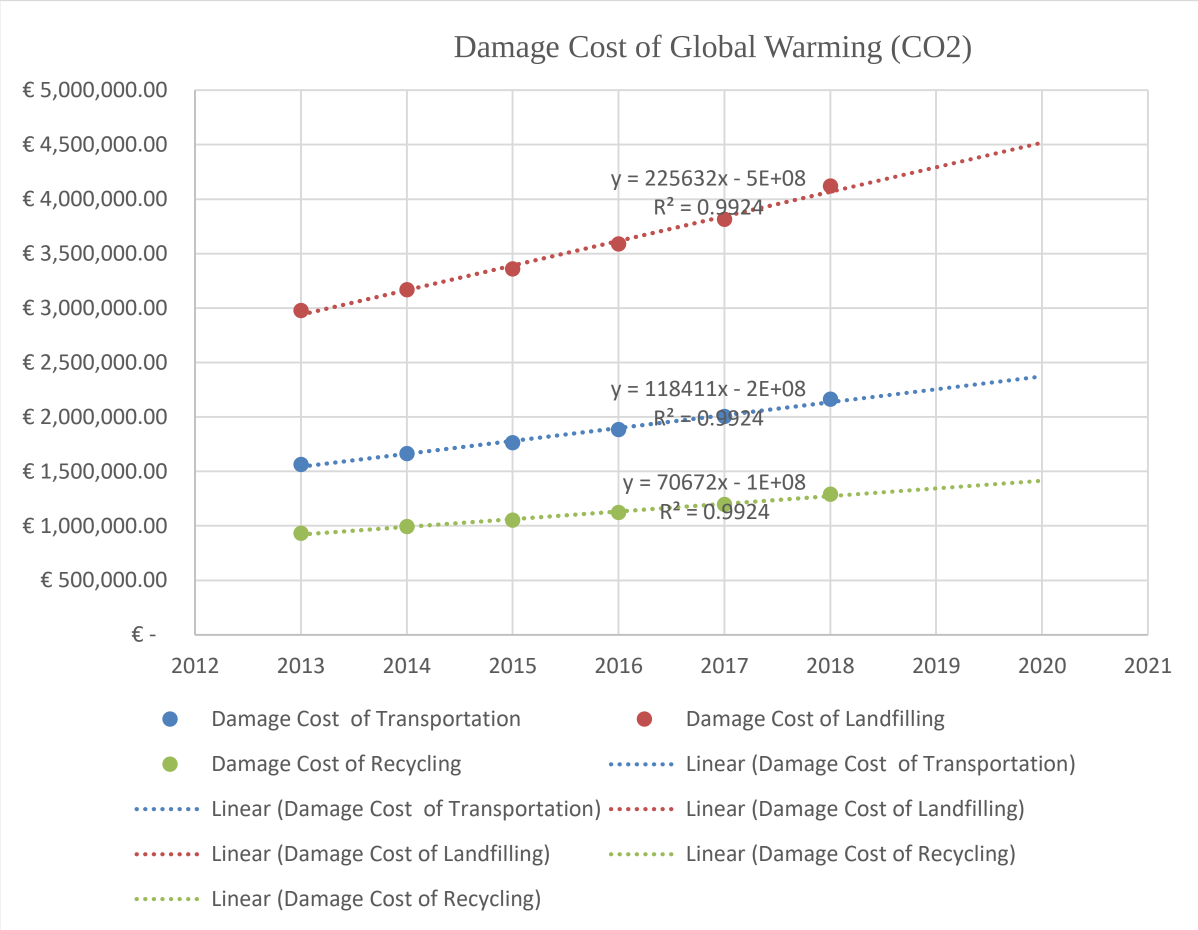


Figure 6.6: Damage Cost of Global Warming

Figure 6.6 determines the damage cost of Global Warming from 2013 to 2018, in which $R^2 = 0.9924$. In addition, the projection for the damage cost of Global Warming for the three methods is increasing yearly. Furthermore, the damage cost of Global Warming shows a huge impact cost in all methods and especially in putting concrete waste into landfill, which cost €4,120,242.50 in 2018.

6.4.8. Water Use/Consumption

With regard to humans, reduction in the availability of freshwater leads to challenges between different water uses. Moreover, a small percentage of irrigation leads to decrease the production and increase in malnutrition between local populations. Although, the relation of people's vulnerability to malnutrition is increasing compared to lower human development indexes (HDI) with a range of industrial countries (HDI>0.88) that have enough evidence in order to buy food, which avoids malnutrition and results in no damage arising to human health. The modelling of terrestrial ecosystems' impacts are through the potential decline in vegetation and plant variety. Moreover, a decrease in blue water, for example, water in lakes, rivers, aquifers and precipitation, leads to a reduction in the available green water in soil moisture and causes a reduction in plant species. Furthermore, a fraction of freshwater fish evaporate based on estimation of water consumption related to species discharge as associated with river entrances (RIVM Report, 2016-0104).

Based on Figure 6.1, which presents the LCA results, it shows that the landfilling method was 87.5% for Water Use/Consumption, while the transportation process was 8.76% and the recycling method 3.75%. In the unit of a cubic metre, landfilling consumed 1,012,710.4 (m3), transportation 101,376.27 and recycling 43,459.147. Water Use has an effect on malnutrition and damage to freshwater terrestrial species, which lead to damage to human health and ecosystems as the endpoint area of protection.

The damage price of Water Use/Consumption is not listed in the Handbook Environmental Prices 2017 or other resources, so damage cost is not considered.

6.4.9. Freshwater Ecotoxicity

The potential impact on the marine environment is strongly related to account of additional inputs for essential metals to oceans, which lead to toxic effects. The egalitarian and hierarchic scenarios provided the calculation for the sea and oceanic compartments of marine ecotoxicological impacts, while the individualistic scenario includes the calculation of the sea compartment for essential metals. Essential metals are cobalt, copper, manganese, molybdenum and zinc. Moreover, a decrease in blue water, for example, water in lakes, rivers, aquifers and precipitation, leads to a reduction in the available green water in soil moisture and causes a reduction in plant species. Furthermore, a fraction of freshwater fish evaporate based on estimation of water consumption related to species discharge as associated with river entrances (RIVM Report, 2016-0104).

The LCA in Figure 6.1 illustrates that the landfilling method has a 51.1% influence on Freshwater Ecotoxicity, transportation is 42.1% and recycling is 6.81%. Waste concrete in landfill results in 630,299.48 (kg 1,4-DCB) compared to transportation and recycling method, which are 519406.790 and 83955.099 respectively. In addition, the damage pathway of Freshwater Eco-toxicity is linked to freshwater species, which results in damage to ecosystems.

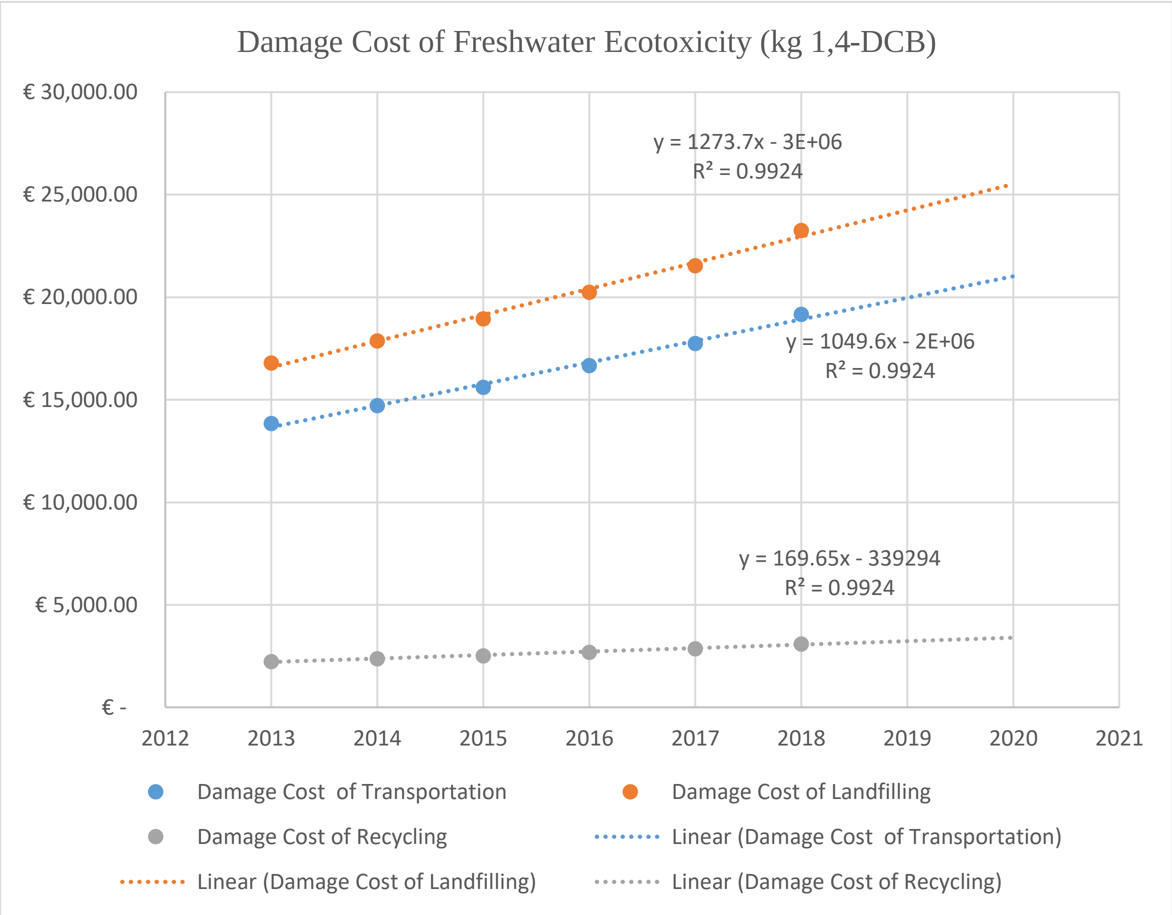


Figure 6.7: Damage Cost of Freshwater Ecotoxicity

Figure 6.7 defines the damage cost of Freshwater Ecotoxicity for the three methods, in which R^2 = 0.992. Moreover, the damage cost forecast for Freshwater Ecotoxicity is accumulative yearly. The damage cost for Freshwater Ecotoxicity is high for the landfilling method and lower for the recycling method.

6.4.10. Freshwater Eutrophication

The occurrence of freshwater eutrophication is due to the discharge of nutrients into the soil or the bodies of freshwater and subsequent rise in nutrient levels, such as phosphorus and nitrogen. Furthermore, ecological influence is correlated with freshwater eutrophication and they are various. Moreover, it is by following structure environmental impacts offset by increasing nutrient emissions into freshwater, which results in increasing nutrient uptake by autotrophic organisms such as cyanobacteria and algae, and heterotrophic species such as fish and invertebrates. In addition, those effects lead to relative loss of species. Moreover, emission impacts on freshwater are based on transfer of phosphorus from the soil to freshwater bodies, which residence time in freshwater systems and on Potentially Disappeared Fraction (PDF) by increase in phosphorus concentrations in freshwater (RIVM Report, 2016-0104).

LCA results in Figure 6.1 show that the landfilling method has a 63.1% impact on Freshwater Eutrophication, transportation is 27.5% and recycling is 9.39%. In addition, from a unit perspective, waste concrete in landfill was 6365.8561 (kg P eq), which is more than transportation and recycling method, which were 2775.797 and 947.888 respectively. Moreover, Freshwater Eutrophication impacts on freshwater species, which leads to damage to ecosystems.

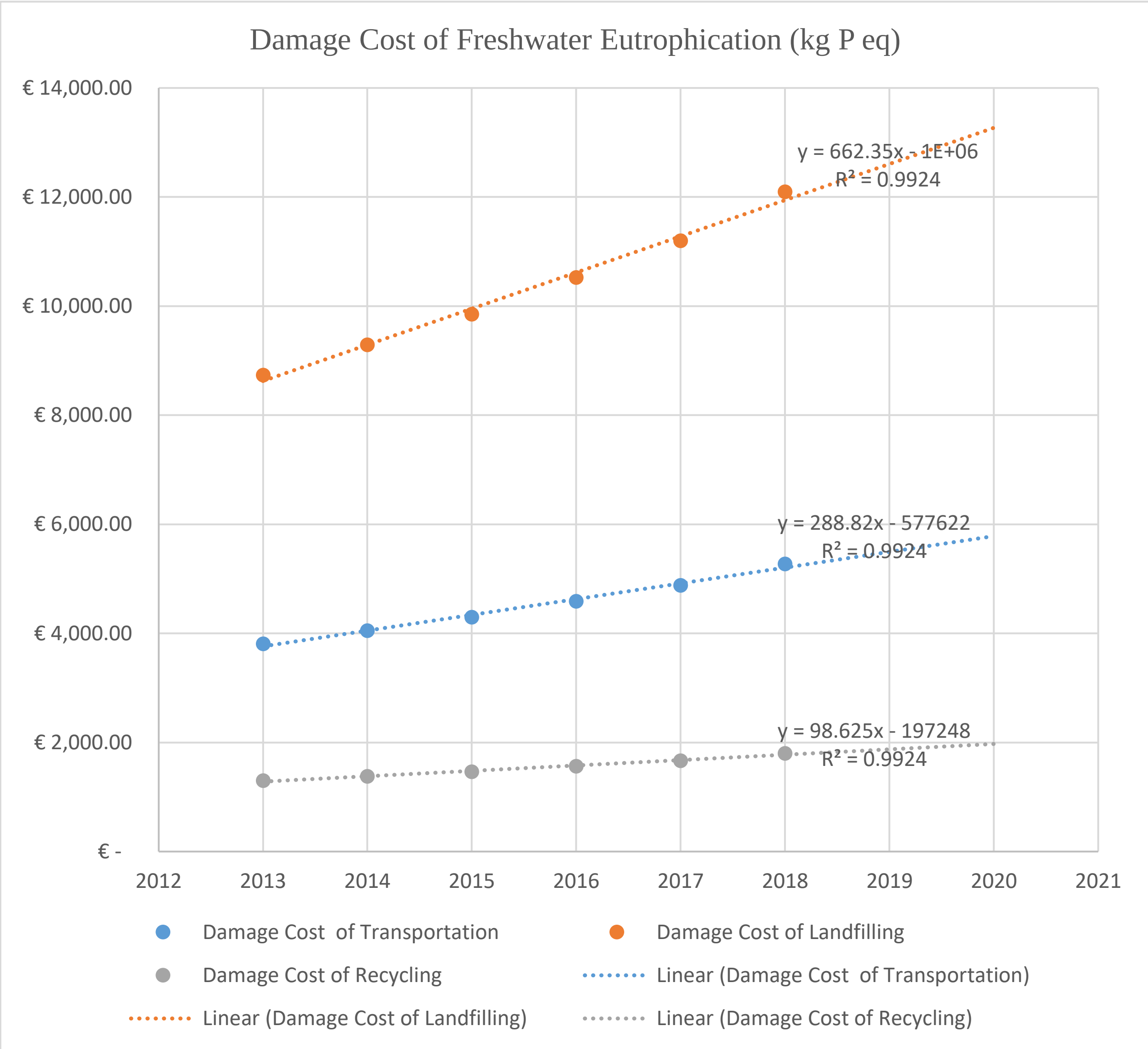


Figure 6.8: Damage Cost of Freshwater Eutrophication

Figure 6.8 elaborates the damage cost of Freshwater Eutrophication from 2013 to 2018, in which $R^2 = 0.992$. Moreover, the projection for Freshwater Eutrophication is increasing yearly and highly in the landfilling method. The damage cost for Freshwater Eutrophication is found to be higher in the landfilling method, at €12,095.13 in 2018, compared to other methods.

6.4.11. Tropical Ozone Formation (Ecosystems)

Ozone Depleting Substances (ODSs) lead to an increase in UVB-radiation, which results in damage to human health. Although, when ozone increases it leads to a decrease in atmospheric ozone concentration, which results in a larger radiation portion of UVB hitting the Earth, more radiation has a negative impact on human health, which leads to an increase in the occurrence of skin cancer and cataracts (RIVM Report, 2016-0104).

Figure 6.1 shows the LCA results of waste concrete in landfill as final disposal has a 54.2% impact on Tropical Ozone Formation (Ecosystems) compared to transportation and recycling method, which are 22.3% and 23.6% respectively. From a unit perspective, waste concrete in landfill as final disposal was 612,126.24 (kg NOx eq) compared to transportation and recycling method, which were 251,954.24 and 266,298.17 respectively. Furthermore, the damaging impact of Tropical Ozone Formation (Ecosystems) is linked to terrestrial species and also leads to damage to ecosystems.

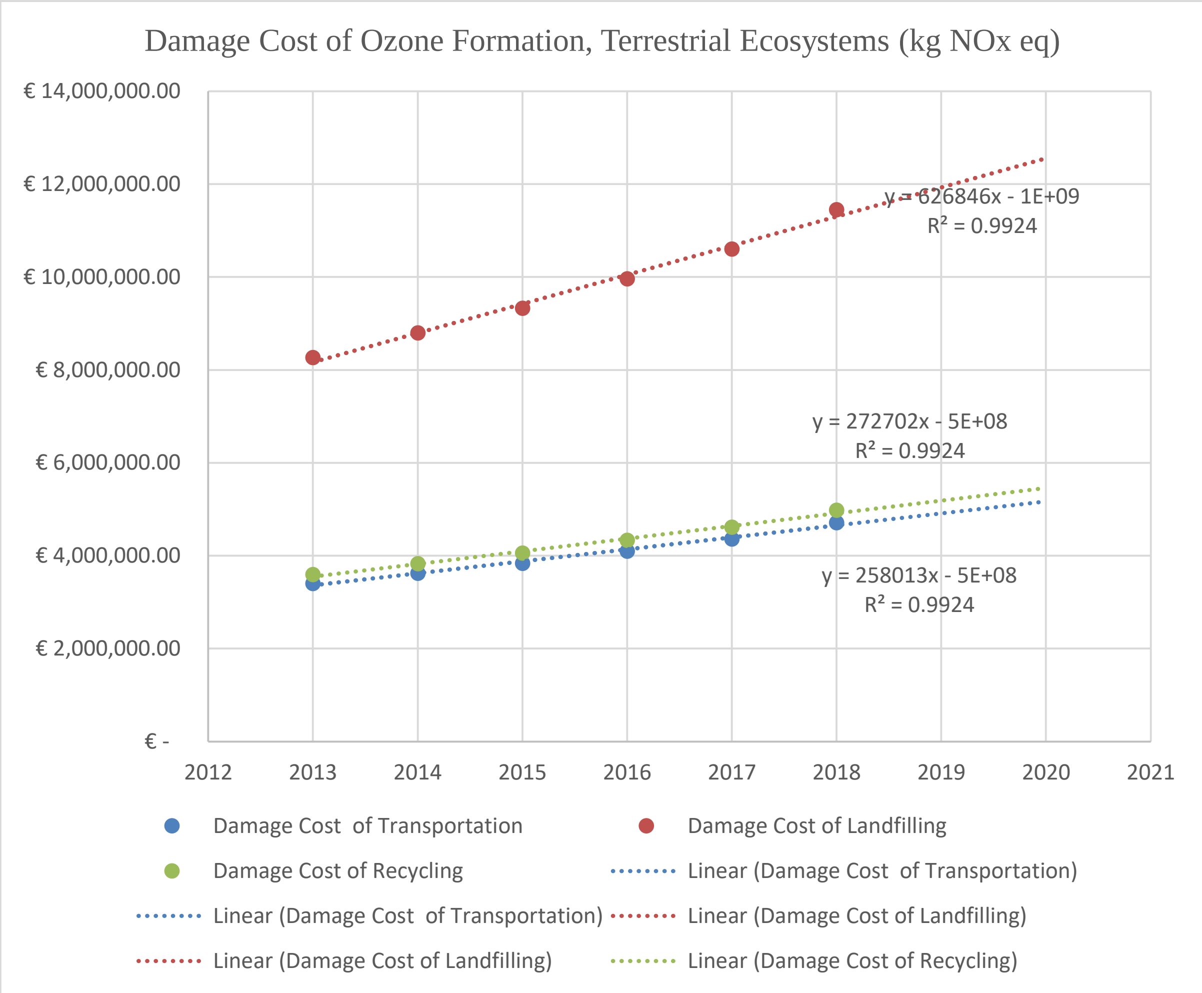


Figure 6.9: Damage Cost of Ozone Formation, Terrestrial Ecosystems

Figure 6.9 illustrates the damage cost of Ozone formation (Terrestrial Ecosystems) for the three methods from 2013 to 2018, in which $R^2 = 0.992$. Moreover, the forecast for Ozone formation (Terrestrial Ecosystems) is growing yearly and more so in the landfilling method. Moreover, the damage cost in the landfilling method was €11,446,760.69 compared to the transportation process and recycling method, which were €4,711,544.29 and €4,979,775.78 respectively.

6.4.12. Terrestrial Ecotoxicity

LCA results in Figure 6.1 show that waste concrete in landfill as final disposal has a 37.5% impact on Terrestrial Ecotoxicity compared to transportation and recycling method, which are 60% and 2.53% respectively. On the other hand, waste concrete in landfill as a final disposal has an impact at 129037190 (kg 1,4-DCB) compared to transportation and recycling method, which are 20,620,2510 and 8,705,868.7 respectively. In addition, the damage pathway of Terrestrial Ecotoxicity is related to damage caused to terrestrial species, which leads to damage to ecosystems.

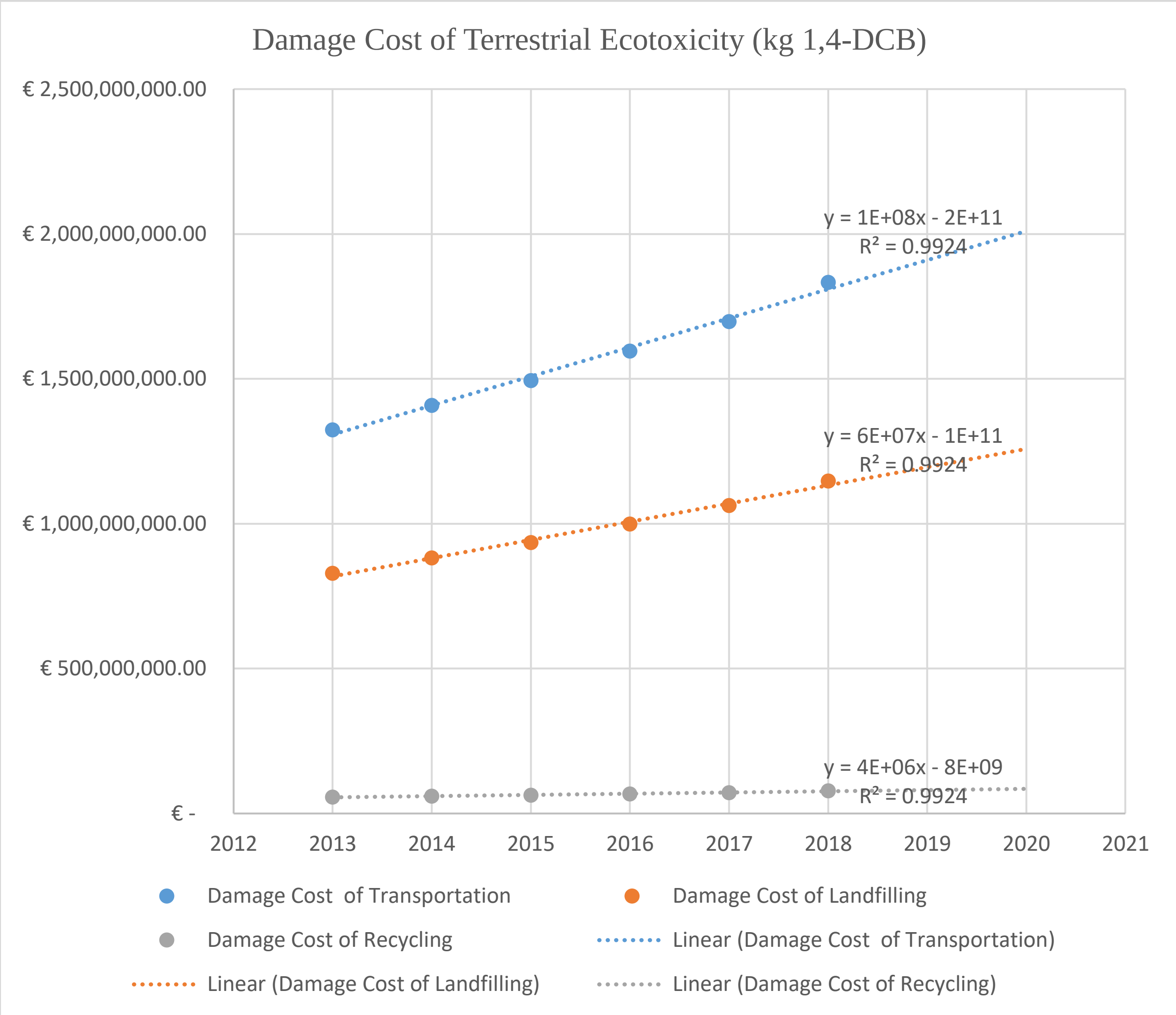


Figure 6.10: Damage Cost of Terrestrial Ecotoxicity

Figure 6.10 clarifies the Terrestrial Ecotoxicity damage cost from 2013 to 2018 for the three methods, and $R^2 = 0.992$. The estimated future damage cost for Terrestrial Ecotoxicity is rising yearly, especially in the transportation process. The influence of the landfilling method has reached €1,833,140,313.90, which shows significant damage compared to landfilling and recycling methods, which are €1,147,140,619.10 in transportation and €77,395,172.74 in recycling.

6.4.13. Terrestrial Acidification

There are some substances associated with atmospheric deposition of inorganic compounds that can result in a change in soil acidity, for example, sulphates, nitrates and phosphates. In addition, the definitions of the optimum levels of acidity for most plant species are provided. Moreover, the optimum level of acidity is harmful for certain types of species and denoted as acidification. Goedkoop et al. (1999) and Hayashi et al. (2004) state that variations in levels of acidity can result in changes in species occurrence. The main acidifying emissions are NO_x, NH_3 or SO_2 (Van Zelm et al. 2015). There is a calculation of characterisation factors for acidification related to vascular plant species in biomes worldwide. In addition, as described by Roy et al. (2012a,b) and Van Zelm et al. (2007b) Fate factors accounting for the environmental persistence of an acidifying substance can be calculated with an atmospheric deposition model correlated with a geochemical soil acidification model. Moreover, effect factors are calculated for the damage to an ecosystem that occurs because of acidification and expressed with dose-response curves of the potential occurrence of plant species as it is resulting from logistic regression functions (Azevedo et al. 2013c).

The results of LCA in Figure 6.1 illustrate the influence of waste concrete in landfill as final disposal, which have a 57% impact on Terrestrial Acidification compared to transportation at 22.8% and recycling at 20.1%. Furthermore, the landfilling method was 348,418.92 (kg SO2 eq) compared to transportation (139,557.24) and recycling (122,878.69). Moreover, Terrestrial Acidification is related to damage to terrestrial species and damage to ecosystems.

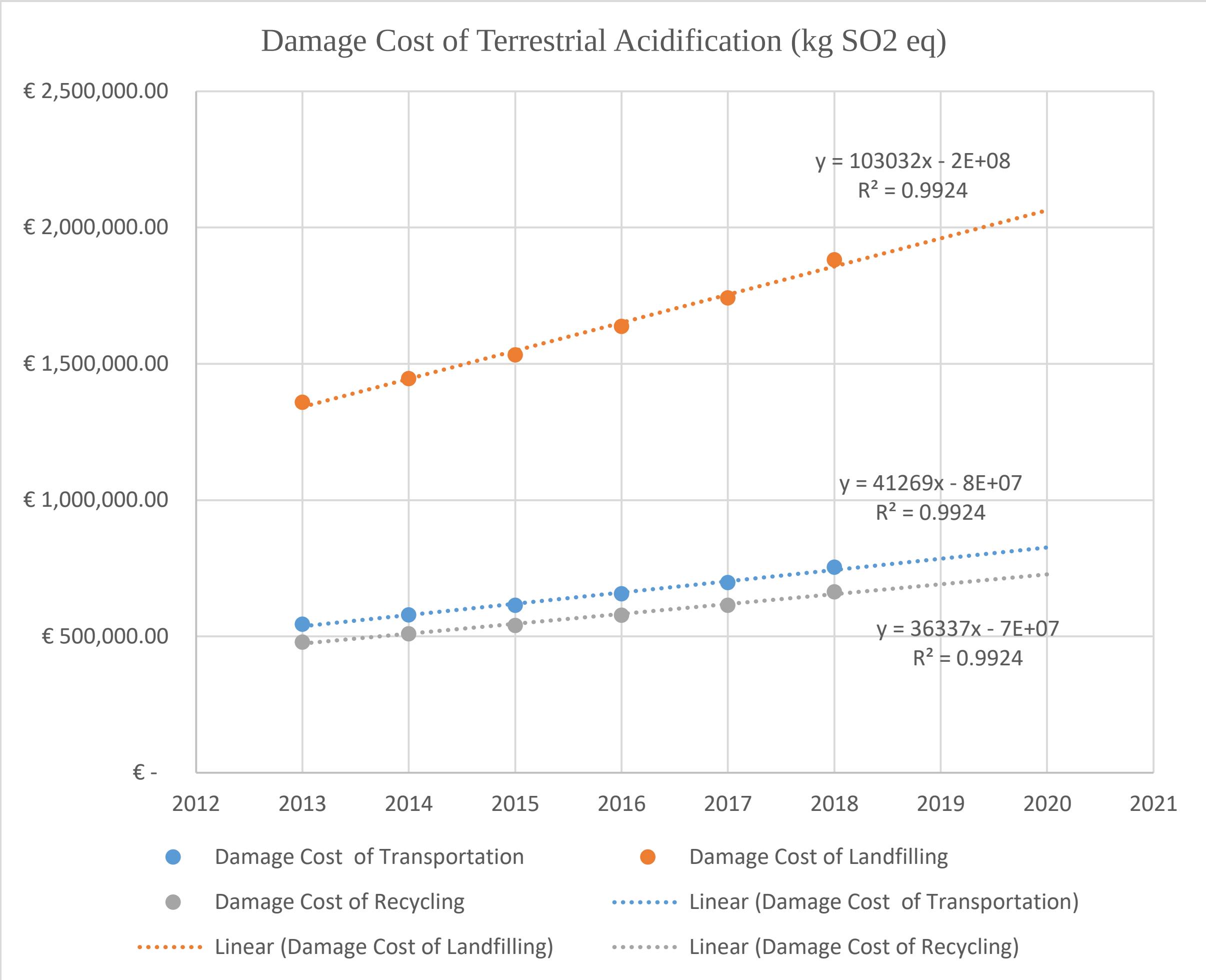


Figure 6.11: Damage Cost of Terrestrial Acidification

Figure 6.11 shows the damage cost of Terrestrial Acidification for the three methods over several years, in which $R^2 = 0.992$. Moreover, the projected damage cost for Terrestrial Acidification is increasing yearly, particularly with regard to the landfilling method, where the cost is €1,881,462.17 compared to transportation at €753,609.10 and recycling at €663,544.93.

6.4.14. Land Use/Transformation

The impact pathway of land use contains the direct, local impact of land use on terrestrial species through change of land cover and actual use of new land. Moreover, variation of land impacts consequently on original habitat and original species composition. In addition, land use signifies agricultural and urban activities, which additionally exclude land as suitable habitat for many species. Moreover, the cause-and-effect of land use leads to correlated species loss in terrestrial ecosystems. (RIVM Report, 2016-0104).

LCA midpoint results in Figure 6.1 show that the landfilling method had a 79.3% impact on Land Use compared to transportation and recycling method, which were 20% and 0.693% respectively. From a unit perspective, waste concrete in landfill as final disposal was 6,073,915 (m2a crop eq), transportation 1,534,466.5 and recycling 53,116.578. Furthermore, the damage pathway of Land Use/Transformation increases the loss of terrestrial species and leads to damage to ecosystems as the endpoint area of protection.

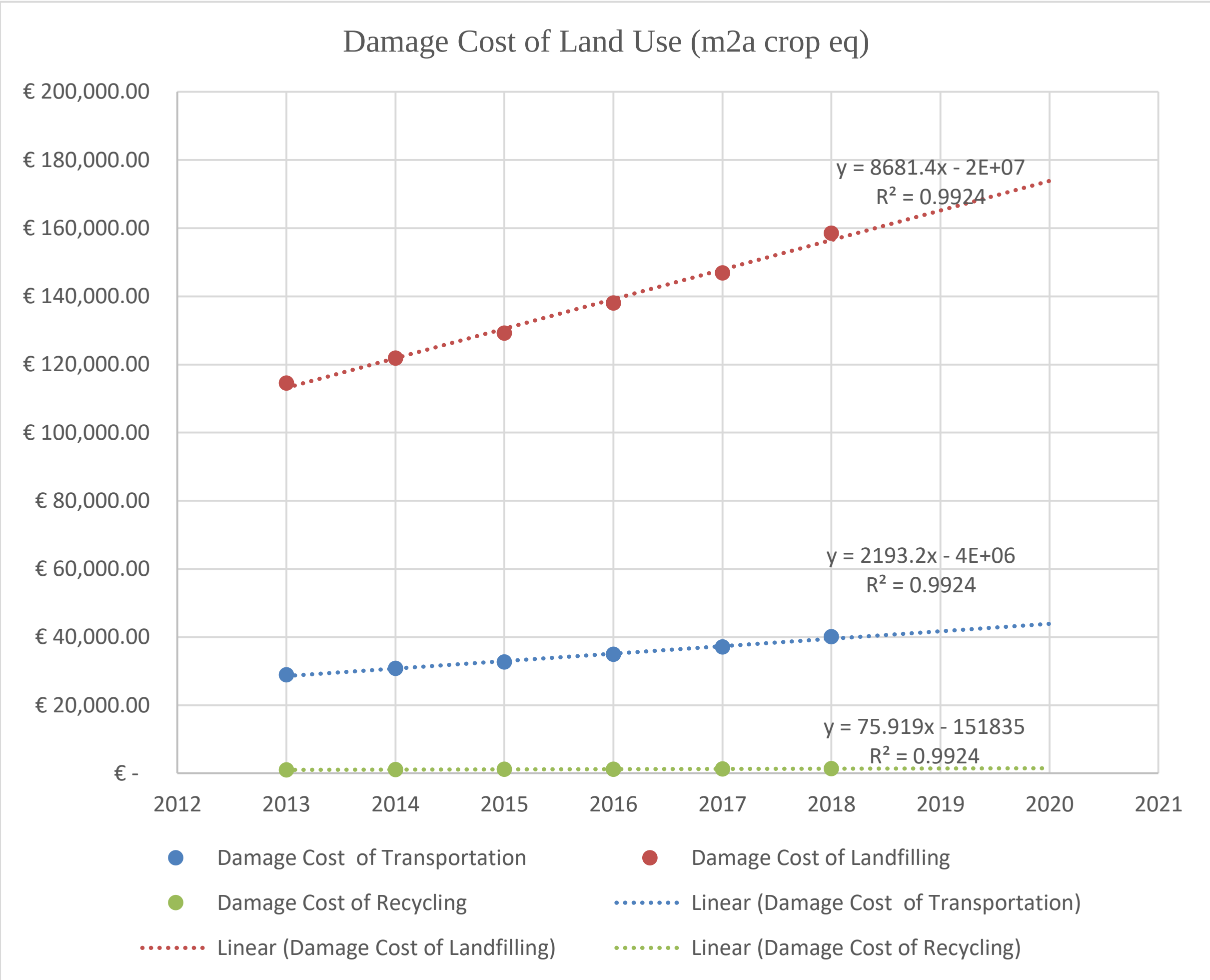


Figure 6.12: Damage Cost of Land Use

Figure 6.12 demonstrates the damage cost of Land Use in different years, in which $R^2 = 0.992$. The forecast damage cost for Land Use accumulates yearly, mainly in the landfilling method compared to recycling and transportation, at €158,529.18.

6.4.15. Marine Ecotoxicity

The potential impact on the marine environment is strongly related to account of additional inputs for essential metals to oceans, which lead to toxic effects. The egalitarian and hierarchic scenarios provided the calculation of the sea and oceanic compartments of marine ecotoxicological impacts, while the individualistic scenario includes the calculation of the sea compartment for essential metals. Essential metals are cobalt, copper, manganese, molybdenum and zinc (RIVM Report, 2016-0104).

LCA results in Figure 6.1 elaborate that waste concrete in landfill has a 49.4% impact on Marine Ecotoxicity compared to the transportation process and recycling method, which are 44.1 % and 6.5% respectively. Moreover, waste concrete in landfill reached 243,805.57 (kg 1,4-DCB) compared to the transportation and recycling method, which are 217533.09 and 32096.588 respectively. Furthermore, the damage pathway of Marine Eco-toxicity is damage to marine species, which leads to damage to ecosystems as the endpoint area of protection.

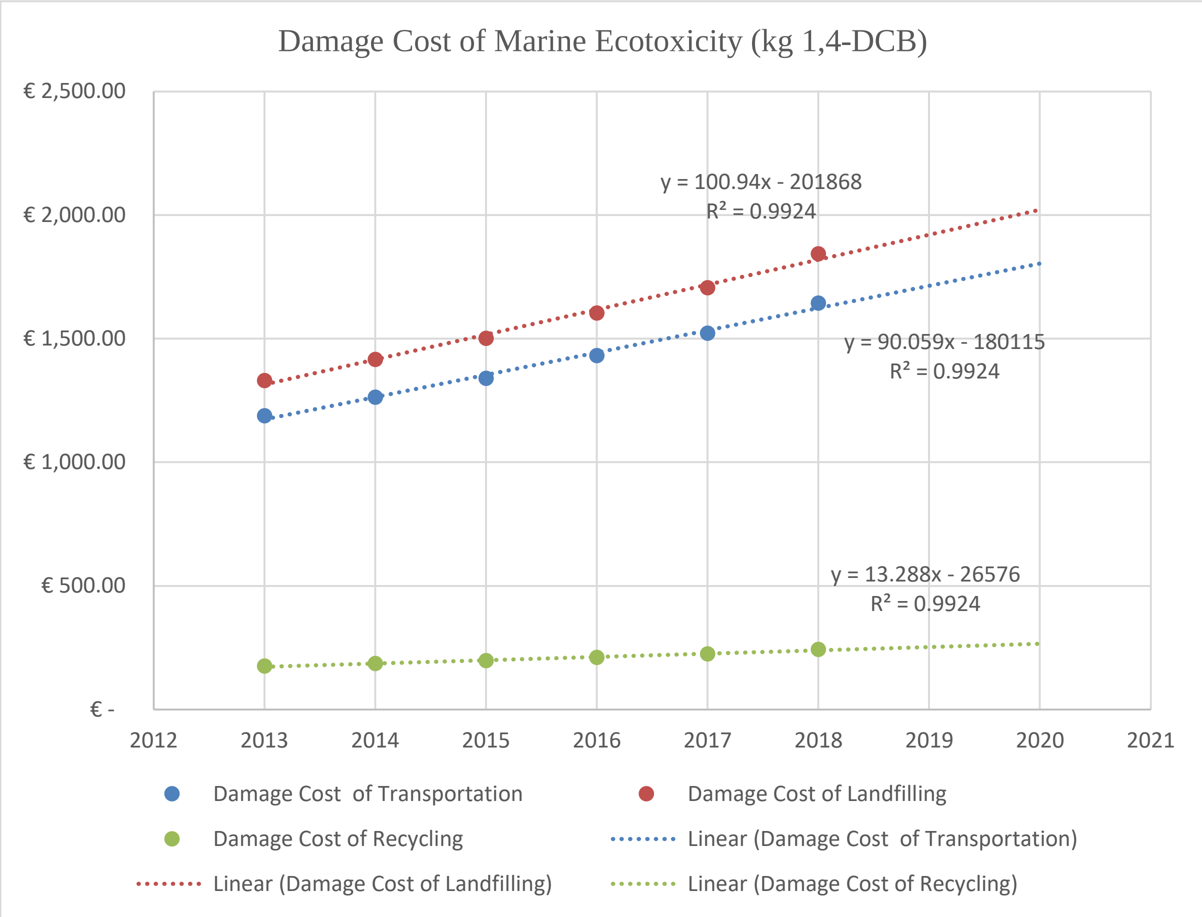


Figure 6.13: Damage Cost of Marine Ecotoxicity

Figure 6.13 determines Marine Ecotoxicity damage cost from 2013 to 2018, in which R^2 = 0.992. Moreover, the predicted damage cost for Marine Ecotoxicity has a slight accumulation yearly in all methods, but is higher for landfilling, which is forecast to increase slightly at an average of €200.

6.4.16. Marine Eutrophication

Marine eutrophication happens because of the runoff and leaching of plant nutrients from the soil to discharge into riverine or marine systems, leading to a subsequent rise in nutrient levels, for example, phosphorus and nitrogen (N). Furthermore, N is assumed to be a limiting nutrient in marine waters (Cosme et al. 2015). On the other hand, ecological influence is correlated with marine eutrophication, because nutrient enrichment points to a variety of ecosystem impacts, one of which is the existence of benthic oxygen depletion. In addition, it leads to the start of hypoxic waters if in excess to anoxia, so-called 'dead zones', which is one of the most severe and widespread root causes of disturbance to marine ecosystems. Moreover, impacts on marine water are related to certain circumstances, such as transfer of dissolved inorganic nitrogen (DIN) from soil and freshwater bodies or straight into marine water, which residence time in marine systems on dissolved oxygen (DO) depletion and on potentially disappeared fraction (PDF) (RIVM Report, 2016-0104).

Figure 6.1 shows that the landfilling method was 62.6% for Marine Eutrophication compared to transportation and recycling method, which were 28.2% and 9.21% respectively. From the unit perspective, the landfilling method was 498.070 (kg N eq), transportation 224.635 and recycling 73.324. Marine Eutrophication impacts on marine species and causes damage to ecosystems.

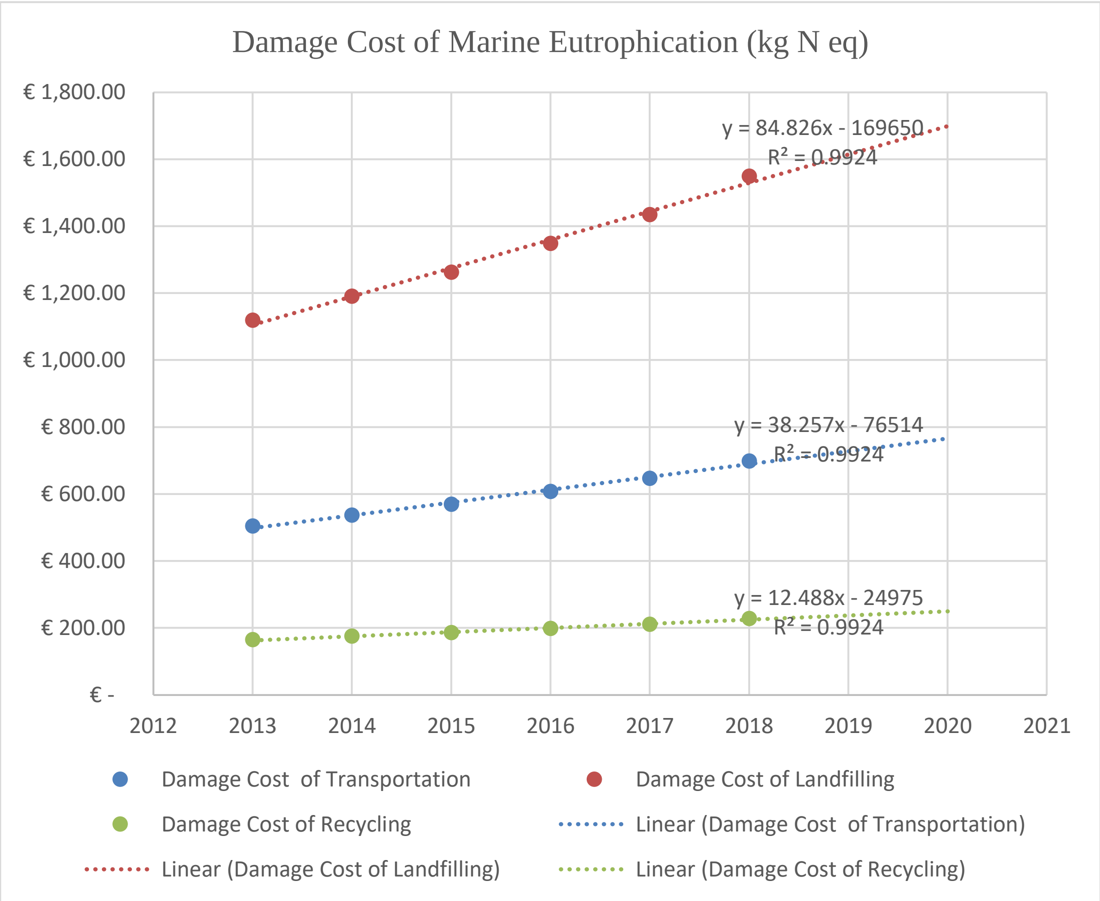


Figure 6.14: Damage Cost of Marine Eutrophication

Figure 6.17 shows the damage cost of Marine Eutrophication over several years, in which $R^2 = 0.992$. Moreover, the forecast damage cost for Marine Eutrophication has a slight accumulation yearly in all methods and significantly in the landfilling method, which has more of a direct impact than transportation and recycling methods.

6.4.17. Mineral Resources Scarcity

The damage modelling for the impact category of mineral resource scarcity is subdivided into some steps. In addition, primary extraction of a Mineral Resource (ME) leads to a decrease in Ore Grade (OG), which has an influence on the concentration related to resource in global ores, since it increases ore produced per kilogram of Mineral Resource Extracted (OP). Furthermore, joined predicted future extraction of mineral resource leads to average Surplus Ore Potential (SOP), which represents the midpoint indicator for the impact category. Moreover, an increase in Surplus Ore Potential (SOP) leads to surplus cost potential, while the two indicators follow the standard of mining sites with more grades or less costs for SOP and SCP respectively (RIVM Report, 2016-0104).

Figure 6.1 illustrates that waste concrete in landfill has a 56.3% impact on Mineral Resources Scarcity compared to transportation at 29.6% and recycling 14.1%. Furthermore, waste concrete in landfill was 88,503.364 (kg Cu eq) compared to the transportation and recycling method, which were 46,520.161 and 22,079.584 individually. Mineral Resources Scarcity influences extraction costs, which leads to damage to resource availability.

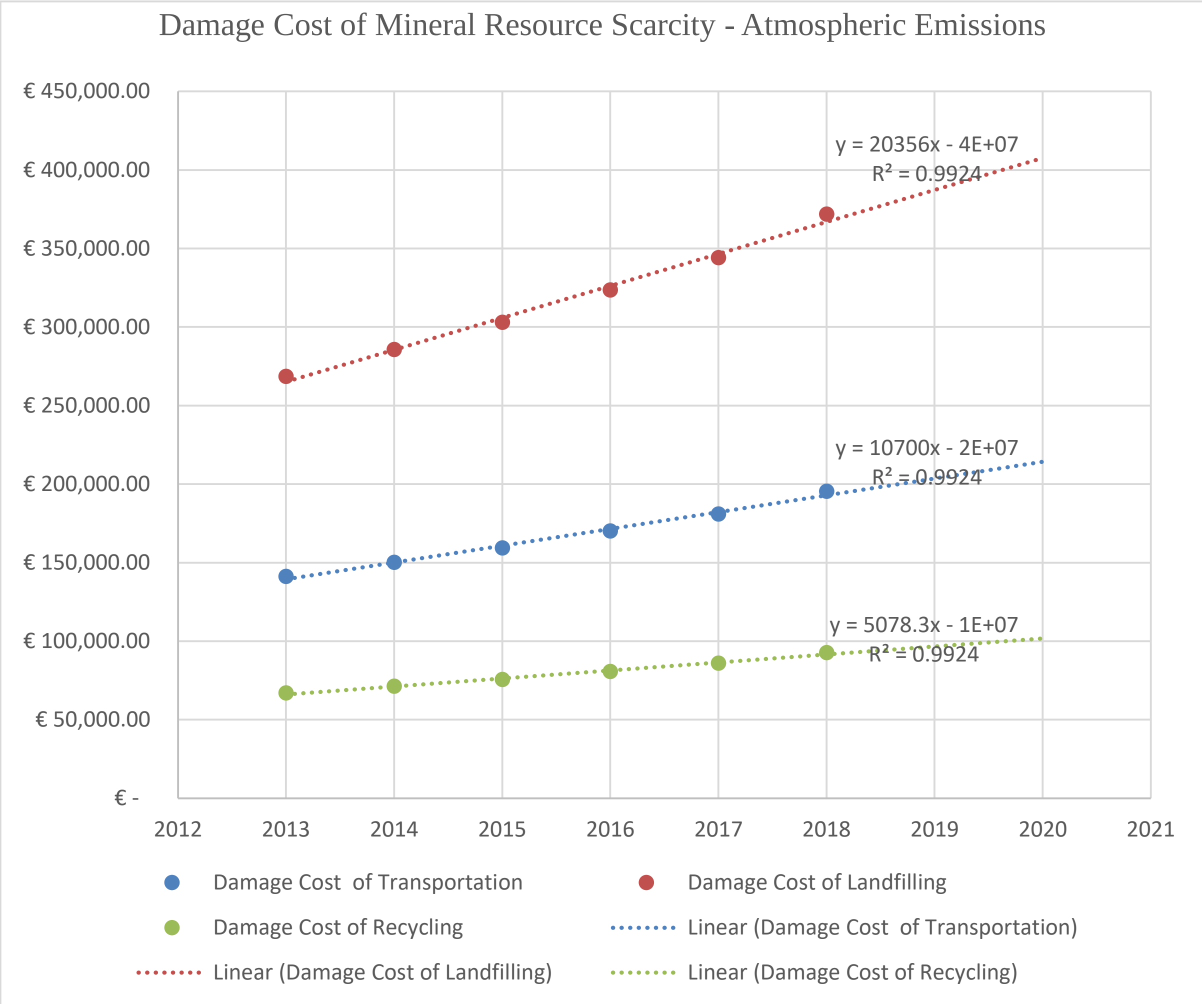


Figure 6.15: Damage Cost for Mineral Resource Scarcity

Figure 6.15 demonstrates the damage cost for Mineral Resource Scarcity (Atmospheric Emissions) over several years, in which $R^2 = 0.992$. In addition, the estimated future damage cost for Mineral Resource Scarcity increases annually in all methods, but landfilling has more of an influence than the other methods.

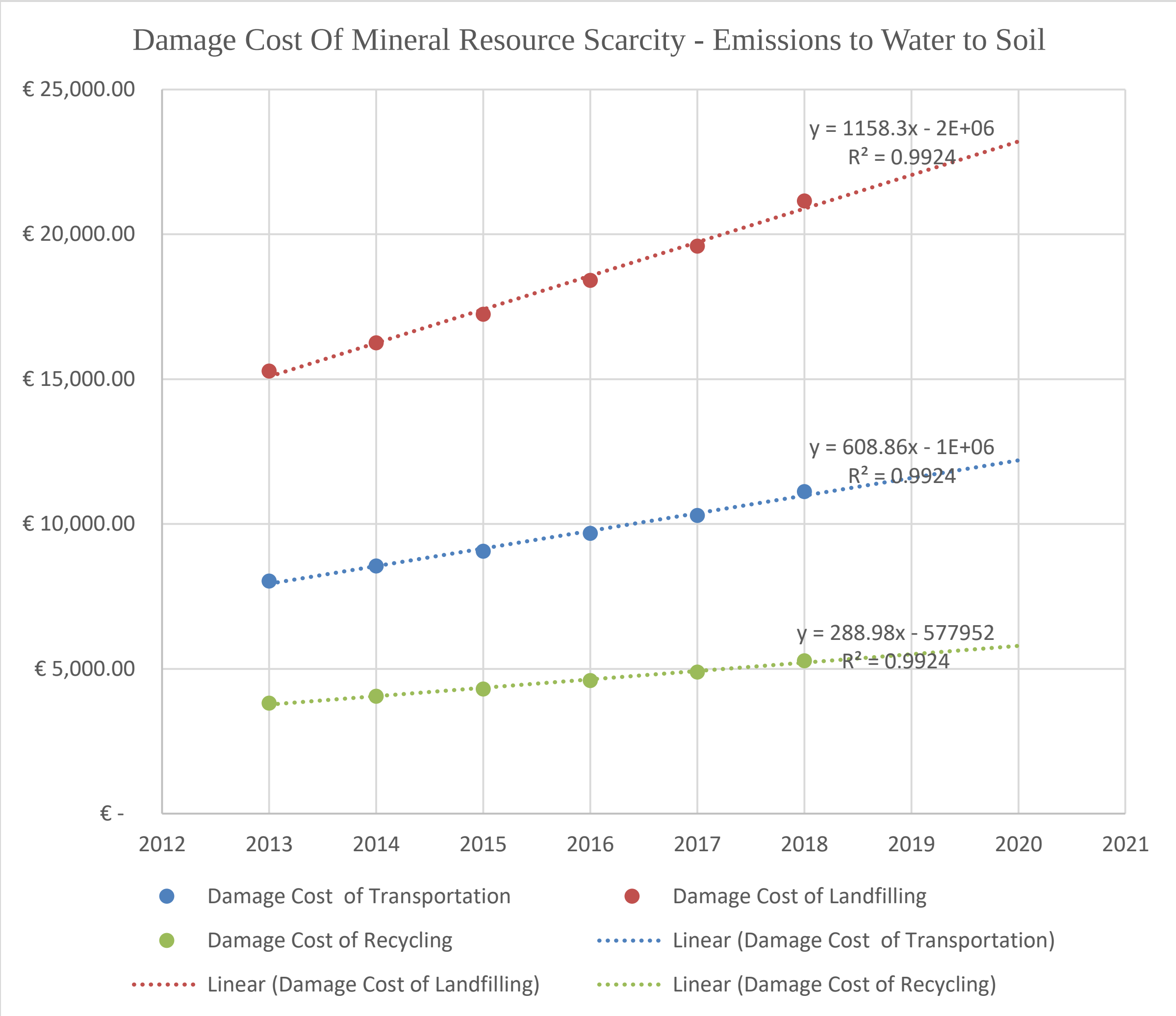


Figure 6.16: Mineral Resource Scarcity - Emissions to Water to Soil

Figure 6.16 shows the damage cost of Mineral Resource Scarcity (Atmospheric Emissions) from 2013 to 2018, in which $R^2 = 0.992$. In addition, the projection of damage cost for Mineral Resource Scarcity (Emissions to Water to Soil) increases gradually in all methods, although there is a huge damage cost from landfilling compared to recycling and transportation methods.

6.4.18. Fossil Resources Scarcity

The impact category of fossil resource scarcity has damage modelling which is subdivided into a number of steps. Moreover, it is predicted in endpoint modelling that fossil fuels with the low costs are extracted firstly. In addition, an increase in fossil fuel extraction has an influence on increase in costs due to change in production technique or sourcing from a higher location. For instance, when there is a depletion of all conventional oil, other techniques, for example, enhanced oil recovery, are used or oil is produced in different geographical locations with higher costs, for example, Arctic regions (Ponsioen et al. 2014). On the other hand, combining expected future extraction of fossil resource results in surplus cost potential (SCP), which represents the endpoint indicator for the impact category, although estimation of damage to natural resource scarcity is included. Moreover, fossil fuel potential with higher heating value is used as the midpoint indicator (RIVM Report, 2016-0104).

Figure 6.1 elaborates that waste concrete in landfill as final disposal has aa 61.5% impact on Fossil Resources Scarcity compared to transportation and recycling method, which are 24.4% and 14.1% respectively. From a unit perspective, waste concrete in landfill as final disposal has an impact at 31,723,569 (kg oil eq) compared to transportation and recycling method, which are 12,570,694 and 7,266,799.6 respectively. In addition, Fossil Resources Scarcity increases extraction costs and leads to damage to resource availability.

The damage price for Fossil Resources Scarcity is not available in the Handbook Environmental Prices 2017 or other resources, so it was calculated only as an endpoint assessment based on ReCipe2016.

6.5. Descriptive Statistics for the Damage Cost of Concrete Waste

Table 6.3 shows the descriptive analysis of damage cost for the year 2018, which includes 15 impacts and three methods of concrete waste management. Descriptive analyses for the years 2013 to 2017 are available in Appendix 6.

Table 6.3: Descriptive Analysis of 2018 Damage Cost

Descriptive Analysis of 2018 Damage Cost			
Name	**Damage Cost of Transportation**	**Damage Cost of Landfilling**	**Damage Cost of Recycling**
Mean	123138485.6	79215221.29	6561921.394
Standard Error	122143710.1	76289648.54	5107303.007
Median	40049.576	158529.181	5277.021
Mode	#N/A	#N/A	#N/A
Standard Deviation	473060555.1	295468538.3	19780499.49
Sample Variance	2.23786E+17	8.73017E+16	3.91268E+14
Kurtosis	15.0	14.991	14.314
Skewness	3.873	3.871	3.754
Range	1833140028	1147139714	77394944.71
Minimum	285.962	904.842	228.037
Maximum	1833140314	1147140619	77395172.74
Sum	1847077284	1188228319	98428820.92
Count	15	15	15

The best-known measure for the fundamental tendency of scores distribution is the mean. On the other hand, factors by median define the midpoint of distribution or standard deviation, which is known as the measure of the variability of a distribution. Additional characterisations of the statistics are skewness and kurtosis, which define the lack of steadiness in which the data is high or normal.

Table 6.3 shows the Descriptive Statistics for 2018 damage cost. The Mean of damage cost for transportation is equal to 123,138,485.6 compared to landfilling and recycling at 79,215,221.29 and 6,561,921.394 respectively. The result of Standard Deviation for transportation is 473,060,555.1 compared to landfilling and recycling, which are 295,468,538.3 and 19,780,499.49 respectively. In addition, the Kurtosis and Skewness results for transportation are higher than landfilling and recycling: 15.0 for transportation, 14.991 for landfilling and 14.314 for recycling. The same goes for Skewness, which is 3.873 for transportation process, 3.871 for landfilling method and 3.754 for recycling method. Overall, transportation had high results in all aspects compared to the methods of landfilling and recycling.

6.6. Endpoint LCIA by ReCipe2016

Characterisation factors of the endpoint level relate to three areas, which are human health, quality of ecosystem and lack of resources. In addition, the endpoint method provides superior information on ecological flows and more uncertain than the midpoint method (Hauschild & Huijbregts, 2015).

Figure 6.17 illustrates the endpoint impact results in percentage for the three methods, which are landfilling, recycling and transportation. Moreover, Figure 6.2 shows the results of ReCipe2016 endpoint LCIA that consists of 22 endpoint impacts, which are Global Warming (Human Health), Global Warming (Terrestrial Ecosystems), Global Warming (Freshwater Ecosystems), Stratospheric Ozone Depletion, Ionising Radiation Ozone Formation (Human Health), Fine Particulate Matter Formation Ozone Formation (Terrestrial Ecosystems), Terrestrial Acidification, Freshwater Eutrophication, Marine Eutrophication, Terrestrial Ecotoxicity, Freshwater Ecotoxicity, Marine Ecotoxicity, Human Carcinogenic Toxicity, Human Non-Carcinogenic Toxicity, Land Use, Mineral Resource Scarcity, Fossil Resource Scarcity, Water Consumption (Human Health), Water Consumption (Terrestrial Ecosystem) and Water Consumption (Aquatic Ecosystems). In addition, endpoint assessment concludes the LCA by three endpoint damages as required by ReCipe2016 LCIA, which are Human Health, Ecosystems and Resources.

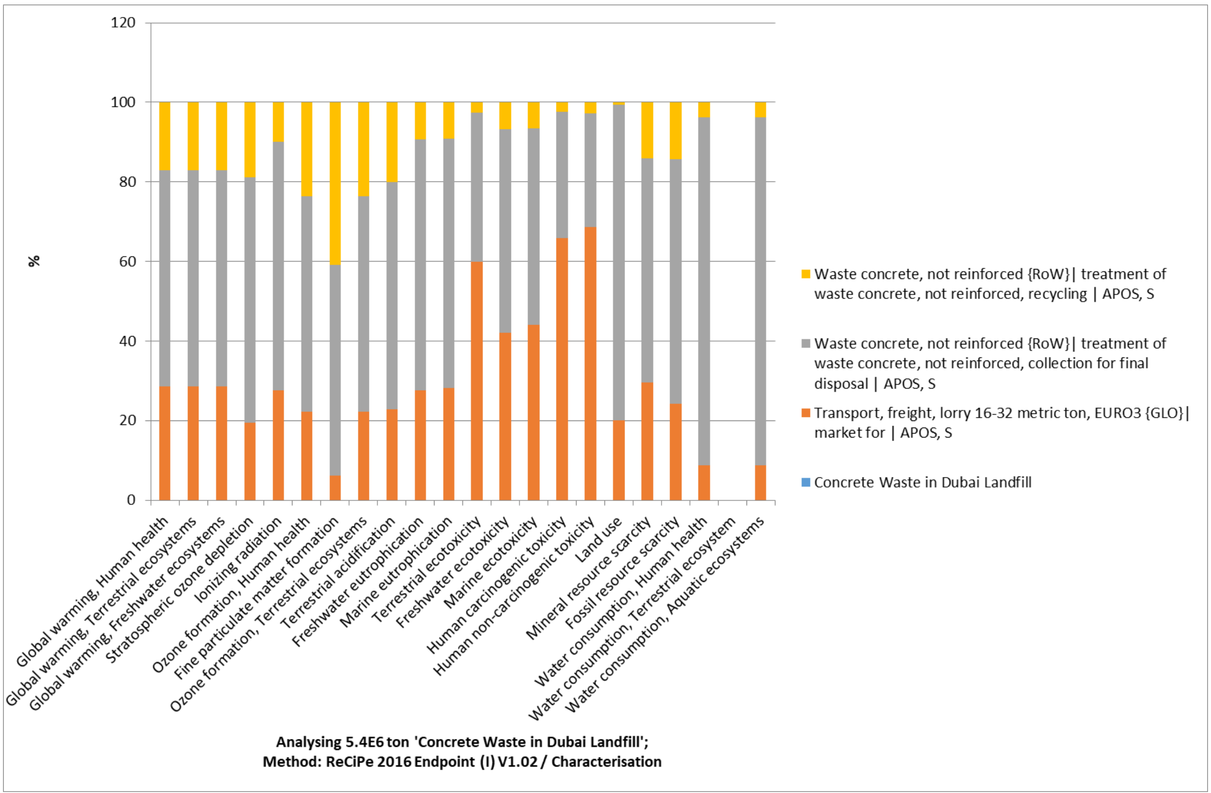


Figure 6.17: Results of each endpoint impact in percentage (SimaPro software, ReCiPe2016 Endpoint Method)

The results in Figure 6.17 show that putting concrete waste into landfill has a greater ecological impact compared to recycling and transportation. From an overall standpoint, landfilling method and transportation process have more ecological impacts compared to the recycling method in all endpoint impacts, but transportation process had high results in both Human Carcinogenic Toxicity and Human Non-Carcinogenic Toxicity. Appendix 2 includes the endpoint results in units.

6.6.1. Fine Particulate Matter Formation

Figure 6.17 shows the endpoint assessment results of waste concrete in landfill as final disposal, which has a 53% impact on Fine Particulate Matter Formation compared to transportation and recycling method, which are 6.22% and 40.8% respectively. From a unit perspective, waste concrete in landfill as final disposal was 93.4 (DALY) compared to transportation and recycling method, which were 11.0 and 71.9 respectively. Although, this impact increases in respiratory disease and leads to damage to human health.

6.6.2. Tropical Ozone Formation (Human Health)

Figure 6.17 shows that landfilling was 54.1% for Ozone Formation (Human Health), transportation was 22.3% and recycling was 23.6 %. In quantity of (DALY) unit, landfill impacts were 0.547 (DALY), transportation 0.226 and recycling 0.234. Moreover, the damage pathway of Tropical Ozone Formation (Human Health) raises the influence in respiratory disease and damage to human health as the endpoint area of protection.

6.6.3. Ionising Radiation

Figure 6.17 shows that waste concrete in landfill as final disposal has a 62.5% impact on Ionising Radiation compared to transportation and recycling method, which are 27.5% and 9.97% respectively. The landfill method was 0.0093 (DALY) compared to transportation and recycling, which were 0.004 and 0.002 individually. Furthermore, Ionising Radiation leads to various types of cancer and an increase in other diseases or causes, which impacts on the damage to human health.

6.6.4. Stratospheric Ozone Depletion

Figure 6.17 shows that waste concrete in landfill as final disposal, has a 61.7% impact on Stratospheric Ozone Depletion compared to transportation and recycling method, which were 19.5% and 18.8% respectively. In addition, the landfill method has an impact of 0.007 (DALY), while transportation is 0.002 and recycling is 0.002. In addition, Stratospheric Ozone Depletion increases the risk of several kinds of cancer, which cause diseases and lead to damage to human health as the endpoint area of protection.

6.6.5. Human Carcinogenic Toxicity (Cancer)

Figure 6.17 elaborates that waste concrete in landfill has a 31.6% impact on Human Carcinogenic Toxicity (Cancer), while transportation is 66.0% and recycling is 2.48%. The same goes for waste concrete in landfill, which was 0.072 (DALY) compared to high ecological impacts from the transportation process at 0.149 and recycling at 0.006. Furthermore, Human Carcinogenic Toxicity (Cancer) increases the numerous types of cancer, which leads to damage to human health as the endpoint area of protection.

6.6.6. Human Non-Carcinogenic Toxicity (Non-Cancer)

It is clarified in Figure 6.17 that the landfilling method was 28.6% for Human Non-Carcinogenic Toxicity (Non-Cancer) compared to transportation and recycling method, which were 68.6% and 2.77% respectively. From a unit perspective, waste concrete in landfill as final disposal was 0.170 (DALY) compared to transportation and recycling method, which were 0.408 and 0.016 respectively. Moreover, the damage pathway of Human Non-Carcinogenic Toxicity (Non-Cancer) causes damage to human health.

6.6.7. Global Warming (Human Health)

Table 6.19 shows that waste concrete in landfill as final disposal has a 54.4% impact on Global Warming (Human Health), while transportation is 28.6% and recycling is 17%. From a unit perspective, waste concrete in landfill as final disposal was 5.87 (DALY) compared to transportation and recycling method, which were 3.10 and 1.83 respectively. Moreover, the damage pathway of Global Warming (Human Health) increases malnutrition and damage to freshwater and damage to terrestrial species, which lead to damage to human health and ecosystems.

6.6.8. Global Warming (Terrestrial Ecosystems)

Table 6.19 shows the endpoint results of the three methods; for example, landfilling was 54.4% for Global Warming (Terrestrial Ecosystems) compared to transportation and recycling method, which were 28.6% and 17% respectively. From a unit perspective, waste concrete in landfill as final disposal was 0.0385 (species.yr) compared to transportation and recycling methods, which were 0.0202 and 0.0120 respectively. In addition, the damage pathway of Global Warming (Terrestrial Ecosystems) causes a rise in malnutrition and damage to freshwater and terrestrial species, which lead to harm to human health and ecosystems.

6.6.9. Global Warming (Freshwater Ecosystems)

Table 6.19 shows that waste concrete in landfill as final disposal has a 54.4% impact on Global Warming (Freshwater Ecosystems), while transportation is 28.6% and recycling is 17%. Moreover, landfilling was 1.048E-06 (species.yr), transportation 5.50E-07 and recycling 3.283E-07. In addition, damage from Global Warming (Freshwater Ecosystems) leads to an increase in malnutrition and damage to freshwater and terrestrial species, which influence human health and ecosystems.

6.6.10. Water Use/Consumption (Human Health)

Table 6.19 shows that the landfill method has a major effect on Water Use/Consumption (Human Health) at 87.5% compared to transportation and recycling method, which are 8.76% and 3.75% respectively. Moreover, waste concrete in landfill was 3.1394023 (DALY) compared to the transportation and recycling method, which were 0.314 and 0.135. Although, the damage of Water Use/Consumption (Human Health) increases malnutrition and impacts on freshwater and terrestrial species, which influence human health and ecosystems as the endpoint area of protection.

6.6.11. Water Use/Consumption (Terrestrial Ecosystems)

Table 6.19 illustrates the endpoint impacts of the three methods; all results were zero, because it is not applicable for concrete waste by landfilling, recycling and transportation.

6.6.12. Water Use/Consumption (Aquatic Ecosystems)

Table 6.19 shows that the landfilling method has an 87.5% impact on Water Use/Consumption (Aquatic Ecosystems), transportation is 8.76% and recycling 3.75%. In addition, the damage from landfilling was 6.117E-07 (species.yr) compared to transportation and recycling method, which were 6.123E-08 and 2.625E-08 respectively. Moreover, the damage pathway of Water Use/Consumption (Aquatic Ecosystems) causes an increase in malnutrition and damage to freshwater and terrestrial species, which lead to damage to human health and ecosystems.

6.6.13. Freshwater Ecotoxicity

Table 6.19 shows that landfilling as final disposal has a 51.1% impact on Freshwater Ecotoxicity compared to transportation and recycling method, which are 42.1% and 6.81% respectively. From a unit perspective, waste concrete in landfill has a 0.0004 (species.yr) impact, transportation is 0.0004 and recycling is 5.818E-05. Moreover, the damage pathway of Freshwater Ecotoxicity affects the freshwater species and leads to damage to ecosystems.

6.6.14. Freshwater Eutrophication

Table 6.19 elaborates the three methods of concrete waste, in which the landfilling method has a 63.1% impact on Freshwater Eutrophication, transportation is 27.5% and recycling is 9.39%. Waste concrete in landfill as a final disposal was 0.004 (species.yr) compared to transportation and recycling methods, which were 0.002 and 0.0006 respectively.

In addition, Freshwater Eutrophication damages the freshwater species, which leads to damage to ecosystems as the endpoint area of protection.

6.6.15. Tropical Ozone Formation (Ecosystems)

The LCA results in Table 6.19 illustrate the three methods of concrete waste. Landfill was 54.2%, which is equal to 0.079 (species.yr), transportation was 22.3%, which is 0.032501875, and recycling was 23.6%, which denotes 0.344. Furthermore, the damage pathway of Tropical Ozone Formation (Ecosystems) impacts terrestrial species, which results in damage to ecosystems.

6.6.16. Terrestrial Ecotoxicity

Table 6.19 shows that waste concrete in landfill has a 37.5% impact on Terrestrial Ecotoxicity, transportation is 60% and recycling 2.53%. From a unit perspective, waste concrete in landfill as final disposal was 0.001 (species.yr) compared to transportation and recycling methods, which were 0.002 and 9.922E-05 respectively. Although, the damage pathway of Terrestrial Ecotoxicity has an influence on terrestrial species and causes damage to ecosystems.

6.6.17. Terrestrial Acidification

Table 6.19 clarifies the results of the landfilling method for concrete waste, which has a 57.0% impact on Terrestrial Acidification compared to transportation and recycling method, which are 22.8% and 20.1% respectively. Moreover, the landfilling method was 0.074 (species.yr), transportation was 0.030 and recycling was 0.030. Furthermore, Terrestrial Acidification is damaging terrestrial species, which leads to damage to ecosystems as the endpoint area of protection.

6.6.18. Land Use/Transformation

Table 6.19 illustrates the LCA results for endpoint assessment of the three waste concrete methods. Putting concrete waste into landfill has a 79.3% impact on Land Use/Transformation, transportation is 20.0% and recycling is 0.693%. Moreover, waste concrete in landfill as final disposal was 0.054 (species.yr) compared to transportation and recycling methods, which were 0.014 and 0.0004 respectively. Additionally, the damage pathway of Land Use/Transformation increases the damage to terrestrial species, which leads to damage to ecosystems.

6.6.19. Marine Ecotoxicity

Table 6.19 demonstrates that waste concrete in landfill had an impact of 49.4% on Marine Ecotoxicity compared to transportation and recycling methods, which were 44.1% and 6.51% respectively. In a unit perspective, waste concrete in landfill as final disposal was 2.559E-05 (species.yr), transportation was 2.283E-05 and recycling 3.368E-06. Although, the damage pathway of Marine Ecotoxicity impacts on marine species, which results in damage to ecosystems as the endpoint area of protection.

6.6.20. Marine Eutrophication

Table 6.19 shows that waste concrete in landfill as final disposal has a 62.6% impact on Marine Eutrophication, transportation process is 28.2% and recycling method is 9.21%. Moreover, the landfilling method was 8.46E-07 (species.yr) compared to transportation and recycling methods, which were 3.82E-07 and 1.25E-07. Although, the damage pathway of Marine Eutrophication damages marine species, and leads to damage to ecosystems.

6.6.21. Mineral Resources Scarcity

Table 6.19 shows that the landfilling method has an impact cost on Mineral Resources Scarcity of 56.3%, which is equal to $14,104.968 (USD2013), transportation is 29.6%, which costs $7413.812, and recycling is 14.1%, which is $3519.240. On the other hand, the damage pathway of Mineral Resources Scarcity increases the extraction costs, which leads to damage to resource availability.

6.6.22. Fossil Resources Scarcity

Table 6.19 indicates that waste concrete in landfill as final disposal has a cost impact of 61.3% for Fossil Resources Scarcity compared to transportation and recycling method, which are 24.3% and 14.4% respectively. From a unit perspective, waste concrete in terms of landfilling method cost was $13,632,163 (USD2013) compared to transportation and recycling methods, at $5402,381.3 and $3,194,154.8 respectively. Furthermore, the damage pathway of Fossil Resources Scarcity is impacts on oil, gas, coal and energy cost, which lead to damage to resource availability as the endpoint area of protection.

6.7. Final endpoint LCIA by ReCipe2016

Characterisation factors of the endpoint level relate to three areas, which are human health, quality of ecosystem and lack of resources. In addition, the endpoint method provides superior information on ecological flows and more uncertain than the midpoint method (Hauschild & Huijbregts, 2015).

After conducting the midpoint and endpoint assessment, Figure 6.18 illustrates the final endpoint damage results in percentage by referring to ReCipe2016 LCIA which influence Human Health, Ecosystems and Resources. The results indicate that the landfilling method had the highest ecological impact on Human Health, Ecosystems and Resources. Moreover, the landfilling method for concrete waste caused 53.6% damage to Human Health compared to the transportation process of concrete waste and recycling of concrete waste, which are 7.87% and 38.5% respectively. Moreover, the landfilling method has more impact on Ecosystems at 59.1%, whilst transportation process is 23.6% and recycling method is 17.3%. In addition, putting concrete waste into landfill had a 61.3% influence on resources compared to transportation process and recycling method, which are 24.3% and 14.4% individually.

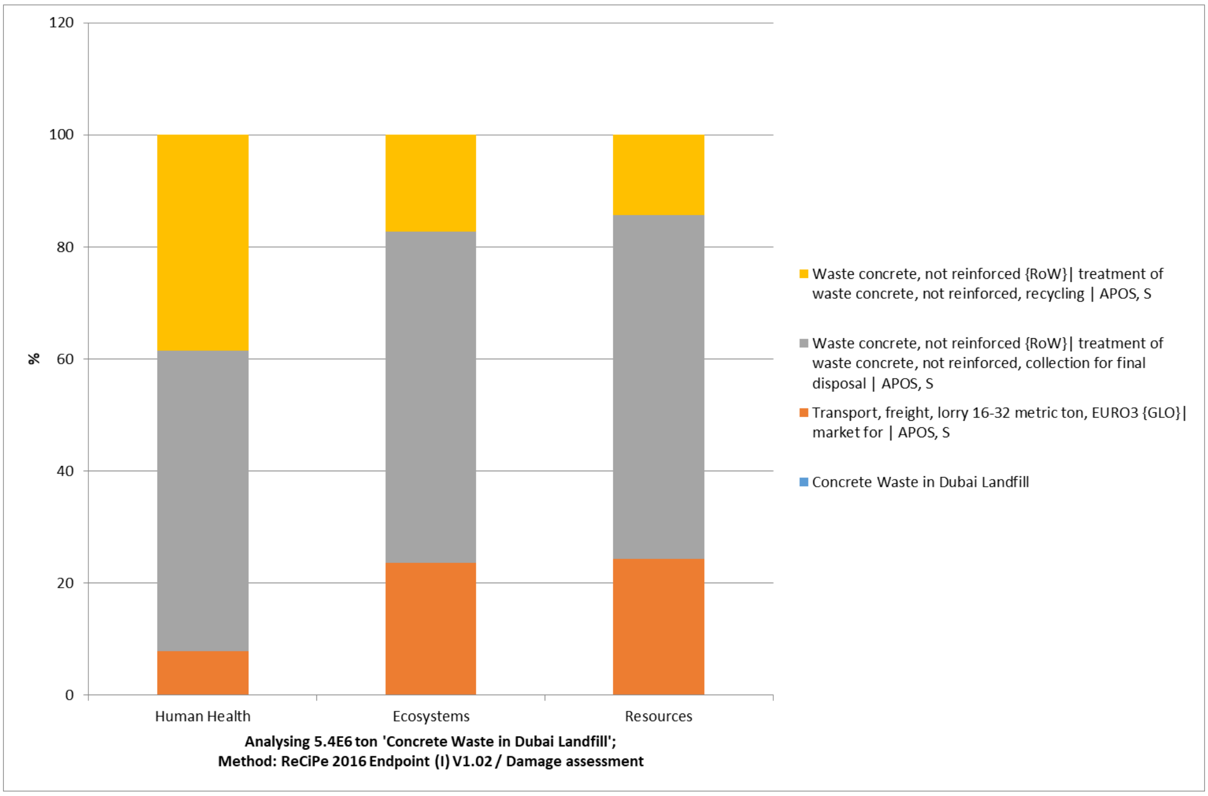


Figure 6.18: Endpoint damages results in percentage (SimaPro software, ReCiPe2016 Endpoint Method)

The final endpoint impact results in units are available in Appendix 3. From the results, it is obvious that concrete waste as a final disposal method has more impact on Human Health at 103.18889 Disability-Adjusted Life Years (DALY) compared to transportation of concrete waste and recycling of concrete waste, which are 15.155 and 74.185 respectively. Moreover, the landfilling method had the highest impact on ecosystems at 0.251 Species-Years (species.yr), whilst transportation process was 0.100 and recycling method was 0.074. Moreover, landfilling cost more in terms of Resources (USD2013), at $13,646,268, compared to transportation of concrete waste and recycling of concrete waste, at $5,409,795.1 and $3,197,674.1 respectively.

6.8. Modelling LCA Impacts and Damage Cost

6.8.1. Mathematical Relationships between LCA and Damage Cost

This section defines the most significant findings of the research, which are linked to the research aim and objectives. Furthermore, based this chapter's results of LCIA and damage cost, mathematical relationships are found to show the influence of the damage cost on the concrete waste for the three methods. The assessment focuses on finding the mathematical relationship between the results of LCA and damage cost for concrete waste based on 15 impacts by using natural logarithm and chooses an exponential curve, which shows the more applicable curve. In addition, natural logarithm was used to find the relationship of 15 impacts and 15 damage costs for the three methods for the year 2018; data for the years 2013 to 2017 is available in Appendix 7.

Furthermore, monetisation is the process signifying the impacts that are linked with social and natural capital in monetary value, which makes them more tangible. In addition, monetising environmental influences is known as natural capital valuation, which is considered a kind of sustainability return on investment analysis and delivers a different financial perception of sustainability. Another method is to find the costs that are correlated with a particular activity, for example, the cost of damage or the cost to replace the service. For instance, the social cost of carbon (SCC) is an estimate of the economic damages associated with an increase of CO2 emissions. Furthermore, monetary methodologies might be variable and include original value decisions. Moreover, one criticism of monetisation is that it could oversimplify complex concerns and it is difficult to quantify or monetise every ecosystem value (Pre-sustainability 2018, p. 1). The association of costs with ecological damages imposed on society (Nguyen et al., 2016). For instance, there are several existing methods to evaluate the monetisation of environmental damages as classification endpoints or safeguard subjects such as ecosystems, humans and resources. In addition, the methods of evaluating monetisation of environmental damages are European models such as EPS 2000 by Steen (1999), ExternE project by Bickel and Friedrich (2005), Ecotax 2002 by Finnveden et al. (2006), and Stepwise2006 by Weidema (2009). Moreover, LIME is a Japanese model by Itsubo et al. (2004), which is a non-European model. Most of the monetary assessment models are not fully completed, since not all environmental impacts are subjected to monetisation (Bos & Vleugel, 2005).

The Handbook Environmental Prices (2017) is used as a method and cost for valuation of ecological impacts by CE Delft (2018). It contains the environmental prices of many impacts. The environmental prices are applicable for social cost or pollution, and are expressed in euros per kilogram pollutant. In addition, environmental prices specify the damage to economic welfare that occurs when one added kilogram of the pollutant makes its way into the atmosphere (Handbook Environmental Prices, 2017). Environmental prices are also determined for immaterial forms of pollution such as noise nuisance and ionising radiation (Handbook Environmental Prices, 2017). Furthermore, they provide average values for the Netherlands, for instance, emissions from a common source of emissions with an average emission site in the year 2015 (Handbook Environmental Prices, 2017). The handbook present the prices at three levels (Handbook Environmental Prices, 2017). Firstly, at the pollutant level, which gives the environmental emissions values of damaging substances (Handbook Environmental Prices, 2017). Secondly, at the midpoint level, which gives values for environmental themes, for example, climate change or acidification (Handbook Environmental

Prices, 2017). Thirdly, at the endpoint level, which gives a value for the environmental impacts of pollution, for instance, damage to human health or ecosystem services (Environmental Prices Handbook, 2017).

Natural logarithm equation was used in Microsoft Excel 2013 for each method, landfilling, recycling and transportation, and for 15 LCA and damage cost impacts, which are Climate change, Ozone layer depletion, Acidification, Freshwater eutrophication, Marine eutrophication, Land use, Terrestrial ecotoxicity, Freshwater ecotoxicity, Marine ecotoxicity, Human toxicity, PM2.5, Nitrogen oxides (Nox) (Human health), Nitrogen oxides (Nox) (Terrestrial ecosystems), Mineral resource scarcity (Atmospheric) and Mineral resource scarcity (Soil). To find the relationships between LCA results and damage costs results, the impact is denoted by the x-axis and the cost is denoted by the y-axis. In addition, the exponential curve was found to be the appropriate curve with R^2 compared to the linear curve. Furthermore, mathematical relationships between LCA and damage cost show that there is always a positive relationship between impact and cost. Moreover, the natural logarithm equation can be used for forecasting the LCA and damage cost for all 15 impacts and to conduct the statistical modelling. The same calculation was conducted for the previous years, from 2013 to 2017, as available in Appendix 7.

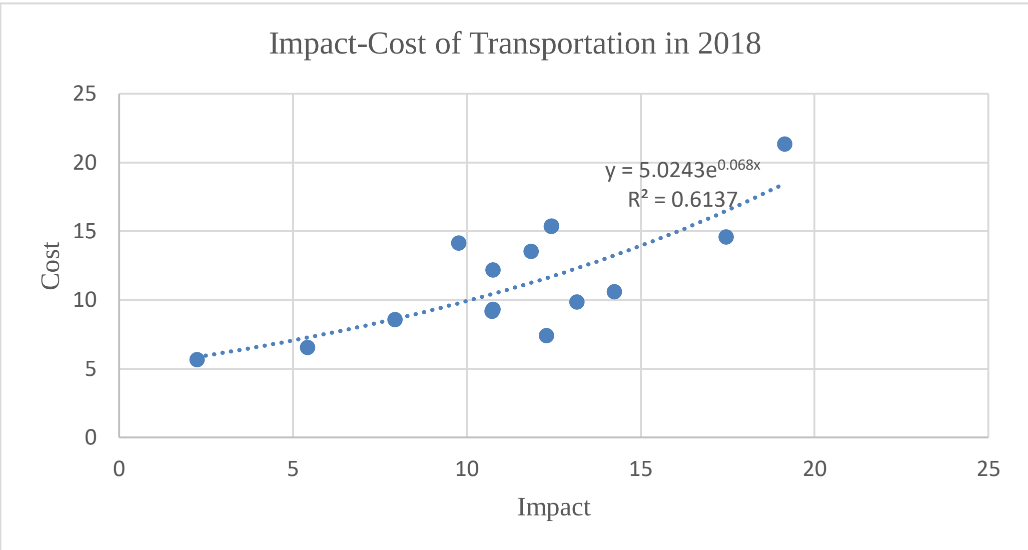


Figure 6.19: The relationship between impact and cost for transportation method in 2018

Figure 6.19 shows the results of LCA and damage cost for the transportation method in 2018 by using the natural logarithm equation. The results consist of 15 impacts on LCA that are linked with damage cost. There is a positive relationship between the LCA impact and damage cost for transportation method in 2018 based on the exponential curve; it shows that $R^2 = 0.614$ and the equation for the impact cost of transportation: $y = 5.024e^{0.068x}$

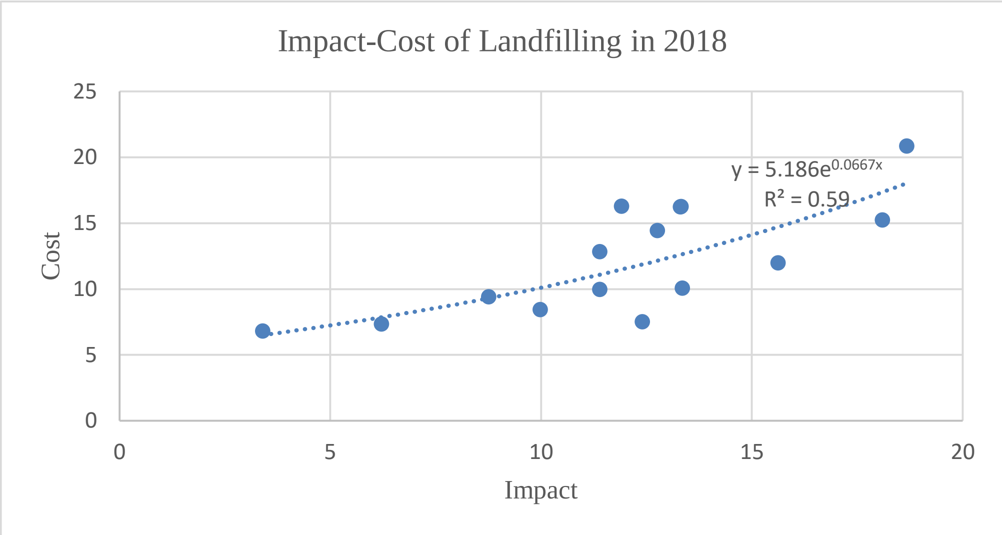


Figure 6.20: The relationship between impact and cost for the landfilling method in 2018

Figure 6.20 illustrates the outcomes of the landfilling method for concrete waste in LCA and damage cost by calculation of the natural logarithm equation, in which the results cover 15 impacts for LCA and damage cost. The mathematical relationship between the LCA impact and damage cost for the landfilling method for concrete waste have a positive relationship, in which as the impact increases the damage cost increases, while the exponential curve of $R^2 = 0.59$ with the equation $y = 5.186e^{0.0667x}$

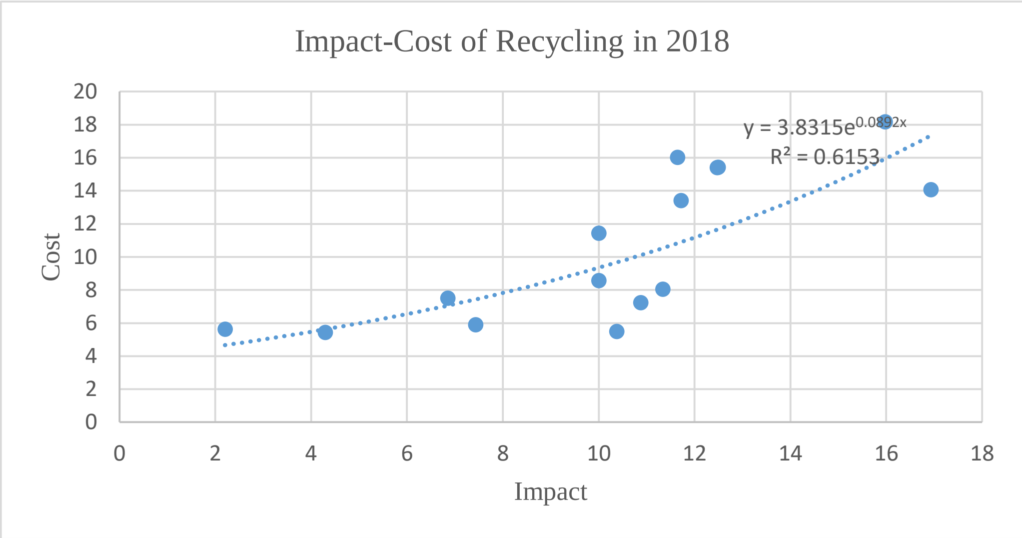


Figure 6.21: The relationship between impact and cost for recycling method in 2018

Figure 6.21 demonstrates the recycling method for concrete waste in terms of LCA impact and damage cost, which is based on the natural logarithm equation. Moreover, the outcomes indicate 15 impacts, which have a positive relationship at curve $R^2 = 0.615$ and the equation $y = 3.832e^{0.0892x}$; when the impact increases, the cost increases also.

6.9. Summary

The chapter concludes the LCIA by using ReCiPe2016 of 18 impacts for midpoint damage, 22 impacts for endpoint damage and three endpoint areas of protections. The results were based on three methods, which are final disposal of concrete waste without any treatment compared to recycling method in parallel with the transportation process of concrete waste to the landfill. Furthermore, impact results for all nine damage pathways and three endpoint areas of protections are negatively affected more in the final disposal method, then the transportation process and, finally, the recycling method has less impact. On the other hand, as can be seen from the damage cost results for the three methods of concrete waste disposal, the damage cost for putting concrete waste into landfill was found to be the highest in all midpoint impacts compared to the method of recycling and the transportation process. Furthermore, the recycling and landfilling methods had high damage cost in some impacts compared to the transportation method, for instance, Ozone Formation in Human Health, Fine Particulate Matter Formation and Ozone Formation in Terrestrial Ecosystems, while the transportation method had a higher damage cost than the landfilling and recycling methods in Terrestrial Ecotoxicity and Human Carcinogenic Toxicity. Furthermore, based on Descriptive Analysis for damage cost for 2018, transportation method has a high value compared to landfilling and recycling methods in Mean, Standard Deviation, Kurtosis and Skewness. The mathematical relationships by natural logarithm equation for LCA and damage cost of the three concrete waste management methods were found to be near to each other in terms of R^2 results. Moreover, the recycling method has more correlation between impact and damage cost compared to landfilling and transportation. On the other hand, the recycling method seems to be the best in terms of LCA results and damage cost compared to the other concrete waste management options. Although, mathematical relationships between LCA and damage cost show that there is always a positive relationship between impact and cost.

CHAPTER 7: DISCUSSION

7.1. Introduction

This chapter includes a discussion of the key research aim and objectives through this thesis. The first section introduces a discussion of the research's three objectives, which is followed by a discussion of the strengths of the research methodology, implications and validation. Furthermore, the last section discusses the thesis conclusion, which covers research limitation, summary of the research contribution, future research and recommendations.

7.2. Main Findings and Summary of Contribution

7.2.1. Objective 1: Investigate the environmental impact of concrete waste by LCA

The first objective was to conduct the LCA of three waste management options, landfilling, recycling and transportation. LCA was conducted in this study based on ISO14040:2006 Environmental Management of Life Cycle Assessment: Principles and Framework. The LCA consists of four phases, which are goal and scope definition, inventory analysis, impact assessment and, finally, interpretation of results. Furthermore, the phases should be conducted sequently (ISO14040:2006). Methods for management of concrete waste were classified into two main phases, for instance, LCA by using SimaPro software and EcoInvent database. The LCA approach was used to find the ecological impact for the management of concrete waste as per ReCiPe2016 LCIA, which consists of 18 midpoint 18 impacts, 22 endpoint impacts and three endpoint areas of protections.

To achieve the objectives of the investigation, the environmental impact of concrete waste was examined by LCA, using ReCipe2016 LCIA, which includes midpoint and endpoint. Midpoint results show that the damage cost for putting concrete waste into landfill was found to be the highest in all midpoint impacts compared to the method of recycling and transportation process. Furthermore, recycling and landfilling methods had a higher damage cost in some impacts compared to the transportation method, for instance, Ozone Formation (Human health), Fine Particulate Matter Formation and Ozone formation in Terrestrial Ecosystems. In contrast, the transportation method had a higher damage cost than landfilling and recycling methods in Terrestrial Ecotoxicity and Human Carcinogenic Toxicity.

On the other hand, the endpoint results show that concrete waste as final disposal has more impact on Human Health at 103.189 Disability-Adjusted Life Years (DALY) compared to transportation of concrete waste and recycling of concrete waste, which were 15.155 and 74.185 respectively. Moreover, the landfilling method had the highest impact on ecosystems at 0.251 Species-Years (species.yr), while transportation process was 0.100 and recycling method was 0.074. Moreover, landfilling cost more in terms of Resources (USD2013), which was $13,646,268, compared to transportation of concrete waste and recycling of concrete waste, which were $5,409,795.1 and $3,197,674.1 respectively.

7.2.2. Objective 2: Investigate the Environmental Damage Cost of Concrete Waste

The second objective was to find the damage cost of the three waste management by referring to LCA, specifically LCIA results. Monetisation is a process that signifies the impacts linked with social and natural capital in monetary value, which makes them more tangible. In addition, monetising environmental influences is known as natural capital valuation, which is considered to be a kind of sustainability return on investment analysis and delivers a different financial perception of sustainability (Pre-sustainability 2018, p. 1). The Handbook Environmental Prices (2017) was used as a method and cost for valuation of ecological impacts by CE Delft (2018). Moreover, the handbook contains the environmental prices of many impacts. In addition, they are applicable for social cost or pollution, and are expressed in euros per kilogram pollutant (Handbook Environmental Prices, 2017). The assessment of damage cost used the midpoint damage prices applicable for the LCIA midpoint as per the ReCipe2016 method.

Chapter 6 presented the damage cost results and LCA results, which indicate that many impacts have a serious damage cost on the environment, such as Ozone Formation (Human Health), which shows an increase in yearly average at €200,000 to €900,000 in all three methods. In addition, Global Warming shows a huge damage cost in all methods and especially in putting concrete waste into landfill, which cost €4,120,242.50. Moreover, Ozone formation (Terrestrial Ecosystems) in the landfilling method was €11,446,760.69 compared with the transportation process and recycling method, which were €4,711,544.29 and €4,979,775.78 respectively. The highest damage cost was Terrestrial Ecotoxicity in the landfilling method, which reached €1,833,140,313.90, which shows significant damage compared to landfilling and recycling methods, which are €1,147,140,619.10 and €77,395,172.74 respectively. On the other hand, Terrestrial Ecotoxicity had more impact on the landfilling method, which cost €1,881,462.17 compared to transportation at €753,609.10 and recycling at €663,544.93. On the other hand, some impacts had a low damage cost, such as Stratospheric Ozone Depletion, Human Carcinogenic Toxicity (Cancer), Freshwater Ecotoxicity, Freshwater Eutrophication, Land Use, Marine Ecotoxicity and Marine Eutrophication.

7.2.3. Objective 3: Find the Mathematical Relationship between LCA Results and Damage Cost Results

The relationships between many stages are involved in environmental prices as per the Handbook Environmental Prices (2017), which consists of emissions, midpoints, endpoints, valuation and related fields of study such as ReCiPe LCIA, which was used in this research.

The natural logarithm equation was used in Microsoft Excel 2013 for each method and for 15 LCA and damage cost impacts, which are Climate change, Ozone layer depletion, Acidification, Freshwater eutrophication, Marine eutrophication, Land use, Terrestrial ecotoxicity, Freshwater ecotoxicity, Marine ecotoxicity, Human toxicity, PM2.5, Nitrogen oxides (Nox) (Human health), Nitrogen oxides (Nox) (Terrestrial ecosystems), Mineral resource scarcity (Atmospheric) and Mineral resource scarcity (Soil). In addition, the exponential curve was found to be the appropriate curve with R^2 compared to the linear curve. Furthermore, mathematical relationships between LCA and damage cost show that there is always a positive relationship between impact and cost. Moreover, the natural logarithm equation can be used for forecasting the LCA and damage cost for 15 impacts of the three concrete waste methods.

CHAPTER 8: CONCLUSION

8.1. Strengths of the Research Methodology

There are strengths in the research methodology that led to the research aim and objectives being accomplished, for example, modelling of concrete waste by LCA and damage cost. Moreover, database for the quantity of concrete waste in Dubai city landfill from government sources was achieved. In addition, the SimaPro software and EcoInvent database were used in this research.

On the other hand, the efficient literature review of various studies showed that significant ecological impacts occur from construction using and demolition of concrete waste. In addition, the serious risks of concrete waste were highlighted in relation to human health, ecosystems and resources availability, which were analysed in this study by conducting LCA based on ReCipe2016 LCIA to achieve the research objectives. The LCA shows the results of the ecological impacts of concrete waste in the landfilling method, transportation of concrete waste and recycling of concrete waste by 18 midpoint impacts, 22 endpoint impacts and three final endpoint impacts of protection.

The importance of the environmental impact results from LCA is to measure the damage that concrete waste causes to the environment, find the best waste management practices and options, and find the damage cost, which all lead to protect and save the environment, resources, human health and ecosystems. Furthermore, calculating the ecological impact and damage cost can be identified at the early stage of construction and demolition processes to reduce and minimise the generated waste concrete by selecting the Best Disposal Option Practice (BDOP). For example, the recycling method is an eco-friendly method that is recommended worldwide, which helps to protect the environment and reduce the damage cost compared to the landfilling method.

Methods for management of concrete waste were classified into two main phases: the first phase used LCA by using SimaPro software and EcoInvent database, while the second phase examined monetisation of environmental indicators by referring to the Handbook Environmental Prices 2017. The LCA approach was used to find the ecological impact for the management of concrete waste as per ReCiPe2016 LCIA, which consists of 18 impacts as illustrated in detail in Chapter 5. Furthermore, the monetisation of environmental indicators was based on 15 environmental prices, which were applicable for ReCiPe2016 LCIA, and which were used to find the damage cost of 15 impacts that have an effect on the environment.

LCA of waste concrete in this study shows that it has huge environmental impacts and damage costs in relation to human health, ecosystems and resources throughout its waste life cycle, which includes emissions from dump truck transportation, dismantling, handling and disposal. Additionally, the process of depositing concrete waste into landfill without any treatment method has a significant influence. For example, emissions released from concrete waste in the landfill can contaminate soil and water, as mentioned in detail in the LCA results in Chapter 6. On the other hand, recycling concrete waste also contributes to environmental impacts and damage costs, but it has fewer impacts compared to the landfilling method. In addition, LCA results and damage costs show that recycling of concrete waste does less damage compared to landfilling and transportation.

Based on the literature review, recycling of concrete waste has always been preferential in terms of reducing waste and achieving sustainability. However, in terms of LCA and damage cost, it was not clearly evident how the millions of tonnes of concrete can influence the environment based on 18 impacts as midpoint and endpoint methods and damage costs for 15 impacts. In addition, there are many findings in different pieces of research which show that waste concrete has ecological impacts, but when it comes to the damage cost to human health, resources and ecosystems and the certain huge quantity of concrete waste and different waste management methods, this research explores the results in more detail. Although, the damage cost results for the three methods of concrete waste disposal show that the damage cost for putting concrete waste into landfill is the highest in all midpoint impacts compared to the method of recycling and transportation process. Furthermore, the recycling and landfilling methods have a high damage cost in some impacts compared to the transportation method, for instance, Ozone Formation in Human Health, Fine Particulate Matter Formation and Ozone Formation in Terrestrial Ecosystems, while the transportation method had a higher damage cost than the landfilling and recycling methods in terms of Terrestrial Ecotoxicity and Human Carcinogenic Toxicity. The Descriptive Analysis for damage cost for 2018 shows that the transportation method has a high value compared to the landfilling and recycling methods in Mean, Standard Deviation, Kurtosis and Skewness.

8.2. Validation and Implications

LCA was conducted in this study based on ISO14040:2006 Environmental Management of Life Cycle Assessment: Principles and Framework. The LCA consists of four phases, which are goal and scope definition, inventory analysis, impact assessment and, finally, interpretation of results. Furthermore, the phases should be conducted sequently (ISO14040:2006). Moreover, methods for management of concrete waste were classified into two main phases, for instance, LCA by using SimaPro software and EcoInvent database. The Handbook Environmental Prices (2017) was used as a method and cost for valuation of ecological impacts by CE Delft (2018). Moreover, the handbook contains the environmental prices of many impacts. In addition, environmental prices are applicable for social cost or pollution, which are expressed in euros per kilogram pollutant (Handbook Environmental Prices, 2017). The sustainability of the environment, along with social and economic sustainability, creates three pillars of sustainability (Moldan et al., 2011).

Concrete waste management options such as final disposal in landfill is the primary method used in the UAE in general, and the city of Dubai, while concrete waste management options such as recycling are not practised here. Moreover, it was discovered that the current concrete waste disposal method practised in the UAE including the city of Dubai was not an efficient waste disposal method. This is because both LCA and damage cost outcomes show that generation of concrete waste from C&D sites in the city of Dubai, which is sent to landfill without any treatment method, is considered not substantial process and has a massive impact on the environment.

Referring to the EcoInvent database and SimaPro software in this study: SimaPro provides a collaborative outcomes analysis, which helps to trace the results back to their origins in the real period (Peuportier et al. 2009). SimaPro LCA software and EcoInvent LCA database were used in a study of the environmental evaluation of concrete, plaster and brick components' manufacture (Giama & Papadopoulos, 2015). Furthermore, SimaPro provides a native user boundary by following ISO14040, and it can be used to create comprehensive modelling with scenario analysis (Peuportier et al. 2009). Moreover, the SimaPro tool helps to find direct impact assessment results from every stage of the studied system and it can analyse composite waste treatment and recycling methods (Peuportier et al. 2009).

To validate the results, for both LCA and damage cost of the three methods of concrete waste, they were studied in relation to other researchers' findings. In this study, the relationship between LCA and damage cost by the exponential curve was found to be the appropriate curve with R^2 compared to the linear curve, which shows that the recycling method has more correlation between impact and damage cost compared to landfilling and transportation. Although, mathematical relationships between LCA and damage cost show that there is always a positive relationship between impact and cost. Moreover, there was no error in LCA in terms of inventory and analysis of all three methods in this study by referring to the EcoInvent database and SimaPro software, since the modelling was run based on collected data.

8.3. Summary of Contribution

This research has contributed to increase the understanding of modelling the damage cost besides LCA results for the management of concrete waste, which were based on different waste management options: landfilling, recycling and transportation process. On the other hand, it has made a contribution to the existing body of knowledge in terms of both research and practice in the following areas:

1) In terms of academia and research, the study will open up opportunities for researches in the future, for example, regarding which impact indicator and method of concrete waste impacts on the environment and which indicator is more or less critical in terms of ecological impact and damage cost of management of concrete waste, which has not previously been available in the region.

2) Life Cycle Cost Assessment (LCCA) can be conducted in future to find the financial benefit of different management options for concrete waste. Since the study is based on LCA and environment damage cost, a fully comprehensive LCCA can be conducted in future research, which has implications for academia. Furthermore, LCCA will be an opportunity for the whole construction project in terms of cost saving and other financial purposes. In addition, this study is only the first phase, and it is opening opportunities for other researchers to follow this research modelling, and then to be able to conduct phase two. It is impossible to conduct LCCA without establishing the LCA outcome. Moreover, no study has undertaken LCA and monetisation of environmental indicators in the region in the management of concrete waste by considering the massive quantity in landfill with comparison to the

recycling method and transportation process. Finally, the study will open opportunities to achieve sustainability in LCA of the management of concrete waste and LCCA.

3) In terms of practice, the study will help the construction projects, whether run by the government or private organisations, to achieve sustainability and efficiency in the management of concrete waste by having a mobile or stationary recycling machine at the construction sites, which will reduce the ecological impacts on the environment as concrete debris will not be transported to or end up in landfill.

4) The study will help the implication in areas of concrete waste disposal and management of concrete waste in line with the requirements of government and federal legislation such as Dubai Plan 2021 and UAE Vision 2021. Moreover, the study is going to enhance the decision-making since it is built and based on tangible benefits, for example, UAE 2021 Vision, which mandating and utilising the concrete waste to achieve sustainability. Furthermore, the primary practice implication of this study is to reduce the ecological impact and damage cost of concrete waste, while the secondary implication is providing future capacity in the landfill, reducing the trips per truck in the dumping of concrete waste into the landfill, achieving income from the recycling method, and reducing the ecological impacts and damage costs to human health, ecosystems and resources.

8.4. Limitations of the Research

Comprehensive research has some limitations in fulfilling the research aim and objectives. Some limitations were found in this research, and supplementary development can be made as follows:

1) The raw data and segregation of the quantity of concrete waste in the landfill were limited to certain years, which had an influence on the study in terms of finding the yearly amount of unreinforced concrete waste in the landfill.

2) The investigation into the current methods of concrete waste management options in the UAE differed in some cities, which found that recycling plants for concrete waste are available in some cities in the United Arab Emirates, but this method is not in used for recycling concrete waste.

3) Conducting LCA by using SimaPro software required an inventory database for a specific region, which might have some variations in LCIA results, but some global inventories were found in the EcoInvent database.

4) Many damage prices were found from different resources, but the appropriate and validated damage prices that meet the ReCip2016 LCIA of this research were in a Dutch database included in the Handbook Environmental Prices 2017, which was used as method and cost for valuation of ecological impacts by the CE Delft organisation.

5) The Handbook Environmental Prices 2017 was found to have limited damage prices, which were available for only 15 impacts out of 18 impacts. The damage prices for Fossil Resources Scarcity, Ionising Radiation and Human Non-Carcinogenic Toxicity were not available in the handbook, so damage cost was not conducted for these three impacts.

8.5. Recommendations for Future Research

This investigation identified some areas that would benefit from future research, for example:

1) Conduct a Life Cycle Cost Assessment (LCCA) on concrete waste management to find the financial benefit of different management options for concrete waste.

2) Conduct a full LCA on different management methods for concrete waste, starting from the extraction of raw materials, to the processing, the final product and waste treatment methods.

3) Develop an LCA cost model for different treatment methods of concrete waste.

4) Redevelop the model of concrete waste management in the UAE by including different scenarios of recycling certain quantities of concrete waste and find the LCA and damage cost results.

5) Incorporate more factors in finding the damage cost of concrete waste.

REFERENCES & BIBLIOGRAPHY

(2018). *Beeah.ae* [online]. [Accessed 20 April 2018]. Available at: https://beeah.ae/en

(2018). *Global Carbon Atlas* [online]. [Accessed 20 April 2018]. Available at: http://globalcarbonatlas.org/en/CO2-emissions

(2018). *Lc-impact.eu* [online]. [Accessed 25 April 2018]. Available at: http://www.lc-impact.eu/methodology-home

Abdelhamid, M. (2014). Assessment of different construction and demolition waste management approaches. *HBRC Journal*, vol. 10 (3), pp. 317-326.

Abrelpe - Associação Brasileira de Empresas de Limpeza Pública e Resíduos Especiais". (2018). [Accessed 19 April 2018]. Available at: http://www.abrelpe.org.br

ACI Committee (2005). Building code requirements for structural concrete (ACI 318-05) and commentary (ACI 318R-05). *American Concrete Institute*.

Ahlroth, S., Nilsson, M., Finnveden, G., Hjelm, O. & Hochschorner, E. (2011). Weighting and valuation in selected environmental systems analysis tools – suggestions for further developments. *Journal of Cleaner Production*, vol. 19 (2-3), pp. 145-156.

Akpinar, P. & Khashman, A. (2017). Intelligent classification system for concrete compressive strength. *Procedia Computer Science*, vol. 120, pp. 712-718.

Amaratunga, D., Haigh, R., Kulatunga, U., & Rameezdeen, R. (2006). Management of Environmental Quality: An International Journal. Retrieved fromhttp://www.emeraldinsight.com/journals.htm?articleid=1537410&show=html.

Anenberg, SC., Horowitz, LW., Tong, DQ., West, JJ. (2010) An Estimate of the Global Burden of Anthropogenic Ozone and Fine Particulate Matter on Premature Human Mortality Using Atmospheric Modelling. *Environmental Health Perspectives* vol. 118 (9), pp. 1189-1195.

Arslan, H., Coşgun, N. & Salgın, B. (2012). Construction and Demolition Waste Management in Turkey. pp. 313-332.

Arulrajah, A., Mohammadinia, A., D'Amico, A. & Horpibulsuk, S. (2017). Effect of lime kiln dust as an alternative binder in the stabilization of construction and demolition materials. *Construction and Building Materials*, vol. 152, pp. 999-1007.

Ashmore, MR (2005). Assessing the future global impacts of ozone on vegetation. *Plant Cell and Environment*, vol. 28 (8), pp. 949-964.

Aşıcı, A.A. (2013). Economic growth and its impact on environment: A panel data analysis. *Ecological indicators*, vol. 24, pp. 324-333.

Asif, M., Muneer, T. & Kelley, R. (2007). Life cycle assessment: A case study of a dwelling home in Scotland. *Building and Environment*, vol. 42 (3), pp. 1391-1394.

Assamoi, B. & Lawryshyn, Y. (2012). The environmental comparison of landfilling vs. incineration of MSW accounting for waste diversion. *Waste Management*, vol. 32 (5), pp.1019-1030.

Azevedo, L. B., van Zelm, R., Elshout, P. M. F., Hendriks, A. J., Leuven, R. S. E. W., Struijs, J., de Zwart, D., Huijbregts, M. A. J. (2013b). Species richness–phosphorus relationships for lakes and streams worldwide. *Global Ecology and Biogeography*, vol. 22 (12): pp. 1304-1314.

Azevedo, LB. (2014). Development and application of stressor – response relationships of *nutrients*, vol. 8. *Ph.D. Dissertation, Radboud University Nijmegen, the Netherlands*. [Accessed 18 April 2019]. Available at: http://repository.ubn.ru.nl

Azevedo, LB., Henderson, AD., van Zelm, R., Jolliet, O., Huijbregts, MAJ. (2013a). Assessing the Importance of Spatial Variability versus Model Choices in Life Cycle Impact Assessment: The Case of Freshwater Eutrophication in Europe. *Environmental Science & Technology*, vol. 47 (23), pp. 13565-13570.

Azevedo, LB., Van Zelm, R., Elshout, PMF., Hendriks, AJ., Leuven, RSEW., Struijs, J., de Zwart, D., Huijbregts, MAJ. (2013b). Species richness–phosphorus relationships for lakes and streams worldwide. *Global Ecology and Biogeography*, vol. 22 (12), pp. 1304-1314.

Azevedo, LB., Van Zelm, R., Hendriks, AJ., Bobbink, R., Huijbregts, MAJ. (2013c). Global assessment of the effects of terrestrial acidification on plant species richness. *Environmental Pollution*, vol. 174 pp. 10-15.

Babak, N. (2017). Transport Construction Negative Impact on the Environment. *Procedia Engineering*, vol. 189, pp. 867-873.

Bahr, B., Hanssen, O.J., Vold, M., Stoltenberg-Hansson, E. & Steen, B. (2003). Experiences of environmental performance evaluation in the cement industry. Data quality of environmental performance indicators as a limiting factor for benchmarking and rating, *Journal of Cleaner Production*, vol. 11, pp.713-725.

Bakshan, A., Srour, I., Chehab, G., El-Fadel, M. & Karaziwan, J. (2017). Behavioral determinants towards enhancing construction waste management: A Bayesian Network analysis. *Resources, Conservation and Recycling*, vol. 117, pp. 274-284.

Barbieri, J. C., & Cajazeira, J. E. R. (2009). Avaliação do ciclo de vida do produto como instrumento de gestão da cadeia de suprimento – o caso do papel reciclado. *Simpoi*, pp. 1-16.

Bare J.C., Norris G.A., D.W., P. and McKone T. (2003). "TRACI - The Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts." Journal of Industrial Ecology. 6(3-4): 49-78.

Beckenbach, F., Hampicke, U., Schulz, W. (1998): Möglichkeiten und Grenzen der Monetarisierung von Natur und Umwelt. *Possibilities and Limitations of Monetization of Nature and Environment*, pp. 3-18.

Bell, ML., Dominici, F., Samet, JM. (2005) A meta-analysis of time-series studies of ozone and mortality with comparison to the national morbidity, mortality, and air pollution study. *Epidemiology*, vol. 16 (4), pp. 436-445.

Bernstein, P. (2014). *Quantifying Sustainability II – Natural Capital and Monetisation*. [Accessed 18 April 2019]. Available at: https://www.pre-sustainability.com/news/quantifying-sustainability-ii-natural-capital-and-monetisation.

Bey, I., Jacob, D., Yantosca, R., Logan, J., Field, B., Fiore, A. (2001). Global modelling of tropospheric chemistry with assimilated meteorology: model description and evaluation. *Journal of Geophysical Research*, vol. 106, pp. 23073–905.

Bickel, P., Friedrich, R.. (2005). Externalitites of Energy-Methodology 2005 Update. *European Commission Directorate-General for Research Sustainable EnergySystems (EUR 21951), Brussels.*

Björklund, T. and Tillman, A-M. (1997) *LCA of Building Frame Structures – Environmental Impact Over the Life Cycle of Wooden and Concrete Frames*, Technical Environmental Planning, Chalmers University of Technology.

Björklund, T., Jönsson, Å. & Tillman, A-M. (1996). *LCA of Building Frame Structures – Environmental Impact Over the Life Cycle of Concrete and Steel Frames*, Technical Environmental Planning, Chalmers University of Technology.

Bohne, R., Brattebø, H. & Bergsdal, H. (2008). Dynamic Eco-Efficiency Projections for Construction and Demolition Waste Recycling Strategies at the City Level. *Journal of Industrial Ecology*, vol. 12 (1), pp. 52-68.

Bonnieux, F., Carpentier, A., Weaver, R. (1998). Reducing soil contamination: economic incentives and potential benefits. *Agriculture, ecosystems & environment*, vol. 67 (2-3), pp.275-288.

Bos, E.J., Vleugel, J.M. (2005). Incorporating nature valuation in cost-benefit analysis.In: Tiezzi, E., Brebbia, C.A., Jorgensen, S.E., Almorza Gomar, D. (Eds.), *Ecosystems and Sustainable Development V. WIT Press, Southampton*, pp. 281–290.

Bossink, B. & Brouwers, H. (1996). Construction Waste: Quantification and Source Evaluation. *Journal of Construction Engineering and Management*, vol. 122 (1), pp. 55-60.

Boulatoff, C., Jenkins, M. (2010). Long-term nexus between openness, income and environmental quality. *International advances in economic research*. Vol. 16 (4), pp. 410–418.

Bouwman, AF., Beusen, AHW., Billen, G. (2009) Human alteration of the global nitrogen and phosphorus soil balances for the period 1970-2050. *Global Biogeochemical Cycles 23*.

Bovea, M. & Powell, J. (2016). Developments in life cycle assessment applied to evaluate the environmental performance of construction and demolition wastes. *Waste Management*, vol. 50, pp. 151-172.

Bravo, M., Brito, J. & Evangelista, L. (2017). Thermal Performance of Concrete with Recycled Aggregates from CDW Plants. *Applied Sciences*, vol. 7 (7), p. 740.

Burnett, RT., Pope, CA., Ezzati, M., Olives, C., Lim, SS., Mehta, S., Shin, HH., Singh, G., Hubbell, B., Brauer, M., Anderson, HR., Smith, KR., Balmes, JR., Bruce, NG., Kan, H., Laden, F., Pruess-Ustuen, A., Turner, MC., Gapstur, SM., Diver, WR., Cohen, A. (2014) An Integrated Risk Function for Estimating the Global Burden of Disease Attributable to Ambient Fine Particulate Matter Exposure. *Environmental Health Perspectives*, vol. 122 (4), pp. 397-403.

Butera, S., Christensen, T. H. & Astrup, T. F. (2015). Life cycle assessment of construction and demolition waste management. *Waste Management*, vol. 44, pp. 196-205.

Chen, T., Burnett, J. & Chau, C. (2001). Analysis of embodied energy use in the residential building of Hong Kong. *Energy*, vol. 26 (4), pp. 323-340.

Cheng, J. & Ma, L. (2013). A BIM-based system for demolition and renovation waste estimation and planning. *Waste Management*, vol. 33 (6), pp. 1539-1551.

CIESIN (2005). Gridded Population of the World, Version 3 (GPWv3): Population Density Grid, Future Estimates. Palisades, NY: NASA Socioeconomic Data and Applications Center (SEDAC). *Center for International Earth Science Information Network - CIESIN -*

Coelho, A. & Brito, J. de. (2013). Environmental analysis of a construction and demolition waste recycling plant in Portugal - Part I: Energy consumption and CO2 emissions. *Waste Management*, vol. 33 (5), pp. 1258–1267.

Coelho, A. & de Brito, J. (2013). Economic viability analysis of a construction and demolition waste recycling plant in Portugal – part I: location, materials, technology and economic analysis. *Journal of Cleaner Production*, vol. 39, pp. 338-352.

Collins, F. & Sanjayan, J. (2002). The Challenge of the Cement Industry Towards the Reduction of Greenhouse Emissions. *IABSE Symposium Report*, vol. 86 (6), pp. 1-11.

Columbia University, and Centro Internacional de Agricultura Tropical - CIAT. [Accessed 18 April 2019]. Available at: http://dx.doi.org/10.7927/H4ST7MRB.

Cosme N., Hauschild, MZ. (2016). Effect Factors for marine eutrophication in LCIA based on species sensitivity to hypoxia. *Ecological Indicators*, vol. 69, pp. 453-462.

Cosme, N., Hauschild, MZ. (2016). Effect Factors for marine eutrophication in LCIA based on species sensitivity to hypoxia. *Ecological Indicators*, vol. 69, pp. 453-462.

Cosme, N., Hauschild, MZ. (2017). Characterization of waterborne nitrogen emissions for marine eutrophication modelling in life cycle impact assessment at the damage level and global scale. *International Journal of Life Cycle Assessment*, vol. 22, pp. 1558-1570.

Cosme, N., Hauschild, MZ. (2017). Characterization of waterborne nitrogen emissions for marine eutrophication modelling in life cycle impact assessment at the damage level and global scale. *International Journal of Life Cycle Assessment*, vol. 22, pp. 1558-1570.

Cosme, N., Jones, MC., Cheung, WWL., Larsen, HF. (2017a). Spatial differentiation of marine eutrophication damage indicators based on species density. *Ecological Indicators*, vol. 73, pp. 676-685.

Cosme, N., Koski, M., Hauschild, MZ. (2015). Exposure factors for marine eutrophication impacts assessment based on a mechanistic biological model. *Ecological Modelling*, vol. 317, pp. 50-63.

Cosme, N., Mayorga, E., Hauschild, MZ. (2017b). Spatially explicit fate factors of waterborne nitrogen emissions at the global scale. *International Journal of Life Cycle Assessment*.

Crowson, P. (2003). Mine size and the structure of costs. *Resources Policy*, vol. 29 (1-2), pp. 15-36.

Curran, M., Hellweg, S., Beck, J. (2014). Is there any empirical support for biodiversity offset policy? *Ecological Applications*, vol. 24 (4), pp. 617- 632.

Curran, M., Hellweg, S., Beck, J. (2014). Is there any empirical support for biodiversity offset policy? *Ecological Applications*, vol. 24 (4), pp. 617- 632.

Dahlbo, H., Bachér, J., Lähtinen, K., Jouttijärvi, T., Suoheimo, P., Mattila, T., Sironen, S., Myllymaa, T. & Saramäki, K. (2015). Construction and demolition waste management – a holistic evaluation of environmental performance. *Journal of Cleaner Production*, vol. 107, pp. 333-341.

Dahlbo, H., Bachér, J., Lähtinen, K., Jouttijärvi, T., Suoheimo, P., Mattila, T. & Saramäki, K. (2015). Construction and demolition waste management - A holistic evaluation of environmental performance. *Journal of Cleaner Production*, vol. 107, pp. 333-341.

Dajadian, S. & Koch, D. (2014). Waste Management Models and Their Applications on Construction Sites. *International Journal of Construction Engineering and Management*, vol. 3 (3), pp. 91-98.

Danai, G. I., & Panos, K. (2012). Survey Regarding Control and Reduction of Construction Waste. Retrieved from http://www.plea2012.pe/pdfs/T05-20120130-0006.pdf.

Das, A. & Swamy, A. (2014). Reclaimed Waste Materials in Sustainable Pavement Construction. *Climate Change, Energy, Sustainability and Pavements, Green Energy and Technology*, pp. 419-438.

Davidsen, C. & Fischer, C. (2010). Europe as a Recycling Society. European Recycling Policies in relation to the actual. *European Topic Centre on Sustainable Consumption and Production.*

Davis, M., de la Rosa, A., Weldon, B., Kurama, Y.C. & McGinnis, M.J. (2016). Environmental considerations of recycled concrete aggregates (RCA) for improved sustainability of reinforced concrete building structures, Proc. of SCMT 4, Las Vegas, USA, 2016, August.

De Baan, L, Alkemade, R., Köllner, T. (2013). Land use impacts on biodiversity in LCA: A global approach. *International Journal of Life Cycle Assessment*, vol. 18 (6), pp. 1216-1230.

De Baan, L., Alkemade, R., Köllner, T. (2013). Land use impacts on biodiversity in LCA: A global approach. *International Journal of Life Cycle Assessment*, vol. 18 (6), pp. 1216-1230.

de Magalhães, R., Danilevicz, Â. & Saurin, T. (2017). Reducing construction waste: A study of urban infrastructure projects. *Waste Management*, vol. 67, pp. 265-277.

De Schryver, AM., Brakkee, KW., Goedkoop, M., Huijbregts, MAJ. (2009) Characterization factors for global warming in life cycle assessment based on damages to humans and ecosystems. *Environmental Science & Technology*, vol. 43 (6), pp. 1689-1695.

De Schryver, AM., Brakkee, KW., Goedkoop, M., Huijbregts, MAJ. (2009). Characterization factors for global warming in life cycle assessment based on damages to humans and ecosystems. *Environmental Science & Technology*, vol. 43 (6), pp. 1689-1695.

De Schryver, AM., Van Zelm, R., Humbert, S., Pfister, S., McKone, TE., Huijbregts, MAJ. (2011). Value choices in life cycle impact assessment of stressors causing human health damage. *Journal of Industrial Ecology*, vol.15 (5), pp. 796–815.

De Schryver, AM., Van Zelm, R., Humbert, S., Pfister, S., McKone, TE., Huijbregts, MAJ. (2011). Value choices in life cycle impact assessment of stressors causing human health damage. *Journal of Industrial Ecology*, vol. 15 (5), pp. 796–815.

Derwent, RG., Jenkin, ME. (1991). Hydrocarbons and the long-range transport of ozone and PAN across Europe. *Atmospheric Environment*, vol. 25A (8), pp. 1661-1678.

Derwent, RG., Jenkin, ME., Passant, NR., Pilling, MJ. (2007a). Photochemical ozone creation potentials (POCPs) for different emission sources of organic compounds under European conditions estimated with a Master Chemical Mechanism. *Atmospheric Environment*, vol. 41, pp. 2570- 2579.

Derwent, RG., Jenkin, ME., Passant, NR., Pilling, MJ. (2007b). Reactivity-based strategies for photochemical ozone control in Europe. *Environmental Science & Policy*, vol. 10, pp. 445-453.

Derwent, RG., Jenkin, ME., Saunders, SM., Pilling, MJ. (1998). Photochemical ozone creation potentials for organic compounds in northwest Europe calculated with a master chemical mechanism. *Atmospheric Environment*, vol. 32 (14/15), pp. 2429-2441.

DIN Deutsches Institut für Normung e.V, "DIN EN ISO 14040: Environmental management - Life cycle assessment - Principles and framework," 2009.

DIN Deutsches Institut für Normung e.V, "DIN EN ISO 14044: Environmental management - Life cycle assessment - Requirements and guidelines," 2006.

Directive 2002/91/EC Energy Performance of Buildings, [Accessed 18 April 2018]. Available at: http://eur-lex.europa.eu/ LexUriServ/LexUriServ.do?uri¼OJ:L:2003:001:0065:0065:EN:PDF. 04/04/2014.

DIRECTIVE 2008/98/EC of the EUROPEAN PARLIAMENT and of the COUNCIL of 19 November 2008 on Waste and Repealing Certain Directives 312". (2008). pp. 3-30.

Döll, P., Siebert, S. (2002). Global Modelling of irrigation water requirements. *Water Resources Research*, vol. 38 (4), pp. 1037.

Dosal, E., Galán, B., Andrés, A. & Viguri, J. (2013). Introduction of social criteria for the optimal location of Construction and Demolition Waste management facilities in Cantabria (Spain). *Computer Aided Chemical Engineering*, vol. *32*, pp. 1027-1032.

Dreicer, M., Tort, V. Manen, P. (1995). ExternE, Externalities of Energy, Nuclear, Centre d'étude sur l'Evaluation de la Protection dans le domaine Nucléaire (CEPN), edited by the European Commission DGXII, S*cience, Research and Development JOULE, Luxembourg*, vol. 5.

Dreyer, L. C., Niemann, A. L. & Hauschild, M. Z. (2003). Comparison of Three Different LCIA Methods: EDIP97, CML2001 and Eco-indicator 99. *The International Journal of Life Cycle Assessment*, vol. *8* (4), pp. 191-200.

Dubai Central Laboratory". (2018). [Accessed 18 April 2018]. Available at: http://www.dcl.ae

Dubai Municipality Portal". (2010). [Accessed 18 April 2018]. Available at: https://www.dm.gov.ae

Dubai Municipality Portal". (2013). [Accessed 18 April 2018]. Available at: https://www.dm.gov.ae

Dubai Municipality Portal". (2018). [Accessed 18 April 2018]. Available at: https://www.dm.gov.ae

Dubai Plan 2021. [Accessed 18 April 2019]. Available at: https://www.dubaiplan2021.ae/dubai-plan-2021.

Ehrhardt, M.K., Crenshaw, E.M., Jenkins, J.C. (2002). Deforestation and the environmental kuznets curve: a cross-national investigation of intervening mechanisms. *Social Science Quarterly*, vol. 83 (1), pp. 226-243.

Eisa, A. (2014). Properties of Concrete Incorporating Recycled Post-Consumer Environmental Wastes. *International Journal of Concrete Structures and Materials*, vol. 8 (3), pp. 251–258.

Elchalakani, M. & Elgaali, E. (2012). Sustainable Concrete made of Construction and Demolition Wastes using Recycled Wastewater in the UAE. *Journal of Advanced Concrete Technology*, vol. 10 (3), pp. 110-125.

Elchalakani, M., Basarir, H. & Karrech, A. (2017). Green Concrete with High-Volume Fly Ash and Slag with Recycled Aggregate and Recycled Water to Build Future Sustainable Cities. *Journal of Materials in Civil Engineering*, vol. *29* (2), pp. 1-12

Elgaali1, E. & Elchalakani, M. (2012). Concrete hollow blocks made with recycled coarse aggregate and recycled water “Green blocks”. *SB13 Dubai Paper – 148*

Elshout, PMF., Van Zelm, R., Karuppiah, R., Laurenzi, IJ., Huijbregts, MAJ. (2014). A spatially explicit data-driven approach to assess the effect of agricultural land occupation on species groups. *The International Journal of Life Cycle Assessment*, vol. 19 (4), pp. 758-769.

Envirocities eMagazine, Issue 4, January 2013. [Accessed 18 April 2018]

Environment Agency in Abu Dhabi, 2013. [Accessed 18 April 2018]

EPA (2008). Environmental Protection Agency, Municipal solid waste, generation, recycling and disposal in the USA-Facts and figures for 2008.

EPS. (2015). Environmental Priority Strategies [Accessed 18 April 2019]. Available at: https://www.ivl.se/english/startpage/pages/our-focus-areas/environmental-engineering-and-sustainable-production/lca/eps.html.

Estanqueiro, B. (2011). Life cycle assessment of the use of recycled aggregates in the production of concrete. pp. 1-10.

Estanqueiro, B., Dinis Silvestre, J., de Brito, J. & Duarte Pinheiro, M. (2016). Environmental life cycle assessment of coarse natural and recycled aggregates for concrete. *European Journal of Environmental and Civil Engineering*, pp. 1-21.

Estévez, B., Aguadoa, A. & Josaa. A. (2006). ENVIRONMENTAL IMPACT OF CONCRETE RECYCLING, COMING FROM CONSTRUCTION AND DEMOLITION WASTE (C&DW). pp. 3-11.

Esty, D.C., Porter, M.E. (2005). National environmental performance: an empirical analysis of policy results and determinants. *Environment and development economics*, vol. 10 (4), pp. 391-434.

Euratom Supply Agency. (2015). *ESA average uranium prices*. [Accessed 18 April 2019]. Available at: http://ec.europa.eu/euratom/observatory_price.html

European Environment Agency. (2005). *The European environment-State and outlook 2005. Copenhagen, Denmark.*

Eurostat, Waste statistics in Europe, Available in epp.eurostat.ec.europa.eu (accessed 09/03/2015).

Evangelista, L., & de Brito, J. (2006). Environmental life cycle assessment of concrete made with fine recycled concrete aggregates.

Falkenmark, M., Rockstrom, J. (2004). Balancing Water for Humans and Nature. *The New Approach in Ecohydrology*, Earthscan, London.

FAO, Food and Agriculture Organization of the United Nations. (2012). "AQUASTAT - Review of agricultural water use per country." [Accessed 18 April 2019]. Available at: http://www.fao.org/nr/water/aquastat/water_use_agr/index.stm.

Federal Ministry for Economic Cooperation and Development, Managing Urbanisation –Towards Sustainable Cities, Federal Ministry for Economic Cooperation and Development, Berlin, 2014.

Ferrari, G., Miyamoto, M. & Ferrari, A. (2014). New sustainable technology for recycling returned concrete. *Construction and Building Materials*, vol. 67, pp. 353-359.

Fib, (2009). Fib-International Federation for Structural Concrete. Structural Concrete – Textbook on behavior, design & performance, vol. 51 (1).

Finnveden, G., Eldh, P., Johansson, J. (2006). Weighting in LCA based on ecotaxes:development of a mid-point method and experiences from case studies. *Int. J.Life Cycle Assess*, Vol. 11 (1), pp. 81–88.

Fisher, C. & Werge, W. (2009). EU as a recycling society, present recycle level of municipal waste and constriction & demolition waste, *European Commission Agency*.

Flower, D. J. M. & Sanjayan, J. G. (2007). Green house gas emissions due to concrete manufacture. *International Journal of Life Cycle Assessment*, vol. *12* (5), pp. 282–288.

Franco, A., Trapp, S. (2008). Estimation of the soil-water partition coefficient normalized to organic carbon for ionizable organic chemicals. *Environmental Toxicology and Chemistry*, vol. 27 (10), pp. 1995- 2004.

Franco, A., Trapp, S. (2010). A multimedia activity model for ionisable compounds: validation study with 2,4-dichlorophenoxyacetic acid, aniline, and trimethoprim Environmental Toxicology and Chemistry, vol. 29 (4), pp. 789-799.

Friedrich, R., Kuhn, A., Bessagnet, B., Blesl, M., Bruchof, D., Cowie, H., Fantke, P., Gerharz, L., Grellier, J., Gusev, A., Haverinen-Shaughnessy, U., Hout, D., Hurley, F., Huynen, M., Kampffmeyer, T., Karabelas, A., Karakitsios, S., Knol, A., Kober, T., Kollanus, V., Kontoroupis, P., Kuder, R., Kugler, U., Loh, M., Meleux, F., Miller, B., Müller, W., Nikolaki, S., Panasiuk, D., Preiss, P., Rintala, T., Roos, J., Roustan, Y., Salomons, E., Sánchez Jiménez, A., Sarigiannis, D., Schenk, K., Shafrir, A., Shatalov, V., Solomou, E., Theloke, J., Thiruchittampalam, B., Torras Ortiz, S., Travnikov, O., Tsyro, S.,

Frischknecht, R., Braunschweig, A., Hofstetter, P. & Suter,P. (2000). Human health damages due to ionising radiation in life cycle impact assessment. *Environmental Impact Assessment Review*, vol. 20 (2), pp. 159-189.

Frischknecht, R., Braunschweig, A., Hofstetter, P. Suter, P. (2000). Human health damages due to ionising radiation in life cycle impact assessment. *Environmental Impact Assessment Review*, vol. 20 (2), pp. 159-189.

Fu, W., Franco, A. (2009). Methods for estimating the bioconcentration factor of ionizable organic chemicals. *Environmental Toxicology and Chemistry*, vol. 28 (7), pp. 1372-1379.

Fuertes, A., Casals, M., Gangolells, M., Forcada, N., Macarulla, M. & Roca, X. (2013). An Environmental Impact Causal Model for improving the environmental performance of construction processes. *Journal of Cleaner Production*, vol. 52, pp. 425-437.

Gabel, K., Forsberg, P. & Tillman, A.M. (2004). The design and building of a life cycle-based process model for simulating environmental performance and cost in cement manufacturing, *Journal of Cleaner Production*, vol. 12, pp.77-93.

Gerosa, G., Fusaro, L., Monga, R., Finco, A., Fares, S., Manes, F., Marzuoli, R. (2015). A flux-based assessment of above and below ground biomass of Holm oak: seedlings after one season of exposure to high ozone concentrations. *Atmos Environ*, vol. 113, pp. 41-49.

Ghanbari, M., Monir Abbasi, A. & Ravanshadnia, M. (2017). Economic and Environmental Evaluation and Optimal Ratio of Natural and Recycled Aggregate Production. *Advances in Materials Science and Engineering*, vol. 2017, pp. 1-10.

Ghosh, S., Haldar, H., Chatterjee, S. & Ghosh, P. (2016). An Optimization Model on Construction and Demolition Waste Quantification from Building. *Procedia Environmental Sciences*, vol. 35, pp. 279-288.

Giama, E. & Papadopoulos, A. (2015). Assessment tools for the environmental evaluation of concrete, plaster and brick elements production. *Journal of Cleaner Production*, vol. 99, pp. 75-85.

Giama, E. & Papadopoulos, A. M. (2015). Assessment tools for the environmental evaluation of concrete, plaster and brick elements production. *Journal of Cleaner Production* vol. *99*, pp. 75-85.

Goedkoop MJ., Spriensma, R. (1999). The eco-indicator '99: A damage-oriented method for life-cycle impact assessment. *The Hague (the Netherlands): Ministry of Housing, Spatial Planning and the Environment*.

Goedkoop, M. & Spriensma, R. (2000) The Eco-Indicator 99: A Damage Oriented Method for Life Cycle Impact Assessment-Methodology Report, Pré Consultants B.V., Amersfoort, The Netherlands.

Goedkoop, M., Heijungs, R., Huijbregts, MAJ., De Schryver, A., Struijs, J., Van Zelm, R. (2009). ReCiPe 2008: A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and endpoint levels. *First edition. Report i: Characterization. the Netherlands: Ruimte en Milieu, Ministerie van Volkshuisvesting, Ruimtelijke Ordening en Milieubeheer*.

Goedkoop, M., Hofstetter, P., Müller-Wenk, R. & Spriemsma, R. (1998). The ECO-indicator 98 explained. *The International Journal of Life Cycle Assessment*, vol. 3 (6), pp. 352-360.

Goedkoop, M., Oele, M. and Effting, S. (2004). SimaPro 6 Database Manual - Methods library. Netherlands, PRé Consultants.

Golsteijn, L., Van Zelm, R., Hendriks, AJ., Huijbregts, MAJ. (2013). Statistical uncertainty in hazardous terrestrial concentrations estimated with aquatic ecotoxicity data. *Chemosphere*, vol. 93, pp. 366-372.

Goodland, R. (1995). The concept of environmental sustainability. *Annual review of ecology and systematics*, vol. 26 (1), pp. 1-24.

Grossman, G.M. and Krueger, A.B. (1991). Environmental impacts of a North American free trade agreement (No. w3914). *National Bureau of Economic Research.*

Grossman, G.M., Krueger, A.B. (1995). Economic growth and the environment. *The quarterly journal of economics*, vol. 110 (2), pp. 353-377.

Guardigli, L., Monari, F. & Bragadin, M. A. (2011). Assessing Environmental Impact of Green Buildings through LCA Methods: Acomparison between Reinforced Concrete and Wood Structures in the European Context. *Procedia Engineering*, vol. 21, pp. 1199-1206.

Guerriero, C. and Cairns, J. (2011). Cost-benefit analysis of the clean-up of hazardous waste sites. *In Integrated Waste Management*, vol. 1.

GUINEE J. B., (final editor), GOREE M., HEIJUNGS R., HUPPES G., KLEIJN R., de KONING A., van OERS L., WEGENER SLEESWIJK A., SUH S., UDO de HAES H. A., de BRUIJN H., van DUIN R., HUIBREGTS M. A. J., LINDEIJER E., ROORDA A. A. H., WEIDEMA B. P. (2001) “Life cycle assessment; An operational guide to the ISO standards”, Ministry of Housing, Spatial Planning and Environment (VROM) and Centre of Environmental Science (CML), Den Haag and Leiden, The Netherlands, 704 p.

Guinee J.B. (2002). Handbook of Life Cycle Assessment: Operation Guide to ISO Standards. Secaucus, NJ, USA, Kluwer Academic Publishers.

Häkkinen, T. & Mäkelä, K. (1996). Environmental Adaptation of Concrete and Asphalt Pavements.

Han, J. & Thakur, J. K. (2012). Use of Geosynthetics to Stabilize Recycled Aggregates in Roadway Construction. In *ICSDEC 2012* (pp. 473–480). Reston, VA: American Society of Civil Engineers.

Hanafiah, MM., Xenopoulos, MA., Pfister, S., Leuven, RS., Huijbregts, MAJ. (2011). Characterization Factors for Water Consumption and Greenhouse Gas Emissions Based on Freshwater Fish Species Extinction. *Environmental Science & Technology*, vol. 45 (12), pp. 5572-5278.

Hanafiah, MM., Xenopoulos, MA., Pfister, S., Leuven, RS., Huijbregts, MAJ. (2011). Characterization Factors for Water Consumption and Greenhouse Gas Emissions Based on Freshwater Fish Species Extinction. *Environmental Science & Technology*, vol. 45 (12), pp. 5572-5278.

Hao, J., Hills, M. & Tam, V. (2008). The effectiveness of Hong Kong's Construction Waste Disposal Charging Scheme. *Waste Management & Research*, vol. 26 (6), pp. 553-558.

Hauschild, M. Z., Huijbregts, M. A. J. (2015). Introducing life cycle impact assessment. *In Life cycle impact assessment*.

Hayashi, K., Itsubo, N., Inaba, A. (2000). Development of damage function for stratospheric ozone layer depletion. *The International Journal of Life Cycle Assessment*, vol 5 (5), pp. 265-272.

Hayashi, K., Itsubo, N., Inaba, A. (2000). Development of damage function for stratospheric ozone layer depletion. *The International Journal of Life Cycle Assessment*, vol. 5 (5), pp. 265-272.

Hayashi, K., Nakagawa, A., Itsubo, N., Inaba, A. (2006). Expanded Damage Function of Stratospheric Ozone Depletion to Cover Major Endpoints Regarding Life Cycle Impact Assessment. *The International Journal of Life Cycle Assessment*, vol.11 (3), pp. 150-161.

Hayashi, K., Nakagawa, A., Itsubo, N., Inaba, A. (2006). Expanded Damage Function of Stratospheric Ozone Depletion to Cover Major Endpoints Regarding Life Cycle Impact Assessment. *The International Journal of Life Cycle Assessment*, vol. 11 (3), 150-161.

Hayashi, K., Okazaki, M., Itsubo, N., Inaba, A. (2004). Development of damage function of acidification for terrestrial ecosystems based on the effect of aluminum toxicity on net primary production. *The International Journal of Life Cycle Assessment*, vol. 9, pp. 13-22.

Hayashi, K., Okazaki, M., Itsubo, N., Inaba, A. (2004). Development of damage function of acidification for terrestrial ecosystems based on the effect of aluminum toxicity on net primary production. *The International Journal of Life Cycle Assessment*, vol. 9, pp. 13-22.

Heijungs, R., Goedkoop, M., Struijs, J., Effting, S., Sevenster, M. & Huppes, G. (2003). Towards a life cycle impact assessment method which comprises category indicators at the midpoint and the endpoint level. Report of the first project phase Design of the

Helmes, RJK., Huijbregts, MAJ., Henderson, AD., Jolliet, O. (2012). Spatially explicit fate factors of phosphorous emissions to fresh water at the global scale. *International Journal of Life Cycle Assessment*, vol. 17 (5), pp. 646-654.

Hendriks, AJ., Heikens, A. (2001). The power of size: II. Rate constants and equilibrium ratios for accumulation of inorganic substances. *Environmental Toxicology and Chemistry*, vol. 20 (7), pp. 1421- 1437.

Hendriks, AJ., Smitkova, H., Huijbregts, MAJ. (2007). A new twist on an old regression: Transfer of chemicals to beef and milk in human and ecological risk assessment. *Chemosphere*, vol. 70 (1), pp. 46-56.

Hendriks, C.A., Worrell, E., Price, L., Martin, N., Ozawa Meida, L., de Jager, D. & Riemer, P. (1998). Emission reduction of greenhouse gases from the cement industry, Proc. Fourth International Conference on Greenhouse Gas Control Technologies.

Henry, M. & Kato, Y. (2014). Understanding the regional context of sustainable concrete in Asia: Case studies in Mongolia and Singapore. *Resources, Conservation and Recycling*, vol. *82*, pp. 86-93.

HKEPD", 2013. Construction waste [Accessed 18 April 2018]. Available at: http://www.epd.gov.hk/epd/misc/cdm/introduction.htm

Hoekstra, AY. Mekonnen, MM. (2012). The water footprint of humanity. *PNAS*, vol. 109 (9), pp. 3232-3237.

Hong Kong Environmental Protection Department, Hong Kong SAR.

Hossain, M., Poon, C., Lo, I. & Cheng, J. (2016). Comparative environmental evaluation of aggregate production from recycled waste materials and virgin sources by LCA. *Resources, Conservation and Recycling*, vol. 109, pp. 67-77.

Hosseini, P., Booshehrian, A. and Madari, A. (2011). Developing concrete recycling strategies by utilization of nano-SiO 2 particles. *Waste and Biomass Valorization*, vol. 2 (3), pp. 347-355.

Houghton, J. T., Callender, B. A. and Varney, S. K. (1992). Climate Change 1992 - The supplementary report to the IPCC scientific assessment. Cambridge, UK, Cambridge University Press.

Huang, C., Liang, W., Chuang, H. & Chang, Z. (2012). A novel approach to product modularity and product disassembly with the consideration of 3R-abilities. *Computers & Industrial Engineering*, vol. 62 (1), pp. 96-107.

Huijbregts, M.A.J., Steinmann, Z.J.N., Elshout, P.M.F., Stam, G., Verones, F., Vieira, M.D.M., Hollander, A., Zijp, M., Van Zelm, R. (2016). ReCiPe 2016: A harmonized life cycle impact assessment method at midpoint and endpoint level *Report I: Characterization*, pp. 1-201.

Huijbregts, MAJ., Norris, G., Bretz, R., Ciroth, A., Maurice, B., Von Bahr, B., Weidema, B., De Beaufort, ASH. (2001). Framework for modelling data uncertainty in life cycle inventories. *Int J LCA*, vol. 6 (3), pp. 127-132.

Huijbregts, MAJ., Struijs, J., Goedkoop, M., Heijungs, R., Hendriks, AJ., Van de Meent, D. (2005). Human population intake fractions and environmental fate factors of toxic pollutants in life cycle impact assessment. *Chemosphere*, vol. 61 (10), pp. 1495-1504.

Huntzinger, D. N. & Eatmon, T. D. (2009). A life-cycle assessment of Portland cement manufacturing: comparing the traditional process with alternative technologies. *Journal of Cleaner Production*, vol. *17* (7), pp. 668–675.

Huysegoms, L., Rousseau, S., Cappuyns, V. (2018). Friends or foes? Monetized Life Cycle Assessment and Cost-Benefit Analysis of the site remediation of a former gas plant. *Science of the Total Environment*, vol. 619, pp. 258-271.

IARC. (2004). Summaries and evaluations. [Accessed 18 April 2019]. Available at: www.inchem.org/pages/iarc.html.

Inaba, A. (2013). From Carbon to Environment, From Product to Organization. In: Presentation at International Workshop for Scope 3 Standard and LCA for Organization.

Integrated Pollution Prevention Control (IPPC) (2000) Reference Document on Best Available Techniques in the Cement and Lime Manufacturing Industries, European Commission, Technologies for Sustainable Development European IPPC Bureau.

International Energy Agency. (2013). Resources to Reserves 2010. Oil, gas and coal technologies for the energy markets of the future. *Paris: International Energy Agency*.

International Standards Organization (ISO) 14040:2006

IPCC. (2013). Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change.

Ismail, S. & Ramli, M. (2013). Engineering properties of treated recycled concrete aggregate (RCA) for structural applications. *Construction and Building Materials*, vol. 44, pp. 464-476.

Itsubo, N. & Inaba, A. (2012). Life-cycle Impact assessment Method based on Endpoint modeling.

Itsubo, N., Sakagami, M., Washida, T., Kokubu, K., Inaba, A. (2004). Weighting across safeguard subjects for LCIA through the application of conjoint analysis. *Int. J.Life Cycle Assess*, vol. 9 (3), pp. 196–205.

Jerrett, M., Burnett, RT., Pope, CA., Ito, K., Thurston, G., Krewski, D., Shi, YL., Calle, E., Thun, M. (2009). Long-Term Ozone Exposure and Mortality. *New Engl J Med*, vol. 360 (11), pp. 1085-1095.

Jia, S., Yan, G., Shen, A. & Zheng, J. (2017). Dynamic simulation analysis of a construction and demolition waste management model under penalty and subsidy mechanisms. *Journal of Cleaner Production*, vol. 147, pp. 531-545.

John, A. & Itodo, D. (2013). Professionals' views of material wastage on construction sites and cost overruns. *Organization, Technology & Management in Construction: An International Journal*, vol. 5 (1), pp. 747-757.

Jolliet, O., R. A.Mueller-Wenk, C. Peña, et al. 2003. Life Cycle Impact Assessment Programme of the Life Cycle Initiative. Final report of the LCIA Definition study [Electronic Version]. Retrieved 17 September 2007, from http:// lcinitiative.unep.fr/ includes/ file.asp? site= lcinitandfile=F7BF1ABF-8B98-4A95-9FDE-3E32EB7C4EC4

Joos, F., Roth, R., Fuglestvedt, JS., Peters, GP., Enting, IG., Von Bloh, W., Brovkin, V., Burke, EJ., Eby, M., Edwards, NR., Friedrich, T., Frölicher, TL., Halloran, PR., Holden, PB., Jones, C., Kleinen, T., Mackenzie, FT., Matsumoto, K., Meinshausen, M., Plattner, G-K., Reisinger, A., Segschneider, J., Shaffer, G., Steinacher, M., Strassmann, K., Tanaka, K., Timmermann, A., Weaver, AJ. (2013). Carbon dioxide and climate impulse response functions for the computation of greenhouse gas metrics: a multi-model analysis. *Atmospheric Chemistry and Physics*, vol. 13 (5), pp. 2793-2825.

Jungbluth, N., Frischknecht, R. (2010). Cumulative energy demand. In: Hischier, R., Weidema, B., Eds. Implementation of Life Cycle Impact Assessment Methods. St Gallen: *Ecoinvent centre*, pp. 33-40.

Jungbluth, N., Frischknecht, R. (2010). Cumulative energy demand. In: Hischier, R., Weidema, B., Eds. Implementation of Life Cycle Impact Assessment Methods. St Gallen: *Ecoinvent centre*, pp 33-40.

Jungbluth, N., Frischknecht, R. (2010). Cumulative energy demand. In: Hischier, R., Weidema, B., Eds. Implementation of Life Cycle Impact Assessment Methods. St Gallen: *Ecoinvent Centre*, pp. 33-40.

Kabir, S., Al-Ismaeel, A.A., Aeshah, A.Y.B. & Al-Sadun, F.S. (2013). Sustainable management program for construction waste. In 9th International Concrete Conference & Exhibition, Manama.

Kabir, S., Al-Shayeb, A. & Khan, I. (2016). Recycled Construction Debris as Concrete Aggregate for Sustainable Construction Materials. *Procedia Engineering*, vol. 145, pp. 1518-1525.

Kartam, N., Al-Mutairi, N., Al-Ghusain, I. & Al-Humoud, J. (2004). Environmental management of construction and demolition waste in Kuwait. *Waste Management*, vol. 24 (10), pp. 1049-1059.

Katherine, S., Kyle, F. & Fraser, B. (2012). WASTE AND RECYCLING IN AUSTRALIA 2011. *DEPARTMENT OF SUSTAINABILITY, ENVIRONMENT, WATER, POPULATION AND COMMUNITIES*, (2), pp.37-97.

Kelly, T. D., Matos, G. R. (2013). Historical statistics for mineral and material commodities in the United States (2013 version): Reston, Virginia: *U.S. Geological Survey Data Series 140*. [Accessed 18 April 2019]. Available at: http://minerals.usgs.gov/minerals/pubs/historicalstatistics

Kelly, T. D., Matos, G. R. (2013). Historical statistics for mineral and material commodities in the United States (2013 version): Reston, *Virginia: U.S. Geological Survey Data Series 140*. [Accessed 18 April 2019]. Available at:http://minerals.usgs.gov/minerals/pubs/historicalstatistics.

Kim, T., Chae, C., Kim, G. & Jang, H. (2016). Analysis of CO2 Emission Characteristics of Concrete Used at Construction Sites. *Sustainability*, vol. 8 (4), pp. 348.

Kitco Metals Inc. (2015). Historical Charts and Data-London Fix. [Accessed 18 April 2019]. Available at: http://www.kitco.com/charts

Klee, H., 2009. Recycling Concrete, Report of the Cement Sustainability Initiative.

Kleijer, A. L., Lasvaux, S., Citherlet, S. & Viviani, M. (2017). Product-specific Life Cycle Assessment of ready mix concrete: Comparison between a recycled and an ordinary concrete. *Resources, Conservation and Recycling*, vol. 122, pp. 210-218.

Kleijn R. [Ph.D. thesis] Materials and energy: a story of linkages [Ph.D. thesis].Netherlands: Leiden University; 2012. p. 135.

Klöpffer, W. (1997). Life cycle assessment: From the beginning to the current state. *Environmental Science and Pollution Research International*, vol. 4 (4), pp. 223-228.

Koellner, T., de Baan, L., Beck, T., Brandão, M., Civit, B., Margni, M., Canals, L., Saad, R., de Souza, D., Müller-Wenk, R. (2013). UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA. *The International Journal of Life Cycle Assessment*, vol. 18 (6), pp. 1188-1202.

Kofoworola, O. & Gheewala, S. (2009). Estimation of construction waste generation and management in Thailand. *Waste Management*, vol. 29 (2), pp. 731-738.

Kolay, P.K. & Akentuna, M. (2014). Characterization and utilization of recycled concrete aggregate from Illinois as a construction material. In *Geo-Congress 2014: Geo-characterization and Modeling for Sustainability*. pp. 3561-3570.

Köllner, T, Scholz, RW. (2007). Assessment of land use impacts on the natural environment. Part 1: An analytical framework for pure land occupation and land use change. *International Journal of Life Cycle Assessment*, vol. 12, pp. 16-23.

Koroneos, C. J. & Dompros, A. T. (2009). Environmental assessment of the cement and concrete life cycle in Greece. *International Journal of Environmental Technology and Management*, vol. 10 (1), pp. 71-88.

Koroneos, C.J. & Dompros, A.T. (2008). Environmental assessment of the cement and concrete life cycle in Greece. *International Journal of Environmental Technology and Management*, vol. 10 (1), pp. 71-88.

Kourmpanis, B., Papadopoulos, A., Moustakas, K., Stylianou, M., Haralambous, K. & Loizidou, M. (2008). Preliminary study for the management of construction and demolition waste. *Waste Management & Research*, vol. 26 (3), pp. 267-275.

Kralj, D. (2008). Dialectal system approach supporting environmental innovation for sustainable development. *Kybernetes*, vol. 37 (9/10), pp. 1542-1560.

Krieg, H., Albrecht, S., Jäger, M. (2013). Systematic monetisation of environmental impacts. *WIT Transactions on Ecology and the Environment*, vol. 173, pp. 513-524.

Krol, M., Houweling, S., Bregman, B., van den Broek, M., Segers, A., van Velthoven, P., Peters, W., Dentener, F., Bergamaschi, P. (2005). The two-way nested global chemistry-transport zoom model TM5: algorithm and applications. *Atmospheric Chemistry and Physics*, vol. 5, pp. 417-432.

Kucukvar, M., Egilmez, G. & Tatari, O. (2016). Life cycle assessment and optimization-based decision analysis of construction waste recycling for a LEED-certified university building. *Sustainability (Switzerland)*, vol. *8* (1), pp. 1-13.

Kulatunga, U., Amaratunga, D., Haigh, R. & Rameezdeen, R. (2006). Attitudes and perceptions of construction workforce on construction waste in Sri Lanka. *Management of Environmental Quality: An International Journal*, vol. 17(1), pp.57-72.

Kumar, R. (2017). Influence of recycled coarse aggregate derived from construction and demolition waste (CDW) on abrasion resistance of pavement concrete. *Construction and Building Materials*, vol. *142*, pp. 248–255.

Lamarque, JF., Bond, TC., Eyring, V., Granier, C., Heil, A., Klimont, Z., Lee, D., Liousse, C., Mieville, A., Owen, B., Schultz, MG., Shindell, D., Smith, SJ., Stehfest, E., Van Aardenne, J., Cooper, OR., Kainuma, M., Mahowald, N., McConnell, JR., Naik, V., Riahi, K., van Vuuren, DP. (2010). Historical (1850-2000) gridded anthropogenic and biomass burning emissions of reactive gases and aerosols: methodology and application. *Atmospheric Chemistry and Physics*, vol. 10 (15), pp. 7017-7039.

Lelieveld, J., Evans, JS., Fnais, M., Giannadaki, D., Pozzer, A. (2015). The contribution of outdoor air pollution sources to premature mortality on a global scale. *Nature*, vol. 525, pp. 361-371.

Liang, Y., Ye, Z., Vernerey, F. & Xi, Y. (2015). Development of Processing Methods to Improve Strength of Concrete with 100% Recycled Coarse Aggregate. Journal of Materials in Civil Engineering, vol. 27 (5), p. 04014163.

Lingard, H., Graham, P. & Smithers, G. (1997). Waste management in the Australian construction industry: a human factors approach. *Association of Researchers in Construction Management*, pp.12-203.

Lingard, H., Graham, P. & Smithers, G. (2000). Employee perceptions of the solid waste management system operating in a large Australian contracting organization: implications for company policy implementation. *Construction Management and Economics*, vol. 18 (4), pp. 383-393.

Lippmann, M., Chen, L-C., Gordon, T., Ito, K., Thurston, GD. (2013). National Particle Component Toxicity (NPACT) initiative: integrated epidemiologic and toxicologic studies of the health effects of particulate matter components. *Health Effects Institute, Boston, Massachusetts, USA*, vol. 177.

Lu, W. & Tam, V. (2013). Construction waste management policies and their effectiveness in Hong Kong: A longitudinal review. *Renewable and Sustainable Energy Reviews*, vol. 23, pp. 214-223.

Lu, W. & Yuan, H. (2010). Exploring critical success factors for waste management in construction projects of China. Resources, Conservation and Recycling, vol. 55 (2), pp. 201-208.

Lu, W. & Yuan, H. (2011). A framework for understanding waste management studies in construction. *Waste Management*, vol. 31 (6), pp. 1252-1260.

Lu, W. & Yuan, H. (2012). Off-site sorting of construction waste: What can we learn from Hong Kong?. *Resources, Conservation and Recycling*, vol. 69, pp. 100-108.

Lundström, H. (1997) LCA av Betongtakpannor – en Analys av Miljöbelastning Under Takpannans Livscykel, Chalmers Industriteknik.

Lundström, H., Ölind, G. and Rydberg, T. (1996). Life Cycle Assessment of Concrete Sewage Pipe.

Magalhães, R., Danilevicz, Â. & Saurin, T. (2017). Reducing construction waste: A study of urban infrastructure projects. *Waste Management*, vol. 67, pp. 265-277.

Malhotra, V. M. & Mehta, P. K. (2008). High performance high-volume fly ash concrete for building sustainable and durable structures, 3rd Ed., Supplementary Cementing Materials for Sustainable Developments, Ottawa.

Mália, M., de Brito, J., Pinheiro, M. & Bravo, M. (2013). Construction and demolition waste indicators. *Waste Management & Research*, vol. 31 (3), pp. 241-255.

Manowong, E. (2012). Investigating factors influencing construction waste management efforts in developing countries: an experience from Thailand. *Waste Management & Research*, vol. 30 (1), pp. 56-71.

Marin, G., Mazzanti, M. (2009). The dynamics of delinking in industrial emissions: the role of productivity, trade and R&D. *Journal of Innovation Economics Management*, vol. (1), pp. 91-117

Mark Goedkoop (1995). Eco-indicator 95 - Final report. Netherlands, PRe Consultant.

Martin, N., Worrell, E. & Price, L. (1999). Energy Efficiency and Carbon Dioxide Emissions Reduction Opportunities in the US Cement Industry, Lawrence Berkeley National Laboratory.

Martin, N., Worrell, E. & Price, L. (1999). Energy Efficiency and Carbon Dioxide Emissions Reduction Opportunities in the US Cement Industry, Lawrence Berkeley National Laboratory.

Martínez, E., Nuñez, Y. & Sobaberas, E. (2013). End of life of buildings: three alternatives, two scenarios. A case study. *The International Journal of Life Cycle Assessment*, vol. 18 (5), pp. 1082-1088.

Meddah, M.S. (2017). Recycled aggregates in concrete production: engineering properties and environmental impact. In *MATEC Web of Conferences*, vol. 101, pp. 05021.

Meijer, B. (2018). Weighting: Applying a Value Judgement to LCA Results [Accessed 18 April 2019]. Available at: https://www.pre-sustainability.com/news/weighting-applying-a-value-judgement-to-lca-results.

Meijer, B. (2018). Weighting: Applying a Value Judgement to LCA Results [Accessed 18 April 2019]. Available at: https://www.pre-sustainability.com/news/weighting-applying-a-value-judgement-to-lca-results.

Menard, Y., Bru, K., Touze, S., Lemoign, A., Poirier, J. E., Ruffie, G. & Von Der Weid, F. (2013). Innovative process routes for a high-quality concrete recycling. *Waste Management*, vol. *33* (6), pp. 1561-1565.

Mercante, I., Bovea, M., Ibáñez-Forés, V. & Arena, A. (2012). Life cycle assessment of construction and demolition waste management systems: a Spanish case study. *The International Journal of Life Cycle Assessment*, vol. 17 (2), pp. 232-241.

Milà i Canals, L., Bauer, C., Depestele, J., Dubreuil, A., Freiermuth Knuchel, R., Gaillard, G., Michelsen, O., Müller-Wenk, R., Rydgren, B. (2007). Key elements in a framework for land use impact assessment within LCA. *International Journal of Life Cycle Assessment*, vol 12 (1), pp. 5-15.

Moldan, B., Janoušková, S., Hák, T. (2012). How to understand and measure environmental sustainability: Indicators and targets. *Ecological Indicators*, vol. 17, pp. 4-13.

Murray, CJL., Ezzati, M., Flaxman, A., Lim, S., Lozano, R., Michaud, C., Naghavi, M., Salomon, J., Shibuya, K., Vos, T., Wikler, D., Lopez, A. (2012). GBD 2010: design, definitions, and metrics. *Lancet*, vol. 380, pp. 2063-2066.

Nations Environmental Programme Sustainable Buildings & Climate Initiative, pp. 62.

Nemani, RR., Keeling, CD., Hashimoto, H., Jolly, WM., Piper, SC., Tucker, CJ., Myneni, RB., Running, S. (2003). Climate-Driven Increases in Global Terrestrial Net Primary Production from 1982 to 1999. *Science*, vol. 300 pp. 1560-1563.

Neri, P., (a cura di). Verso la valutazione ambientale degli edifici – Life Cycle Assessment a supporto della progettazione ecosostenibile. Firenze:Alinea; 2007.

new method [Electronic Version]. Retrieved 7 Mac 2007, from http:// www.leidenuniv.nl/ cml/ssp/ publications/ recipe_phase1.pdf

Nguyen, T.L.T., Laratte, B., Guillaume, B., Hua, A. (2016). Quantifying environmental externalities with a view to internalizing them in the price of products, using different monetization models. *Resources, Conservation and Recycling*, vol. 109, pp. 13-23.

Nicoletti, G. M., Notarnicola, B., & Tassielli, G. (2002). Comparative Life Cycle Assessment of flooring materials: Ceramic versus marble tiles. *Journal of Cleaner Production*, vol. 10 (3), pp. 283-296.

Nisbet, M. & van Geem, M. (1997). Environmental life cycle inventory of Portland cement and concrete, *World Cement*, vol. 28 (4), pp.100-103.

Oh, D., Noguchi, T., Kitagaki, R. & Park, W. (2014). CO2 emission reduction by reuse of building material waste in the Japanese cement industry. *Renewable and Sustainable Energy Reviews*, vol. 38, pp. 796-810.

Olson, DM., Dinerstein, E., Wikramanayake, ED., Burgess, ND., Powell, GVN., Underwood, EC., D'amico, JA., Itoua, I., Strand, HE., Morrison, JC., Loucks, CJ., Allnutt, TF., Ricketts TH., Kura, Y., Lamoreux, JF., Wettengel, WW., Hedao, P., Kassem, KR. (2001). Terrestrial ecoregions of the world: A new map of life on earth. *BioScience*, vol. 51 (11), pp. 933-938.

Ortiz, O., Pasqualino, J. & Castells, F. (2010). Environmental performance of construction waste: Comparing three scenarios from a case study in Catalonia, Spain. *Waste Management*, vol. 30 (4), pp. 646-654.

Osmani, M., Glass, J. & Price, A. (2008). Architects’ perspectives on construction waste reduction by design. *Waste Management*, vol. 28 (7), pp. 1147-1158.

Özler, Ş.İ., Obach, B.K. (2009). Capitalism, state economic policy and ecological footprint: an international comparative analysis. *Global Environmental Politics*, vol. *9* (1), pp. 79-108.

Pacheco-Torgal, F., Labrincha, J., Ding, Y., de Brito, J. & Tam, V. (2013). *Handbook of Recycled Concrete and Demolition Waste, Elsevier*.

Pade, C. & Guimaraes, M. (2007). The CO2 uptake of concrete in a 100 year perspective. *Cement and Concrete Research*, vol. *37* (9), pp. 1348–1356.

Peng, C., Scorpio, D. & Kibert, C. (1997). Strategies for successful construction and demolition waste recycling operations. *Construction Management and Economics*, vol. 15 (1), pp. 49-58.

Pennington, D., Potting, J., Finnveden, G., Lindeijer, E., Jolliet, O., Rydberg, T. & Rebitzer, G. (2004). Life cycle assessment Part 2: Current impact assessment practice. *Environment International*, vol. 30 (5), pp. 721-739.

Peprah, K., Amoah, S. & Achana, G. (2015). Assessing '3Rs' Model in Relation to Municipal Solid Waste Management in Wa, Ghana. *World Environment*, vol. 5 (3), pp. 112-120.

Petkovic, G., Engelsen, C.J., Håøya, A.O. & Breedveld, G. (2004). Environmental impact from the use of recycled materials in road construction: method for decision-making in Norway. *Resources, Conservation and Recycling*, vol. 42 (3), pp.249-264.

Peuportier, B., Sabina, S., Glaumann, M., Tove, M., Krigsvol, G., Wetzel, C., Staller, H., Szalay, Z., Degiovanni, V., & Stoykova, E. (2009). Energy Saving through Promotion of Life Cycle Assessment in Buildings. Deliverable D2.1 State of the art report and Deliverable D2.2 Collection of published material.

Pfister, S., Koehler, A., Hellweg, S. (2009). Assessing the Environmental Impacts of Freshwater Consumption in LCA. *Environmental Science & Technology*, vol. 43 (11), pp. 4098-4104.

Pfister, S., Koehler, A., Hellweg, S. (2009). Assessing the Environmental Impacts of Freshwater Consumption in LCA. *Environmental Science & Technology*, vol. 43 (11), pp. 4098-4104.

Polat, G., Damci, A., Turkoglu, H. & Gurgun, A. (2017). Identification of Root Causes of Construction and Demolition (C&D) Waste: The Case of Turkey. *Procedia Engineering*, vol. 196, pp. 948-955.

Ponnada, M. & Kameswari, P. (2015). Construction and Demolition Waste Management – A Review. *International Journal of Advanced Science and Technology*, vol. 84 (19-46).

Ponnada, M. & Kameswari, P. (2015). Construction and Demolition Waste Management – A Review. *International Journal of Advanced Science and Technology*, vol. 84, pp. 19-46.

Ponsioen, TC., Vieira, MDM., Goedkoop, MJ. (2014) Surplus cost as a lifecycle impact indicator for fossil resource scarcity. *International Journal of Life Cycle Assessment*, vol. 19, pp. 872–881.

Poon, C. S., Yu, A. T. W. & Ng, L. H. (2001). On-site sorting of construction and demolition waste in Hong Kong. *Resources, Conservation and Recycling*, vol. *32* (2), pp. 157-172.

Poon, C., Yu, A. & Ng, L. (2001). On-site sorting of construction and demolition waste in Hong Kong. *Resources, Conservation and Recycling*, vol. 32 (2), pp. 157-172.

Poon, C., Yu, A. & Ng, L. (2003). Comparison of low-waste building technologies adopted in public and private housing projects in Hong Kong. *Engineering, Construction and Architectural Management*, vol. 10 (2), pp. 88-98.

Poon, C.S., Yu, A.T.W., & Ng, L.H. (2001). On-site sorting of construction and demolition waste in Hong Kong. *Resources, Conservation and Recycling*, vol. 32 (2), pp. 157-172.

Porhinčák, M., & Eštoková, A. (2013). Environmental analysis of building materials of homes in slovak republic - A case study. *International Journal for Housing Science and Its Applications*, *37*(2), 111–120.

Protection dans le domaine Nucléaire (CEPN), edited by the European Commission DGXII, Science, *Research and Development JOULE*, Luxembourg, 1995.

Pushkar, S., Becker, R. & Katz, A. (2005). A methodology for design of environmentally optimal buildings by variable grouping. *Building and Environment*, vol. 40 (8), pp. 1126-1139.

Radivojevic, A. & Nedic, M. (2008). Environmental evaluation of building materials: Example of two residential buildings in Belgrade. *Facta universitatis - series: Architecture and Civil Engineering*, vol. 6 (1), pp. 97-111.

Reap, J., Roman, F., Duncan, S., Bras, B. (2008). A survey of unresolved problems in life cycle assessment: part 2. *The International Journal of Life Cycle Assessment*, vol. 13 (5), pp. 374-388

Richardson, A. (2013). Reuse of materials and by products in construction: waste minimization and recycling. Springer, London, pp 1-4.

Rodrigues, F., Carvalho, M., Evangelista, L. & de Brito, J. (2013). Physical–chemical and mineralogical characterization of fine aggregates from construction and demolition waste recycling plants. *Journal of Cleaner Production*, vol. 52, pp. 438-445.

Roh, S.J., Tae, S.H., Shin, S.W. & Woo, J.H. (2013). A Study on the Comparison of Characterization of Environmental Impact of Major Building Material for Building Life Cycle Assessment. *Journal of the Architectural Institute of Korea Structure & Construction*, vol. 29 (7), pp. 93-100.

Roy, P-O, Azevedo, LB., Margni, M., Van Zelm, R., Deschênes, L., Huijbregts, MAJ. (2014). Characterization factors for terrestrial acidification at the global scale: A systematic analysis of spatial variability and uncertainty. *Science of the Total Environment*, vol. 500, pp. 270-276.

Roy, P-O., Azevedo, LB., Margni, M., Van Zelm, R., Deschênes, L., Huijbregts, MAJ. (2014). Characterization factors for terrestrial acidification at the global scale: A systematic analysis of spatial variability and uncertainty. *Science of the Total Environment*, vol. 500, pp. 270-276.

Roy, PO., Deschenes, L., Margni, M. (2012a). Life cycle impact assessment of terrestrial acidification: modelling spatially explicit soil sensitivity at the global scale. *Environmental Science & Technology*, vol. 46 (15), pp. 8270–8278.

Roy, PO., Huijbregts, M., Deschenes, L., Margni, M. (2012b) Spatially differentiated atmospheric source-receptor relationships for nitrogen oxides, sulfur oxides and ammonia emissions at the global scale for life cycle impact assessment. *Atmospheric Environment*, vol. 62, pp. 74–81.

Sabai, M.M., Cox, M.G.D.M., Mato, R.R., Egmond, E.L.C. & Lichtenberg, J.J.N. (2013). Concrete block production from construction and demolition waste in Tanzania. Resources, Conservation and Recycling, vol. 72, pp.9-19.

Sadati, S., Arezoumandi, M., Khayat, K. & Volz, J. (2016). Shear performance of reinforced concrete beams incorporating recycled concrete aggregate and high-volume fly ash. *Journal of Cleaner Production*, vol. 115, pp. 284-293.

Saez, P., Merino, R., González, A. & Porras, C. (2013). Best practice measures assessment for construction and demolition waste management in building constructions. *Resources, Conservation and Recycling*, vol. 75, pp. 52-62.

Saghafi, M. D. & Hosseini Teshnizi, Z. S. (2011). Recycling value of building materials in building assessment systems. *Energy and Buildings*, vol. *43* (11), pp. 3181–3188.

Sales, A. & de Souza, F. (2009). Concretes and mortars recycled with water treatment sludge and construction and demolition rubble. *Construction and Building Materials*, vol. 23 (6), pp. 2362-2370.

Sander the Bruyn, S., Ahdour, S., Bijleveld, M., Graaff, L., Schep, E., Schroten, A., Vergeer, R. (2018). Handbook Environmental Prices 2017, *Ce Delft.* pp. 1-177.

Saravanakumar, P. & Dhinakaran, G. (2013). Strength Characteristics of High-Volume Fly Ash–Based Recycled Aggregate Concrete. *Journal of Materials in Civil Engineering*, vol. 25 (8), pp. 1127-1133.

Sartori, D., Catalano, G., Genco, M., Pancotti, C., Sirtori, E., Vignetti, S., Bo, C. (2014). Guide to Cost-Benefit Analysis of Investment Projects. *Economic appraisal tool for Cohesion Policy* 2014-2020.

Scientific Applications International Corporation (2006). Life Cycle Assessment: Principle and Practice. Contract No. 68-C02-067 Work Assignment 3-15. Cincinnati, Ohio, National Risk Management Research Laboratory, Office Of Research And Development, U.S. Environmental Protection Agency U.S. Environmental Protection Agency.

Sharaai, A. H., Mahmood, N. Z. & Sulaiman, A. H. (2010). Life Cycle Impact Assessment (LCIA) Using the Ecological Scarcity (Ecopoints) Method: A Potential Impact Analysis to Potable Water Production. *World Applied Science Journal*, vol. 11 (9), pp. 1077-1088.

Shen, L., Tam, V., Tam, C. & Drew, D. (2004). Mapping Approach for Examining Waste Management on Construction Sites. *Journal of Construction Engineering and Management*, vol. 130 (4), pp. 472-481.

Shen, L., Tam, V., Tam, L. & Ji, Y. (2010). Project feasibility study: the key to successful implementation of sustainable and socially responsible construction management practice. *Journal of Cleaner Production*, vol. 18 (3), pp. 254-259.

Shin, D., Cha, H., Kim, J. & Kim, J. (2006). Development of the Construction Waste Management Performance Evaluation Tool (WMPET). pp. 263-267.

Simões, C. L., Xará, S. M. & Bernardo, C. A. (2011). Influence of the impact assessment method on the conclusions of a LCA study. Application to the case of a part made with virgin and recycled HDPE. *Waste Management and Research*, vol. 29 (10), pp. 1018-1026.

Sleeswijk, A. W., van Oers, L. F., Guinée, J. B., Struijs, J., Huijbregts, M. A. (2008). Normalisation in product life cycle assessment: An LCA of the global and European economic systems in the year 2000. *Science of the Total Environment*, vol. 390 (1), pp. 227-240.

Smith, K., O'Farrell, K. & Brindley, F. (2012). Waste and recycling in Australia 2011. Hyder for Department of Environment Water Heritage and the Arts: Canberra, Australia.

Spangenberg, J.H. (2007). Biodiversity pressure and the driving forces behind. *Ecological economics*, vol. 61 (1), pp. 146-158.

Statistics Centre in Abu Dhabi". (2016) [Accessed 18 April 2018]. Available at: http://www.scad.ae

Steen, B. (1999:5). A systematic approach to environmental priority strategies in product development (EPS). *Version 2000 – Models and data of the default method. CPM report.*

Steen, B. (2015). The EPS 2015d Impact Assessment Method – An Overview. *Swedish Life Cycle Center*, Report number 2015:5.

Steen, B., (1999). A systematic Approach to Environmental Priority Strategies in Product Development (EPS). Version 2000 - *General System Characteristics &Models and Data of the Default Method*. *CPM report, Chalmers*, pp. 4-5.

Stocker, T.F., D. Qin, G.K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y., Xia, V., Bex, P.M Midgley. (2013). *Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA*, pp. 1535.

Struijs, J., van Dijk, A., Slaper, H., van Wijnen, H. J., Velders, G. J., Chaplin, G., Huijbregts, M. A. J. (2009). Spatial-and time-explicit human damage modelling of ozone depleting substances in life cycle impact assessment. *Environmental Science & Technology*, vol. 44 (1), pp. 204-209.

Substudy 6 – what LCA can tell us about the cement industry, *An Independent Study Commissioned by the World Business Council for Sustainable Development*.

Sustainable Urban Futures," United Nations University, Institute for Advanced Study of Sustainability, 30 June 2016. [Online]. Available: http://urban.ias.unu.

Symonds Group, Construction and Demolition Waste Management Practices and their Economic Impacts. (1999). Report to DGXI, European Commission, London: Symonds Group.

Tam, V. & Tam, C. (2006). Evaluations of existing waste recycling methods: A Hong Kong study. Building and Environment, vol. 41 (12), pp. 1649-1660.

Tam, V. (2008). Economic comparison of concrete recycling: A case study approach. *Resources, Conservation and Recycling*, vol. 52 (5), pp. 821-828.

Tam, V. (2009). Comparing the implementation of concrete recycling in the Australian and Japanese construction industries. *Journal of Cleaner Production*, vol. 17 (7), pp. 688-702.

Tam, V. W. Y., Tam, L. & Le, K. N. (2010). Cross-cultural comparison of concrete recycling decision-making and implementation in construction industry. *Waste Management*, vol. 30 (2), pp. 291–297.

Tam, V., Tam, L. & Le, K. (2010). Cross-cultural comparison of concrete recycling decision-making and implementation in construction industry. Waste Management, vol. 30 (2), pp. 291-297.

Thompson, M., Ellis, R., Wildavsky, A., (1990). Cultural Theory; Westview Press: Boulder, CO.

Tillman, A.M., Ekvall, T., Baumann, H. & Rydberg, T. (1994). Choice of system boundaries in life cycle assessment, *Journal of Cleaner Production*, vol. 2 (1), pp. 21-29.

Tojo, N. & Fischer, C. (2011). European Recycling Policies in Relation to the Actual Recycling Achieved. *Europe as a Recycling Society*, working paper, (ETC/SCP working paper 2/2011).

Tojo, N. (2010). European Topic Centre on Sustainable Consumption and Production. *Europe as a Recycling Society*, pp. 6.

Townsend, T., Wilson, C. & Beck, B. (2015). The Benefits of Construction and Demolition Materials Recycling in the United States. *White Paper, Construction and Demolition Recycling Association*.

Toxopeus, M. E., Lutters, E. & van Houten, F. (2006). Environmental Indicators & Engineering: an Alternative for Weighting Factors. *LCE 13 TH CIRP International Conference on Life Cycle Engineering*, pp. 75-80.

Trapp, S. (2009). Bioaccumulation of polar and ionizable compounds in plants. *Ecotoxicology Modelling*.

Tuomisto, J., Vinneau, D., Wagner, S., Yang, A. (2011). Methods and results of the HEIMTSA/INTARESE Common Case Study. *The Institute of Occupational Medicine*. [Accessed 18 April 2019]. Available at: http://www.integratedassessment.eu/sites/default/files/CCS_FINAL_REPORT_final.pdf.

Tuomisto, JT., Wilson, A., Evans, JS., Tainio, M. (2008). Uncertainty in mortality response to airborne fine particulate matter: Combining European air pollution experts. *Reliab Eng Syst Safe*, vol. 93 (5), pp. 732- 744.

U.S. Geological Survey. (2014). Mineral commodity summaries 2014. *U.S. Geological Survey*, pp. 196.

U.S. Geological Survey. (2014). Mineral commodity summaries 2014. *U.S. Geological Survey*, pp. 196.

Udawatta, N., Zuo, J., Chiveralls, K. & Zillante, G. (2015). Improving waste management in construction projects: An Australian study. *Resources, Conservation and Recycling*, vol. 101, pp. 73-83.

UK Essays | UKEssays". (2013). [Accessed 20 April 2018]. Available at: https://www.ukessays.com/

Ulubeyli, S., Kazaz, A. & Arslan, V. (2017). Construction and Demolition Waste Recycling Plants Revisited: Management Issues. Procedia Engineering, vol. 172, pp. 1190-1197.

UNEP (2003). Evaluation of Environmental Impacts in Life Cycle Assessment. Meeting report - Brussels, 29-30 November 1998, and Brighton, 25-26 May 2000. Paris, France, United Nations Environment Programme (UNEP), Division of Technology, Industry and Economics (DTIE).

UNEP. (2009). Buildings and Climate Change, Summary for Decision-Makers United.

United Nations (1991). Protocol to the convention on long-range transboundary air pollution concerning the control of emissions of volatile organic compounds or their transboundary fluxes. Geneva, Switzerland, Economic Commission for Europe (UNECE).

Urban, M.C. (2015). Accelerating extinction risk from climate change. *Science*, vol. 348, pp. 571-573.

USEPA". (2013). Construction & Demolition Materials [Accessed 18 April 2018]. Available at: http://www.epa.gov/osw/nonhaz/industrial/cd/index.htm

Van Dingenen, R., Dentener, FJ., Raes, F., Krol, MC., Emberson, L., Cofala, J. (2009). The global impact of ozone on agricultural crop yields under current and future air quality legislation. *Atmospheric Environment*, vol. 43 (3), pp. 604-618.

Van Ewijk, S. & Stegemann, J. (2016). Limitations of the waste hierarchy for achieving absolute reductions in material throughput. *Journal of Cleaner Production*, vol. 132, pp. 122-128.

Van Geem, M. (1998). The portland cement association's environmental life cycle assessment of concrete', Air and Waste Management Association's 91st Annual Meeting and Exhibition, San Diego, California, USA, June 14–18.

Van Goethem, T., Azevedo, LB., Van Zelm, R., Hayes, RM., Ashmore, MR., Huijbregts, MAJ. (2013a). Plant Species Sensitivity Distributions for ozone exposure. *Environmental Pollution*, vol. 178, pp. 1-6.

Van Goethem, T., Preiss, P., Azevedo, LB., Friedrich, R., Huijbregts, MAJ., Van Zelm, R. (2013b). European characterization factors for damage to natural vegetation by ozone in life cycle impact assessment. *Atmospheric Environment*, vol. 77, pp. 318-324.

Van Zelm, R., Huijbregts, MAJ., Den Hollander, HA., Van Jaarsveld, HA., Sauter, FJ., Struijs, J., Van Wijnen, HJ., Van de Meent, D. (2008). European characterization factors for human health damage due to PM10 and ozone in life cycle impact assessment. *Atmospheric Environment*, 42 (3), pp. 441-453.

Van Zelm, R., Huijbregts, MAJ., Van de Meent, D. (2009). USES-LCA 2.0: a global nested multi-media fate, exposure and effects model. *The International Journal of LCA*, vol. 14 (30), pp. 282-284.

Van Zelm, R., Preiss, P., Van Goethem, T., Van Dingenen, R., Huijbregts, MAJ. (2016). Regionalized life cycle impact assessment of air pollution on the global scale: damage to human health and vegetation. *Atmospheric Environment*, vol. 134, pp. 129-137.

Van Zelm, R., Preiss, P., Van Goethem, T., Van Dingenen, R., Huijbregts, MAJ. (2016). Regionalized life cycle impact assessment of air pollution on the global scale: damage to human health and vegetation. *Atmospheric Environment*, vol. 134, pp. 129-137.

Van Zelm, R., Roy, PO., Hauschild, MZ., Huijbregts, MAJ. (2015). Acidification. In: Life Cycle Impact Assessment. LCA Compendium. *The Complete World of Life Cycle Assessment*, pp. 163-176.

Van Zelm, R., Stam, G., Huijbregts, MAJ., Van de Meent, D. (2013). Making fate and exposure models for freshwater ecotoxicity in life cycle assessment suitable for organic acids and bases. *Chemosphere*, vol. 90 (2), pp. 312-317.

Verones, F., Huijbregts, M.A., Chaudhary, A., de Baan, L., Koellner, T. & Hellweg, S. (2015). Harmonizing the assessment of biodiversity effects from land and water use within LCA. *Environmental science & technology*, vol. 49 (6), pp. 3584-3592.

Verones, F., Saner, D., Pfister, S., Baisero, D., Rondinini, C., Hellweg, S. (2013). Effects of consumptive water use on wetlands of international importance. *Environmental Science & Technology*, vol. 47 (21), pp. 12248-12257.

Verones, F., Saner, D., Pfister, S., Baisero, D., Rondinini, C., Hellweg, S. (2013). Effects of consumptive water use on wetlands of international importance. *Environ. Sci. Technol*, vol. 47 (21), pp. 12248-12257.

Vieira, C. & Pereira, P. (2015). Use of recycled construction and demolition materials in geotechnical applications: A review. *Resources, Conservation and Recycling*, vol. 103, pp. 192-204.

Vieira, MDM., Huijbregts, MAJ. In preparation. Evaluating mineral and fossil resource scarcity trade-offs between energy technologies.

Vieira, MDM., Huijbregts, MAJ. In preparation. Evaluating mineral and fossil resource scarcity trade-offs between energy technologies.

Vieira, MDM., Ponsioen, TC., Goedkoop, M., Huijbregts, MAJ. (2012). Ore grade decrease as life cycle impact indicator for metal scarcity: the case of copper. *Environmental Science & Technology*, vol. 46 (23), pp 12772-12778.

Vieira, MDM., Ponsioen, TC., Goedkoop, M., Huijbregts, MAJ. (2016a). Surplus cost potential as a Life Cycle Impact Indicator for metal extraction. *Resources*, vol. 5 (1), pp. 1-12.

Vieira, MDM., Ponsioen, TC., Goedkoop, M., Huijbregts, MAJ. (2016b). Surplus ore potential as a scarcity indicator for resource extraction. *Journal of Industrial Ecology*.

Vieira, MDM., Ponsioen, TC., Goedkoop, M., Huijbregts, MAJ. (2016a). Surplus cost potential as a Life Cycle Impact Indicator for metal extraction. *Resources*, vol. 5 (2), pp. 1-12.

Vieira, MDM., Ponsioen, TC., Goedkoop, M., Huijbregts, MAJ. (2016b). Surplus ore potential as a scarcity indicator for resource extraction. Journal of Industrial Ecology.

Viswanathan, S. & Norbu, T. (2008). Implementation of 3R concept in the countries of East and South-East Asia. *Municipal Solid Waste*, vol. 11, pp. 54-60.

Wang, H. & Gangaram, R. (2014). Life Cycle Assessment of Asphalt Pavement Maintenance. *Center for Advanced Infrastructure & Transportation Rutgers, The State University of New Jersey*. pp. 1-67.

Warfvinge, P., Sverdrup, H. (1991). Calculating critical loads of acid deposition with PROFILE: a steady-state soil chemistry model. *Water Air Soil Pollution*, pp. 119–43.

Waste Management". (2018). [Accessed 18 April 2018]. Available at: http://government.ae

Weidema, B.P. (2009). Using the budget constraint to monetarise impact assessment results. *Ecol. Econ*, vol. 68, pp. 1591–1598.

Weil, M., Jeske, U. & Schebek, L. (2006). Closed-loop recycling of construction and demolition waste in Germany in view of stricter environmental threshold values. *Waste Management & Research*, vol. 24 (3), pp. 197-206.

Weiler, V., Harter, H. & Eicker, U. (2017). Life cycle assessment of buildings and city quarters comparing demolition and reconstruction with refurbishment. *Energy and Buildings*, vol. 134, pp. 319-328.

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., Weidema, B. (2016). The ecoinvent database version 3 (part I): overview and methodology. *The International Journal of Life Cycle Assessment*, vol. 21 (9), pp. 1218–1230.

'What a Waste : A Global Review of Solid Waste Management". (2012). vol. 15. [Accessed 15 April 2018].

WHO. (2003). Health aspects of air pollution with particulate matter, ozone and nitrogen dioxide; report on a WHO working group. *World Health Organization*, Bonn, Germany, 13-15 January 2003.

WHO. (2004). Health aspects of air pollution – answers to follow-up questions from CAFE. Report on a WHO working group meeting. *World Health Organization*, Bonn, Germany, 15–16 January 2004.

WHO. (2006). Health risks of particulate matter from long-range transboundary air pollution. *World Health Organization*, Copenhagen, Denmark.

WHO. (2013). Health risks of air pollution in Europe-HRAPIE project recommendations for concentration-response functions for cost benefit analysis of particulate matter, ozone and nitrogen dioxide. *World Health Organization*, Geneva, Switzerland.

Wiedmann, T. (2014). EDITORIAL: CARBON FOOTPRINT AND INPUT–OUTPUT ANALYSIS – AN INTRODUCTION. *Economic Systems Research*, vol. 21 (3), pp. 175-186.

WMO. (2011). Scientific assessment of ozone depletion: 2010, Global Ozone Research and Monitoring Project-report no.52. *World Meteorological Organization*, Geneva.

Won, J. & Cheng, J. (2017). Identifying potential opportunities of building information modeling for construction and demolition waste management and minimization. *Automation in Construction*, vol. 79, pp. 3-18.

World Meteorological Organisation (1991). Scientific assessment of ozone depletion. Global Ozone Research and Monitoring Project - Report no. 25 .

World Mine Cost Data Exchange. (2014). Cost Curves. [Accessed 18 April 2019]. Available at: www.minecost.com/curves.htm

World Water Council. (nd). The Use of Water Today. [Accessed 18 April 2019]. Available at: http://www.worldwatercouncil.org/fileadmin/wwc/Library/WWVision/Chapter2.pdf.

Worrell, E., Price, L., Martin, N., Hendriks, C. & Ozawa Meida, L. (2001). Carbon dioxide emissions from the global cement industry, *Annual Review Energy Environment*, vol. 26, pp.303-329.

Wu, Z., Yu, A. & Shen, L. (2017). Investigating the determinants of contractor's construction and demolition waste management behavior in Mainland China. *Waste Management*, vol. 60, pp. 290-300.

Xenopoulos, M. A., Lodge, D. M. (2006). Going with the flow: using species-discharge relationships to forecast losses in fish biodiversity. *Ecology*, vol. 87 (8), pp. 1907-1914.

Xenopoulos, M. A., Lodge, D. M., Alcamo, J., Märker, M., Schulze, K., Van Vuuren, D. P. (2005). Scenarios of freshwater fish extinctions from climate change and water withdrawal. *Global Change Biology*, vol. 11 (10), pp. 1557-1564.

Yahya, K. & Boussabaine, H. (2010). Quantifying environmental impacts and eco-costs from brick waste. Architectural Engineering and Design Management, vol. 6 (3), pp. 189-206.

Yahya, K. (2006). ECO-COSTING OF BUILDING CONSTRUCTION WASTES.

Yahya, K., Boussabaine, H., Alzaed. A. (2016). Using life cycle assessment for estimating environmental impacts and eco-costs from the metal waste in the construction industry, *Management of Environmental Quality: An International Journal*, vol. 27 (2), pp. 227-244.

Yanik, K. Report-global demand for aggregates to rise ". (2016). [Accessed 18 April 2018]. Available at: http://www.pitandquarry.com/report-global-demand-for-aggregates-to-rise.

Young, S., Turnbull, S. & Russell, A. (2002). Toward a sustainable cement industry.

Yu, A., Poon, C., Wong, A., Yip, R. & Jaillon, L. (2013). Impact of Construction Waste Disposal Charging Scheme on work practices at construction sites in Hong Kong. *Waste Management*, vol. 33 (1), pp. 138-146.

Yuan, H. & Shen, L. (2011). Trend of the research on construction and demolition waste management. *Waste Management*, vol. 31 (4), pp. 670-679.

Yuan, H. & Shen, L. (2011). Trend of the research on construction and demolition waste management. *Waste Management*, vol. 31 (4), pp. 670-679.

Yuan, H. (2012). A model for evaluating the social performance of construction waste management. *Waste Management*, vol. 32 (6), pp. 1218-1228.

Yuan, H. (2013). Critical Management Measures Contributing to Construction Waste Management: Evidence From Construction Projects in China. *Project Management Journal*, vol. 44 (4), pp. 101-112.

Yuan, H. (2017). Barriers and countermeasures for managing construction and demolition waste: A case of Shenzhen in China. *Journal of Cleaner Production*, vol. 157, pp. 84-93.

Zabalza Bribián, I., Valero Capilla, A. & Aranda Usón, A. (2011). Life cycle assessment of building materials: Comparative analysis of energy and environmental impacts and evaluation of the eco-efficiency improvement potential. *Building and Environment*, vol. 46 (5), pp. 1133–1140.

Zhao, W., Leeftink, R. & Rotter, V. (2010). Evaluation of the economic feasibility for the recycling of construction and demolition waste in China-The case of Chongqing. *Resources, Conservation and Recycling*, vol. 54 (6), pp. 377-389.

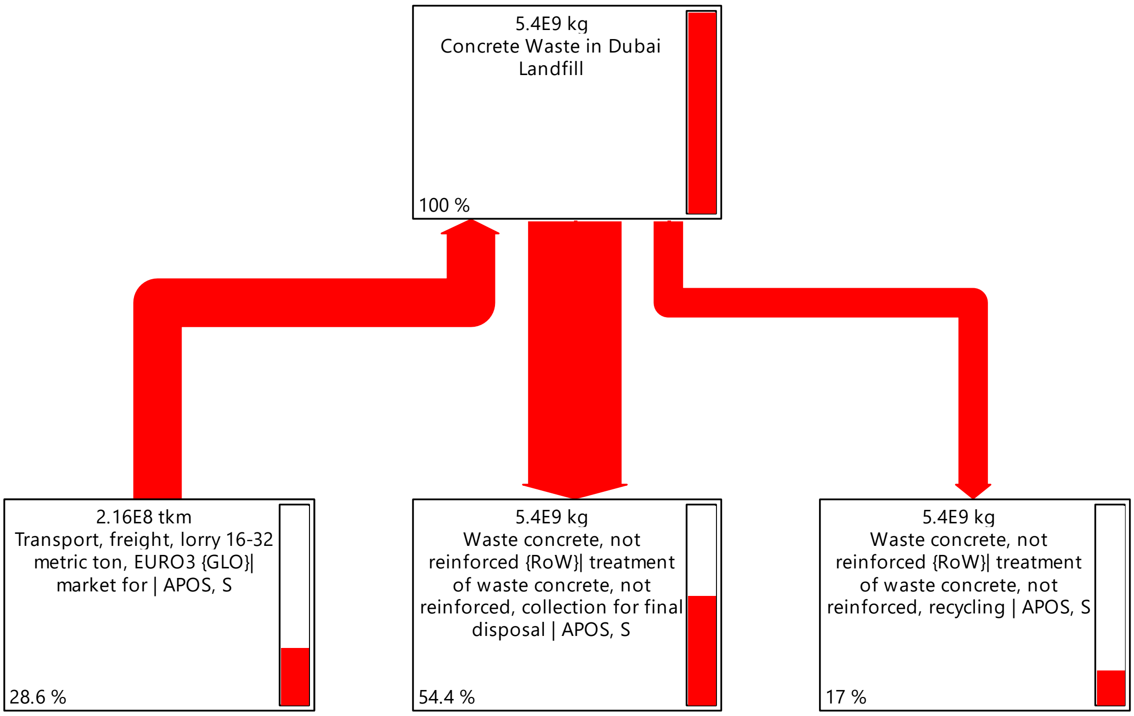
Zhao, W., Leeftink, R. & Rotter, V. (2010). Evaluation of the economic feasibility for the recycling of construction and demolition waste in China - The case of Chongqing. *Resources, Conservation and Recycling*, vol. 54 (6), pp. 377-389.

Zhao, W., Leeftink, R.B. & Rotter, V.S. (2010). Evaluation of the economic feasibility for the recycling of construction and demolition waste in China—The case of Chongqing. *Resources, Conservation and Recycling*, vol. *54* (6), pp.377-389.

Zhao, W., Ren, H. & Rotter, V. (2011). A system dynamics model for evaluating the alternative of type in construction and demolition waste recycling center - The case of Chongqing, China. *Resources, Conservation and Recycling*, vol. 55 (11), pp. 933-944.

APPENDICES

Appendix 1



Overall distribution of 18 midpoint impacts (SimaPro software, ReCiPe2016 Midpoint Method)

Results of each midpoint impact category in percentage (SimaPro software, ReCiPe2016 Midpoint Method)

No.	Impact category	Unit	Total	Transport, freight, lorry 16-32 metric ton, EURO3 {GLO}| market for | APOS, S	Waste concrete, not reinforced {RoW}| treatment of waste concrete, not reinforced, collection for final disposal | APOS, S	Waste concrete, not reinforced {RoW}| treatment of waste concrete, not reinforced, recycling | APOS, S
1	Global warming	%	100	28.552308	54.406576	17.041116
2	Stratospheric ozone depletion	%	100	19.507118	61.724496	18.768385
3	Ionising radiation	%	100	27.52683	62.503311	9.9698591
4	Ozone formation, Human health	%	100	22.325248	54.093047	23.581705
5	Fine particulate matter formation	%	100	6.2227821	52.965797	40.811421
6	Ozone formation, Terrestrial ecosystems	%	100	22.289366	54.152318	23.558316
7	Terrestrial acidification	%	100	22.846219	57.037924	20.115857
8	Freshwater eutrophication	%	100	27.511629	63.093617	9.3947543
9	Marine eutrophication	%	100	28.219441	62.569358	9.2112014
10	Terrestrial ecotoxicity	%	100	6.00E+01	3.75E+01	2.5311763
11	Freshwater ecotoxicity	%	100	42.102866	51.091774	6.8053601
12	Marine ecotoxicity	%	100	44.085439	49.40984	6.5047214
13	Human carcinogenic toxicity	%	100	65.979798	31.546908	2.4732936
14	Human non-carcinogenic toxicity	%	100	68.659339	28.577095	2.7635656
15	Land use	%	100	20.028283	79.278425	0.69329232
16	Mineral resource scarcity	%	100	29.611229	56.334572	14.054199

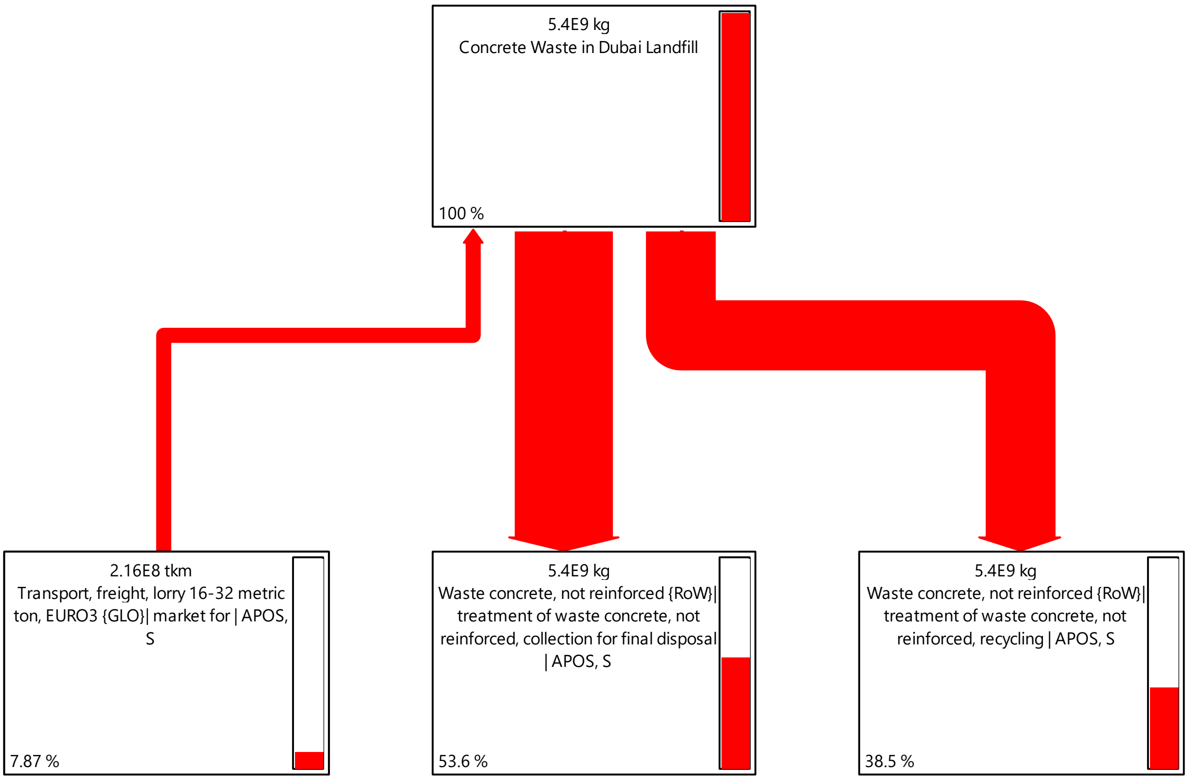
17	Fossil resource scarcity	%	100	24.380207	61.526213	14.09358
18	Water consumption	%	100	8.7578626	87.487716	3.7544213

Results of each midpoint impact category in units (SimaPro software, ReCiPe2016 Midpoint Method)

No.	Impact category	Unit	Total	Transport, freight, lorry 16-32 metric ton, EURO3 {GLO}| market for | APOS, S	Waste concrete, not reinforced {RoW}| treatment of waste concrete, not reinforced, collection for final disposal | APOS, S	Waste concrete, not reinforced {RoW}| treatment of waste concrete, not reinforced, recycling | APOS, S
1	Global warming	kg CO2 eq	1.33E+08	37934796	72284957	22640946
2	Stratospheric ozone depletion	kg CFC11 eq	48.221577	9.4066401	29.764525	9.0504113
3	Ionising radiation	kBq Co-60 eq	2188230.1	602350.37	1367716.3	218163.46
4	Ozone formation, Human health	kg NOx eq	1111351.4	248111.96	601163.86	262075.62
5	Fine particulate matter formation	kg PM2.5 eq	280275.94	17440.961	148450.38	114384.59
6	Ozone formation, Terrestrial ecosystems	kg NOx eq	1130378.6	251954.24	612126.24	266298.17
7	Terrestrial acidification	kg SO2 eq	610854.85	139557.24	348418.92	122878.69
8	Freshwater eutrophication	kg P eq	10089.541	2775.7969	6365.8561	947.88755
9	Marine eutrophication	kg N eq	796.02934	224.63503	498.07045	73.323865
10	Terrestrial ecotoxicity	kg 1,4-DCB	3.44E+08	2.06E+08	1.29E+08	8705868.7
11	Freshwater ecotoxicity	kg 1,4-DCB	1233661.4	519406.79	630299.48	83955.099

12	Marine ecotoxicity	kg 1,4-DCB	493435.25	217533.09	243805.57	32096.588
13	Human carcinogenic toxicity	kg 1,4-DCB	68428.521	45149	21587.083	1692.4382
14	Human non-carcinogenic toxicity	kg 1,4-DCB	2610770.9	1792538.1	746082.49	72150.367
15	Land use	m2a crop eq	7661498.2	1534466.5	6073915	53116.578
16	Mineral resource scarcity	kg Cu eq	157103.11	46520.161	88503.364	22079.584
17	Fossil resource scarcity	kg oil eq	51561063	12570694	31723569	7266799.6
18	Water consumption	m3	1157545.8	101376.27	1012710.4	43459.147

Appendix 2



Overall distribution of 22 endpoint impacts (SimaPro software, ReCiPe2016 Endpoint Method)

Overall distribution of 22 endpoint impacts (SimaPro software, ReCiPe2016 Endpoint Method)

Results of each endpoint impact category in percentage (SimaPro software, ReCiPe2016 Endpoint Method)

2018						
No.	**Impact category**	**Unit**	**Total**	**Transport, freight, lorry 16-32 metric ton, EURO3 {GLO}| market for | APOS, S**	**Waste concrete, not reinforced {RoW}| treatment of waste concrete, not reinforced, collection for final disposal | APOS, S**	**Waste concrete, not reinforced {RoW}| treatment of waste concrete, not reinforced, recycling | APOS, S**
1	Global warming, Human health	%	100	28.552577	54.406241	17.041183
2	Global warming, Terrestrial ecosystems	%	100	28.552773	54.406042	17.041185
3	Global warming, Freshwater ecosystems	%	100	2.86E+01	5.44E+01	1.70E+01
4	Stratospheric ozone depletion	%	100	19.518091	61.722473	18.759436
5	Ionising radiation	%	100	27.525998	62.502827	9.9711752
6	Ozone formation, Human health	%	100	22.325174	54.093153	23.581673
7	Fine particulate matter formation	%	100	6.2227821	52.965797	40.811421
8	Ozone formation, Terrestrial ecosystems	%	100	22.289414	54.152286	23.558301
9	Terrestrial acidification	%	100	22.846319	57.038474	20.115207
10	Freshwater eutrophication	%	100	27.511576	63.093681	9.3947428
11	Marine eutrophication	%	100	2.82E+01	6.26E+01	9.21E+00
12	Terrestrial ecotoxicity	%	100	59.951424	37.517266	2.53E+00
13	Freshwater ecotoxicity	%	100	42.091692	51.099663	6.81E+00
14	Marine ecotoxicity	%	100	4.41E+01	4.94E+01	6.51E+00
15	Human carcinogenic toxicity	%	100	65.965169	31.556087	2.4787441

16	Human non-carcinogenic toxicity	%	100	68.648486	28.585846	2.765668
17	Land use	%	100	20.030437	79.276389	0.69317342
18	Mineral resource scarcity	%	100	29.610219	56.3342	14.055581
19	Fossil resource scarcity	%	100	24.303632	61.326859	14.369509
20	Water consumption, Human health	%	100	8.7578626	87.487716	3.7544213
21	Water consumption, Terrestrial ecosystem	%	100	0	0	0
22	Water consumption, Aquatic ecosystems	%	100	8.76E+00	8.75E+01	3.75E+00

Appendix 3

Results of final endpoint impact in percentage (SimaPro software, ReCiPe2016 Endpoint Method)

2018						
No.	**Damage category**	**Unit**	**Total**	**Transport, freight, lorry 16-32 metric ton, EURO3 {GLO}| market for | APOS, S**	**Waste concrete, not reinforced {RoW}| treatment of waste concrete, not reinforced, collection for final disposal | APOS, S**	**Waste concrete, not reinforced {RoW}| treatment of waste concrete, not reinforced, recycling | APOS, S**
1	Human Health	%	100	7.8713191	53.596642	38.532039
2	Ecosystems	%	100	23.610405	59.069045	17.32055
3	Resources	%	100	24.309603	61.321241	14.369156

Results of final endpoint impact in units (SimaPro software, ReCiPe2016 Endpoint Method)

2018						
No.	**Damage category**	**Unit**	**Total**	**Transport, freight, lorry 16-32 metric ton, EURO3 {GLO}| market for | APOS, S**	**Waste concrete, not reinforced {RoW}| treatment of waste concrete, not reinforced, collection for final disposal | APOS, S**	**Waste concrete, not reinforced {RoW}| treatment of waste concrete, not reinforced, recycling | APOS, S**
1	Human Health	DALY	192.52864	15.154544	103.18889	74.18521
2	Ecosystems	species.yr	0.42555908	0.10047622	0.25137368	0.07370917
3	Resources	USD2013	22253737	5409795.1	13646268	3197674.1

Appendix 4

LCA results of 2018 and its damage cost

LCA Results of 2018											
No.	**Impact category**	**Unit**	**Total**	**Transportation**	**Landfilling**	**Recycling**	**Price per unit in Euro**	**Damage Cost of Transportation**	**Damage Cost of Landfilling**	**Damage Cost of Recycling**	**Total**
1	Global Warming	kg CO2 eq	132860700	37934796	72284957	22640946	0.057	€ 2,162,283.37	€ 4,120,242.55	€ 1,290,533.92	€ 7,573,059.84
2	Stratospheric Ozone Depletion	kg CFC 11 eq	48.221577	9.4066401	29.764525	9.0504113	30.4	€ 285.96	€ 904.84	€ 275.13	€ 1,465.94
3	Ozone Formation, Human Health	kg NOx eq	1111351.4	248111.96	601163.86	262075.62	18.7	€ 4,639,693.65	€ 11,241,764.18	€ 4,900,814.09	€ 20,782,271.93
4	Fine Particulate Matter Formation	kg PM2.5 eq	280275.94	17440.961	148450.38	114384.59	79.5	€ 1,386,556.40	€ 11,801,805.21	€ 9,093,574.91	€ 22,281,936.51
5	Ozone Formation, Terrestrial Ecosystems	kg NOx eq	1130378.6	251954.24	612126.24	266298.17	18.7	€ 4,711,544.29	€ 11,446,760.69	€ 4,979,775.78	€ 21,138,080.76
6	Terrestrial Acidification	kg SO2 eq	610854.85	139557.24	348418.92	122878.69	5.4	€ 753,609.10	€ 1,881,462.17	€ 663,544.93	€ 3,298,616.19
7	Freshwater Eutrophication	kg P eq	10089.541	2775.7969	6365.8561	947.88755	1.9	€ 5,274.01	€ 12,095.13	€ 1,800.99	€ 19,170.13

8	Marine Eutrophication	kg N eq	796.02934	224.63503	498.07045	73.323865	3.11	€ 698.61	€ 1,549.00	€ 228.04	€ 2,475.65
9	Terrestrial Ecotoxicity	kg 1,4-DCB	343945570	206202510	129037190	8705868.7	8.89	€ 1,833,140,313.90	€ 1,147,140,619.10	€ 77,395,172.74	€ 3,057,676,105.74
10	Freshwater Ecotoxicity	kg 1,4-DCB	1233661.4	519406.79	630299.48	83955.099	0.0369	€ 19,166.11	€ 23,258.05	€ 3,097.94	€ 45,522.10
11	Marine Ecotoxicity	kg 1,4-DCB	493435.25	217533.09	243805.57	32096.588	0.00756	€ 1,644.55	€ 1,843.17	€ 242.65	€ 3,730.37
12	Human Carcinogenic Toxicity	kg 1,4-DCB	68428.521	45149	21587.083	1692.4382	0.214	€ 9,661.89	€ 4,619.64	€ 362.18	€ 14,643.70
13	Land Use	m2a crop eq	7661498.2	1534466.5	6073915	53116.578	0.0261	€ 40,049.58	€ 158,529.18	€ 1,386.34	€ 199,965.10
14	Mineral Resource Scarcity (Atmospheric Emissions)	kg Cu eq	157103.11	46520.161	88503.364	22079.584	4.2	€ 195,384.68	€ 371,714.13	€ 92,734.25	€ 659,833.06
15	Mineral Resource Scarcity (Emissions To Water - Soil)	kg Cu eq	157103.11	46520.161	88503.364	22079.584	0.239	€ 11,118.32	€ 21,152.30	€ 5,277.02	€ 37,547.64
16	Ionising radiation	kBq Co-60 eq	2188230.1	602350.37	1367716.3	218163.46	-	-	-	-	-
17	Human non-carcinogenic toxicity	kg 1,4-DCB	2610770.9	1792538.1	746082.49	72150.367	-	-	-	-	-
18	Water consumption	m3	1157545.8	101376.27	1012710.4	43459.147	-	-	-	-	-

LCA results of 2017 and its damage cost

LCA Results of 2017											
No.	**Impact category**	**Unit**	**Total**	**Transportation**	**Landfilling**	**Recycling**	**Price per unit**	**Damage Cost of Transportation**	**Damage Cost of Landfilling**	**Damage Cost of Recycling**	**Total**
1	Global warming	kg CO2 eq	123019170	35124811	66930516	20963839	0.057	2002114.227	3815039.412	1194938.823	€ 7,012,092.46
2	Stratospheric ozone depletion	kg CFC11 eq	44.649608	8.7098519	27.559745	8.3800105	30.4	264.7794978	837.816248	254.7523192	€ 1,357.35
3	Ozone formation, Human health	kg NOx eq	1029029.1	229733.3	556633.2	242662.61	18.7	4296012.71	10409040.84	4537790.807	€ 19,242,844.36
4	Fine particulate matter formation	kg PM2.5 eq	259514.76	16149.038	137454.06	105911.66	79.5	1283848.521	10927597.77	8419976.97	€ 20,631,423.26
5	Ozone formation, Terrestrial ecosystems	kg NOx eq	1046646.9	233290.96	566783.55	246572.38	18.7	4362540.952	10598852.39	4610903.506	€ 19,572,296.84
6	Terrestrial acidification	kg SO2 eq	565606.34	129219.66	322610.11	113776.56	5.4	697786.164	1742094.594	614393.424	€ 3,054,274.18
7	Freshwater eutrophication	kg P eq	9342.1672	2570.1823	5894.3112	877.67366	1.9	4883.34637	11199.19128	1667.579954	€ 17,750.12

8	Marine eutrophication	kg N eq	737.0642	207.9954	461.17634	67.892468	3.11	646.865694	1434.258417	211.1455755	€ 2,292.27
9	Terrestrial ecotoxicity	kg 1,4-DCB	318468120	190928250	119478880	8060989.6	8.89	1697352143	1062167243	71662197.54	€ 2,831,181,583.24
10	Freshwater ecotoxicity	kg 1,4-DCB	1142279	480932.21	583610.63	77736.203	0.0369	17746.39855	21535.23225	2868.465891	€ 42,150.10
11	Marine ecotoxicity	kg 1,4-DCB	456884.49	201419.53	225745.89	29719.063	0.00756	1522.731647	1706.638928	224.6761163	€ 3,454.05
12	Human carcinogenic toxicity	kg 1,4-DCB	63359.741	41804.63	19988.039	1567.0724	0.214	8946.19082	4277.440346	335.3534936	€ 13,558.98
13	Land use	m2a crop eq	7093979.8	1420802.4	5623995.4	49182.017	0.0261	37082.94264	146786.2799	1283.650644	€ 185,152.87
14	Mineral resource scarcity (atmospheric emissions)	kg Cu eq	145465.84	43074.224	81947.559	20444.059	4.2	180911.7408	344179.7478	85865.0478	€ 610,956.54
15	Mineral resource scarcity (emissions to water - soil)	kg Cu eq	145465.84	43074.224	81947.559	20444.059	0.239	10294.73954	19585.4666	4886.130101	€ 34,766.34

LCA results of 2016 and its damage cost

LCA Results of 2016											
No.	**Impact category**	**Unit**	**Total**	**Transportation**	**Landfilling**	**Recycling**	**Price per unit**	**Damage Cost of Transportation**	**Damage Cost of Landfilling**	**Damage Cost of Recycling**	**Total**
1	Global warming	kg CO2 eq	1.16E+08	33017322	62914685	19706009	0.057	1881987.354	3586137.045	1123242.513	€ 6,591,366.91
2	Stratospheric ozone depletion	kg CFC11 eq	41.970631	8.1872608	25.906161	7.8772099	30.4	248.8927283	787.5472944	239.467181	€ 1,275.91
3	Ozone formation, Human health	kg NOx eq	967287.37	215949.3	523235.21	228102.85	18.7	4038251.91	9784498.427	4265523.295	€ 18,088,273.63
4	Fine particulate matter formation	kg PM2.5 eq	243943.87	15180.096	129206.81	99556.961	79.5	1206817.632	10271941.4	7914778.4	€ 19,393,537.43
5	Ozone formation, Terrestrial ecosystems	kg NOx eq	983848.08	219293.5	532776.54	231778.03	18.7	4100788.45	9962921.298	4334249.161	€ 18,397,958.91
6	Terrestrial acidification	kg SO2 eq	531669.96	121466.48	303253.51	106949.97	5.4	655918.992	1637568.954	577529.838	€ 2,871,017.78
7	Freshwater eutrophication	kg P eq	8781.6372	2415.9714	5540.6525	825.01324	1.9	4590.34566	10527.23975	1567.525156	€ 16,685.11
8	Marine eutrophication	kg N eq	692.84035	195.51567	433.50576	63.81892	3.11	608.0537337	1348.202914	198.4768412	€ 2,154.73

9	Terrestrial ecotoxicity	kg 1,4-DCB	2.99E+08	1.79E+08	1.12E+08	7577330.2	8.89	1595510970	998437233.5	67362465.48	€ 2,661,310,668.48
10	Freshwater ecotoxicity	kg 1,4-DCB	1073742.3	452076.28	548593.99	73072.031	0.0369	16681.61473	20243.11823	2696.357944	€ 39,621.09
11	Marine ecotoxicity	kg 1,4-DCB	429471.42	189334.36	212201.14	27935.919	0.00756	1431.367762	1604.240618	211.1955476	€ 3,246.80
12	Human carcinogenic toxicity	kg 1,4-DCB	59558.157	39296.352	18788.757	1473.0481	0.214	8409.419328	4020.793998	315.2322934	€ 12,745.45
13	Land use	m2a crop eq	6668341	1335554.2	5286555.7	46231.096	0.0261	34857.96462	137979.1038	1206.631606	€ 174,043.70
14	Mineral resource scarcity (atmospheric emissions)	kg Cu eq	136737.89	40489.77	77030.705	19217.416	4.2	170057.034	323528.961	80713.1472	€ 574,299.14
15	Mineral resource scarcity (emissions to water - soil)	kg Cu eq	136737.89	40489.77	77030.705	19217.416	0.239	9677.05503	18410.3385	4592.962424	€ 32,680.36

LCA results of 2015 and its damage cost

LCA Results of 2015											
No.	**Impact category**	**Unit**	**Total**	**Transportation**	**Landfilling**	**Recycling**	**Price per unit**	**Damage Cost of Transportation**	**Damage Cost of Landfilling**	**Damage Cost of Recycling**	**Total**
1	Global warming	kg CO2 eq	1.08E+08	30909834	58898854	18448178	0.057	1761860.538	3357234.678	1051546.146	€ 6,170,641.36
2	Stratospheric ozone depletion	kg CFC11 eq	39.291655	7.6646697	24.252576	7.3744092	30.4	233.0059589	737.2783104	224.1820397	€ 1,194.47
3	Ozone formation, Human health	kg NOx eq	905545.62	202165.3	489837.22	213543.1	18.7	3780491.11	9159956.014	3993255.97	€ 16,933,703.09
4	Fine particulate matter formation	kg PM2.5 eq	228372.99	14211.153	120959.57	93202.261	79.5	1129786.664	9616285.815	7409579.75	€ 18,155,652.23
5	Ozone formation, Terrestrial ecosystems	kg NOx eq	921049.26	205296.04	498769.53	216983.69	18.7	3839035.948	9326990.211	4057595.003	€ 17,223,621.16
6	Terrestrial acidification	kg SO2 eq	497733.58	113713.31	283896.9	100123.38	5.4	614051.874	1533043.26	540666.252	€ 2,687,761.39
7	Freshwater eutrophication	kg P eq	8221.1071	2261.7605	5186.9939	772.35282	1.9	4297.34495	9855.28841	1467.470358	€ 15,620.10
8	Marine eutrophication	kg N eq	648.6165	183.03595	405.83518	59.745372	3.11	569.2418045	1262.14741	185.8081069	€ 2,017.20
9	Terrestrial ecotoxicity	kg 1,4-DCB	2.80E+08	1.68E+08	1.05E+08	7093670.8	8.89	1493669885	934707134.9	63062733.41	€ 2,491,439,753.71

10	Freshwater ecotoxicity	kg 1,4-DCB	1005205.6	423220.35	513577.35	68407.858	0.0369	15616.83092	18951.00422	2524.24996	€ 37,092.09
11	Marine ecotoxicity	kg 1,4-DCB	402058.35	177249.19	198656.39	26152.775	0.00756	1340.003876	1501.842308	197.714979	€ 3,039.56
12	Human carcinogenic toxicity	kg 1,4-DCB	55756.572	36788.074	17589.475	1379.0237	0.214	7872.647836	3764.14765	295.1110718	€ 11,931.91
13	Land use	m2a crop eq	6242702.2	1250306.1	4949116	43280.175	0.0261	32632.98921	129171.9276	1129.612568	€ 162,934.53
14	Mineral resource scarcity (atmospheric emissions)	kg Cu eq	128009.94	37905.317	72113.852	17990.772	4.2	159202.3314	302878.1784	75561.2424	€ 537,641.75
15	Mineral resource scarcity (emissions to water - soil)	kg Cu eq	128009.94	37905.317	72113.852	17990.772	0.239	9059.370763	17235.21063	4299.794508	€ 30,594.38

LCA results of 2014 and its damage cost

LCA Results of 2014											
No.	**Impact category**	**Unit**	**Total**	**Transportation**	**Landfilling**	**Recycling**	**Price per unit**	**Damage Cost of Transportation**	**Damage Cost of Landfilling**	**Damage Cost of Recycling**	**Total**
1	Global warming	kg CO2 eq	1.02E+08	29153593	55552328	17399986	0.057	1661754.801	3166482.696	991799.202	€ 5,820,036.70
2	Stratospheric ozone depletion	kg CFC11 eq	37.059175	7.2291771	22.874589	6.9554087	30.4	219.7669838	695.3875056	211.4444245	€ 1,126.60
3	Ozone formation, Human health	kg NOx eq	854094.16	190678.64	462005.56	201409.97	18.7	3565690.568	8639503.972	3766366.439	€ 15,971,560.98
4	Fine particulate matter formation	kg PM2.5 eq	215397.25	13403.701	114086.87	87906.678	79.5	1065594.23	9069906.165	6988580.901	€ 17,124,081.30
5	Ozone formation, Terrestrial ecosystems	kg NOx eq	868716.92	193631.5	470430.35	204655.07	18.7	3620909.05	8797047.545	3827049.809	€ 16,245,006.40
6	Terrestrial acidification	kg SO2 eq	469453.26	107252.32	267766.39	94434.547	5.4	579162.528	1445938.506	509946.5538	€ 2,535,047.59
7	Freshwater eutrophication	kg P eq	7753.9988	2133.2513	4892.2783	728.46914	1.9	4053.17747	9295.32877	1384.091366	€ 14,732.60
8	Marine eutrophication	kg N eq	611.76329	172.63618	382.77636	56.350748	3.11	536.8985198	1190.43448	175.2508263	€ 1,902.58
9	Terrestrial ecotoxicity	kg 1,4-DCB	2.64E+08	1.58E+08	99167469	6690621.3	8.89	1408802301	881598799.4	59479623.36	€ 2,349,880,723.27
10	Freshwater ecotoxicity	kg 1,4-DCB	948091.61	399173.74	484396.82	64521.048	0.0369	14729.51101	17874.24266	2380.826671	€ 34,984.58

11	Marine ecotoxicity	kg 1,4-DCB	379214.12	167178.21	187369.09	24666.822	0.00756	1263.867268	1416.51032	186.4811743	€ 2,866.86
12	Human carcinogenic toxicity	kg 1,4-DCB	52588.585	34697.843	16590.073	1300.6701	0.214	7425.338402	3550.275622	278.3434014	€ 11,253.96
13	Land use	m2a crop eq	5888003.2	1179266	4667916.2	40821.074	0.0261	30778.8426	121832.6128	1065.430031	€ 153,676.89
14	Mineral resource scarcity (atmospheric emissions)	kg Cu eq	120736.65	35751.606	68016.474	16968.569	4.2	150156.7452	285669.1908	71267.9898	€ 507,093.93
15	Mineral resource scarcity (emissions to water - soil)	kg Cu eq	120736.65	35751.606	68016.474	16968.569	0.239	8544.633834	16255.93729	4055.487991	€ 28,856.06

LCA results of 2013 and its damage cost

LCA Results of 2013											
No.	**Impact category**	**Unit**	**Total**	**Transportation**	**Landfilling**	**Recycling**	**Price per unit**	**Damage Cost of Transportation**	**Damage Cost of Landfilling**	**Damage Cost of Recycling**	**Total**
1	Global warming	kg CO2 eq	9.60E+07	27397352	52205803	16351795	0.057	1561649.064	2975730.771	932052.315	€ 5,469,432.15
2	Stratospheric ozone depletion	kg CFC11 eq	34.826694	6.7936845	21.496601	6.5364082	30.4	206.5280088	653.4966704	198.7068093	€ 1,058.73
3	Ozone formation, Human health	kg NOx eq	802642.71	179191.97	434173.9	189276.84	18.7	3350889.839	8119051.93	3539476.908	€ 15,009,418.68
4	Fine particulate matter formation	kg PM2.5 eq	202421.51	12596.25	107214.17	82611.095	79.5	1001401.875	8523526.515	6567582.053	€ 16,092,510.44
5	Ozone formation, Terrestrial ecosystems	kg NOx eq	8.16E+05	181966.95	442091.17	192326.45	18.7	3402781.965	8267104.879	3596504.615	€ 15,266,391.46
6	Terrestrial acidification	kg SO2 eq	441172.95	100791.34	251635.89	88745.719	5.4	544273.236	1358833.806	479226.8826	€ 2,382,333.92
7	Freshwater eutrophication	kg P eq	7286.8904	2004.7422	4597.5627	684.58545	1.9	3809.01018	8735.36913	1300.712355	€ 13,845.09
8	Marine eutrophication	kg N eq	574.91008	162.23641	359.71754	52.956125	3.11	504.5552351	1118.721549	164.6935488	€ 1,787.97
9	Terrestrial ecotoxicity	kg 1,4-DCB	2.48E+08	1.49E+08	93193525	6287571.9	8.89	1323934627	828490437.3	55896514.19	€ 2,208,321,578.14
10	Freshwater ecotoxicity	kg 1,4-DCB	890977.65	375127.13	455216.29	60634.238	0.0369	13842.1911	16797.4811	2237.403382	€ 32,877.08

11	Marine ecotoxicity	kg 1,4-DCB	356369.9	157107.23	176081.8	23180.869	0.00756	1187.730659	1331.178408	175.2473696	€ 2,694.16
12	Human carcinogenic toxicity	kg 1,4-DCB	49420.598	32607.611	15590.671	1222.3165	0.214	6978.028754	3336.403594	261.575731	€ 10,576.01
13	Land use	m2a crop eq	5533304.2	1108225.8	4386716.4	38361.973	0.0261	28924.69338	114493.298	1001.247495	€ 144,419.24
14	Mineral resource scarcity (atmospheric emissions)	kg Cu eq	113463.36	33597.894	63919.096	15946.366	4.2	141111.1548	268460.2032	66974.7372	€ 476,546.10
15	Mineral resource scarcity (emissions to water - soil)	kg Cu eq	113463.36	33597.894	63919.096	15946.366	0.239	8029.896666	15276.66394	3811.181474	€ 27,117.74

Appendix 5

Damage Cost of Fine Particulate Matter Formation

No	Impact Category	Year	Damage Cost of Transportation	Damage Cost of Landfilling	Damage Cost of Recycling
1	Fine Particulate Matter Formation (kg PM2.5 eq)	2013	€ 1,001,401.88	€ 8,523,526.52	€ 6,567,582.05
		2014	€ 1,065,594.23	€ 9,069,906.17	€ 6,988,580.90
		2015	€ 1,129,786.66	€ 9,616,285.82	€ 7,409,579.75
		2016	€ 1,206,817.63	€ 10,271,941.40	€ 7,914,778.40
		2017	€ 1,283,848.52	€ 10,927,597.77	€ 8,419,976.97
		2018	€ 1,386,556.40	€ 11,801,805.21	€ 9,093,574.91

Damage Cost of Ozone Formation (Human Health)

No	Impact Category	Year	Damage Cost of Transportation	Damage Cost of Landfilling	Damage Cost of Recycling
2	Ozone Formation-Human Health (kg NOx eq)	2013	€ 3,350,889.84	€ 8,119,051.93	€ 3,539,476.91
		2014	€ 3,565,690.57	€ 8,639,503.97	€ 3,766,366.44
		2015	€ 3,780,491.11	€ 9,159,956.01	€ 3,993,255.97
		2016	€ 4,038,251.91	€ 9,784,498.43	€ 4,265,523.30
		2017	€ 4,296,012.71	€ 10,409,040.84	€ 4,537,790.81
		2018	€ 4,639,693.65	€ 11,241,764.18	€ 4,900,814.09

Damage Cost of Stratospheric Ozone Depletion

No	Impact Category	Year	Damage Cost of Transportation	Damage Cost of Landfilling	Damage Cost of Recycling
3	Stratospheric Ozone Depletion (kg CFC11 eq)	2013	€ 206.53	€ 653.50	€ 198.71
		2014	€ 219.77	€ 695.39	€ 211.44
		2015	€ 233.01	€ 737.28	€ 224.18
		2016	€ 248.89	€ 787.55	€ 239.47
		2017	€ 264.78	€ 837.82	€ 254.75
		2018	€ 285.96	€ 904.84	€ 275.13

Damage Cost of Human Carcinogenic Toxicity

No	Impact Category	Year	Damage Cost of Transportation	Damage Cost of Landfilling	Damage Cost of Recycling
4	Human Carcinogenic Toxicity (kg 1,4-DCB)	2013	€ 6,978.03	€ 3,336.40	€ 261.58
		2014	€ 7,425.34	€ 3,550.28	€ 278.34
		2015	€ 7,872.65	€ 3,764.15	€ 295.11
		2016	€ 8,409.42	€ 4,020.79	€ 315.23
		2017	€ 8,946.19	€ 4,277.44	€ 335.35
		2018	€ 9,661.89	€ 4,619.64	€ 362.18

Damage Cost of Global Warming

No	Impact Category	Year	Damage Cost of Transportation	Damage Cost of Landfilling	Damage Cost of Recycling
5	Global Warming (kg CO2 eq)	2013	€ 1,561,649.06	€ 2,975,730.77	€ 932,052.32
		2014	€ 1,661,754.80	€ 3,166,482.70	€ 991,799.20
		2015	€ 1,761,860.54	€ 3,357,234.68	€ 1,051,546.15
		2016	€ 1,881,987.35	€ 3,586,137.05	€ 1,123,242.51
		2017	€ 2,002,114.23	€ 3,815,039.41	€ 1,194,938.82
		2018	€ 2,162,283.37	€ 4,120,242.55	€ 1,290,533.92

Damage Cost of Freshwater Ecotoxicity

No	Impact Category	Year	Damage Cost of Transportation	Damage Cost of Landfilling	Damage Cost of Recycling
6	Freshwater Ecotoxicity (kg 1,4-DCB)	2013	€ 13,842.19	€ 16,797.48	€ 2,237.40
		2014	€ 14,729.51	€ 17,874.24	€ 2,380.83
		2015	€ 15,616.83	€ 18,951.00	€ 2,524.25
		2016	€ 16,681.61	€ 20,243.12	€ 2,696.36
		2017	€ 17,746.40	€ 21,535.23	€ 2,868.47
		2018	€ 19,166.11	€ 23,258.05	€ 3,097.94

Damage Cost of Freshwater Eutrophication

No	Impact Category	Year	Damage Cost of Transportation	Damage Cost of Landfilling	Damage Cost of Recycling
7	Freshwater Eutrophication (kg P eq)	2013	€ 3,809.01	€ 8,735.37	€ 1,300.71
		2014	€ 4,053.18	€ 9,295.33	€ 1,384.09
		2015	€ 4,297.34	€ 9,855.29	€ 1,467.47
		2016	€ 4,590.35	€ 10,527.24	€ 1,567.53
		2017	€ 4,883.35	€ 11,199.19	€ 1,667.58
		2018	€ 5,274.01	€ 12,095.13	€ 1,800.99

Damage Cost of Ozone Formation, Terrestrial Ecosystems

No	Impact Category	Year	Damage Cost of Transportation	Damage Cost of Landfilling	Damage Cost of Recycling
8	Ozone formation, Terrestrial Ecosystems (kg NOx eq)	2013	€ 3,402,781.97	€ 8,267,104.88	€ 3,596,504.62
		2014	€ 3,620,909.05	€ 8,797,047.55	€ 3,827,049.81
		2015	€ 3,839,035.95	€ 9,326,990.21	€ 4,057,595.00
		2016	€ 4,100,788.45	€ 9,962,921.30	€ 4,334,249.16
		2017	€ 4,362,540.95	€ 10,598,852.39	€ 4,610,903.51
		2018	€ 4,711,544.29	€ 11,446,760.69	€ 4,979,775.78

Damage Cost of Terrestrial Ecotoxicity

No	Impact category	Year	Damage Cost of Transportation	Damage Cost of Landfilling	Damage Cost of Recycling
9	Terrestrial Ecotoxicity (kg 1,4-DCB)	2013	€ 1,323,934,626.70	€ 828,490,437.25	€ 55,896,514.19
		2014	€ 1,408,802,300.50	€ 881,598,799.41	€ 59,479,623.36
		2015	€ 1,493,669,885.40	€ 934,707,134.90	€ 63,062,733.41
		2016	€ 1,595,510,969.50	€ 998,437,233.50	€ 67,362,465.48
		2017	€ 1,697,352,142.50	€ 1,062,167,243.20	€ 71,662,197.54
		2018	€ 1,833,140,313.90	€ 1,147,140,619.10	€ 77,395,172.74

Damage Cost of Terrestrial Acidification

No	Impact Category	Year	Damage Cost of Transportation	Damage Cost of Landfilling	Damage Cost of Recycling
10	Terrestrial Acidification (kg SO2 eq)	2013	€ 544,273.24	€ 1,358,833.81	€ 479,226.88
		2014	€ 579,162.53	€ 1,445,938.51	€ 509,946.55
		2015	€ 614,051.87	€ 1,533,043.26	€ 540,666.25
		2016	€ 655,918.99	€ 1,637,568.95	€ 577,529.84
		2017	€ 697,786.16	€ 1,742,094.59	€ 614,393.42
		2018	€ 753,609.10	€ 1,881,462.17	€ 663,544.93

Damage Cost of Land Use

No	Impact Category	Year	Damage Cost of Transportation	Damage Cost of Landfilling	Damage Cost of Recycling
11	Land Use (m2a crop eq)	2013	€ 28,924.69	€ 114,493.30	€ 1,001.25
		2014	€ 30,778.84	€ 121,832.61	€ 1,065.43
		2015	€ 32,632.99	€ 129,171.93	€ 1,129.61
		2016	€ 34,857.96	€ 137,979.10	€ 1,206.63
		2017	€ 37,082.94	€ 146,786.28	€ 1,283.65
		2018	€ 40,049.58	€ 158,529.18	€ 1,386.34

Damage Cost of Marine Ecotoxicity

No	Impact Category	Year	Damage Cost of Transportation	Damage Cost of Landfilling	Damage Cost of Recycling
12	Marine Ecotoxicity (kg 1,4-DCB)	2013	€ 1,187.73	€ 1,331.18	€ 175.25
		2014	€ 1,263.87	€ 1,416.51	€ 186.48
		2015	€ 1,340.00	€ 1,501.84	€ 197.71
		2016	€ 1,431.37	€ 1,604.24	€ 211.20
		2017	€ 1,522.73	€ 1,706.64	€ 224.68
		2018	€ 1,644.55	€ 1,843.17	€ 242.65

Damage Cost of Marine Eutrophication

No	Impact Category	Year	Damage Cost of Transportation	Damage Cost of Landfilling	Damage Cost of Recycling
13	Marine Eutrophication (kg N eq)	2013	€ 504.56	€ 1,118.72	€ 164.69
		2014	€ 536.90	€ 1,190.43	€ 175.25
		2015	€ 569.24	€ 1,262.15	€ 185.81
		2016	€ 608.05	€ 1,348.20	€ 198.48
		2017	€ 646.87	€ 1,434.26	€ 211.15
		2018	€ 698.61	€ 1,549.00	€ 228.04

Damage Cost of Mineral Resource Scarcity (Atmospheric Emissions)

No	Impact Category	Year	Damage Cost of Transportation	Damage Cost of Landfilling	Damage Cost of Recycling
14	Mineral Resource Scarcity - Atmospheric Emissions (kg Cu eq)	2013	€ 141,111.15	€ 268,460.20	€ 66,974.74
		2014	€ 150,156.75	€ 285,669.19	€ 71,267.99
		2015	€ 159,202.33	€ 302,878.18	€ 75,561.24
		2016	€ 170,057.03	€ 323,528.96	€ 80,713.15
		2017	€ 180,911.74	€ 344,179.75	€ 85,865.05
		2018	€ 195,384.68	€ 371,714.13	€ 92,734.25

Damage Cost of Mineral Resource Scarcity (Emissions to Water to Soil)

No	Impact category	Year	Damage Cost of Transportation	Damage Cost of Landfilling	Damage Cost of Recycling
15	Mineral Resource Scarcity - Emissions to Water to Soil (kg Cu eq)	2013	€ 8,029.90	€ 15,276.66	€ 3,811.18
		2014	€ 8,544.63	€ 16,255.94	€ 4,055.49
		2015	€ 9,059.37	€ 17,235.21	€ 4,299.79
		2016	€ 9,677.06	€ 18,410.34	€ 4,592.96
		2017	€ 10,294.74	€ 19,585.47	€ 4,886.13
		2018	€ 11,118.32	€ 21,152.30	€ 5,277.02

Appendix 6

Descriptive Analysis of 2017 Damage Cost

Descriptive Analysis of 2017 Damage Cost			
Name	**Damage Cost of Transportation**	**Damage Cost of Landfilling**	**Damage Cost of Recycling**
Mean	114017116.3	73347427.35	6075853.192
Standard Error	113096027.9	70638563.68	4728984.303
Median	37082.94264	146786.2799	4886.130101
Mode	#N/A	#N/A	#N/A
Standard Deviation	438019032.5	273581980.7	18315277.45
Sample Variance	1.91861E+17	7.48471E+16	3.35449E+14
Kurtosis	14.99956317	14.99102819	14.31413173
Skewness	3.872904132	3.871361109	3.753997177
Range	1697351878	1062166405	71661986.4
Minimum	264.7794978	837.816248	211.1455755
Maximum	1697352143	1062167243	71662197.54
Sum	1710256745	1100211410	91137797.88
Count	15	15	15

Descriptive Analysis of 2016 Damage Cost

Descriptive Analysis of 2016 Damage Cost			
Name	**Damage Cost of Transportation**	**Damage Cost of Landfilling**	**Damage Cost of Recycling**
Mean	107176086.4	68946583.34	5711301.979
Standard Error	106310263.2	66400251.52	4445245.231
Median	34857.96462	137979.1038	4592.962424
Mode	#N/A	#N/A	#N/A
Standard Deviation	411737879	257167068.3	17216360.75
Sample Variance	1.69528E+17	6.61349E+16	2.96403E+14
Kurtosis	14.99956317	14.99102819	14.31413173
Skewness	3.872904132	3.871361109	3.753997177
Range	1595510721	998436446	67362267
Minimum	248.8927283	787.5472944	198.4768412
Maximum	1595510970	998437233.5	67362465.48
Sum	1607641296	1034198750	85669529.68
Count	15	15	15

Descriptive Analysis of 2015 Damage Cost

Descriptive Analysis of 2015 Damage Cost			
Name	**Damage Cost of Transportation**	**Damage Cost of Landfilling**	**Damage Cost of Recycling**
Mean	100335062.4	64545733.46	5346750.781
Standard Error	99524504.53	62161933.43	4161506.158
Median	32632.98921	129171.9276	4299.794508
Mode	#N/A	#N/A	#N/A
Standard Deviation	385456748.6	240752133	16117444.05
Sample Variance	1.48577E+17	5.79616E+16	2.59772E+14
Kurtosis	14.99956317	14.99102819	14.31413172
Skewness	3.872904132	3.871361109	3.753997176
Range	1493669652	934706397.6	63062547.6
Minimum	233.0059589	737.2783104	185.8081069
Maximum	1493669885	934707134.9	63062733.41
Sum	1505025935	968186001.9	80201261.72
Count	15	15	15

Descriptive Analysis of 2014 Damage Cost

Descriptive Analysis of 2014 Damage Cost			
Name	**Damage Cost of Transportation**	**Damage Cost of Landfilling**	**Damage Cost of Recycling**
Mean	94634208.03	60878363.88	5042958.107
Standard Error	93869704.61	58630007.03	3925056.932
Median	30778.8426	121832.6128	4055.487991
Mode	#N/A	#N/A	#N/A
Standard Deviation	363555802.7	227073040.8	15201680.13
Sample Variance	1.32173E+17	5.15622E+16	2.31091E+14
Kurtosis	14.99956317	14.99102819	14.31413172
Skewness	3.872904132	3.871361109	3.753997175
Range	1408802081	881598104	59479448.11
Minimum	219.7669838	695.3875056	175.2508263
Maximum	1408802301	881598799.4	59479623.36
Sum	1419513120	913175458.2	75644371.61
Count	15	15	15

Descriptive Analysis of 2013 Damage Cost

Descriptive Analysis of 2013 Damage Cost			
Name	**Damage Cost of Transportation**	**Damage Cost of Landfilling**	**Damage Cost of Recycling**
Mean	88933347.76	57210992.53	4739165.498
Standard Error	88214898.78	55098078.84	3688607.763
Median	28924.69338	114493.298	3811.181474
Mode	#N/A	#N/A	#N/A
Standard Deviation	341654833.8	213393941.8	14285916.44
Sample Variance	1.16728E+17	4.5537E+16	2.04087E+14
Kurtosis	14.99956317	14.99102819	14.31413174
Skewness	3.872904132	3.871361109	3.753997178
Range	1323934420	828489783.8	55896349.5
Minimum	206.5280088	653.4966704	164.6935488
Maximum	1323934627	828490437.3	55896514.19
Sum	1334000216	858164888	71087482.47
Count	15	15	15

Appendix 7

Natural logarithmic calculation for transportation method in 2018

2018			
No.	**Impact category**	**LCA Results of Transportation**	**Damage Cost of Transportation**
1	Global warming	17.45137935	14.58667534
2	Stratospheric ozone depletion	2.241415834	5.655858442
3	Ozone formation, Human health	12.42163537	15.3501589
4	Fine particulate matter formation	9.766576799	14.14233382
5	Ozone formation, Terrestrial ecosystems	12.43700276	15.36552629
6	Terrestrial acidification	11.84623012	13.53262907
7	Freshwater eutrophication	7.928693156	8.570547042
8	Marine eutrophication	5.414476996	6.549099722
9	Terrestrial ecotoxicity	19.1443693	21.32929635
10	Freshwater ecotoxicity	13.16044265	9.860898923
11	Marine ecotoxicity	12.29010626	7.405222167
12	Human carcinogenic toxicity	10.71772341	9.175944146
13	Land use	14.24369332	10.59787336
14	Mineral resource scarcity (atmospheric emissions)	10.74764107	12.18272559
15	Mineral resource scarcity (emissions to water - soil)	10.74764107	9.31634934

Natural logarithmic calculation for landfilling method in 2018

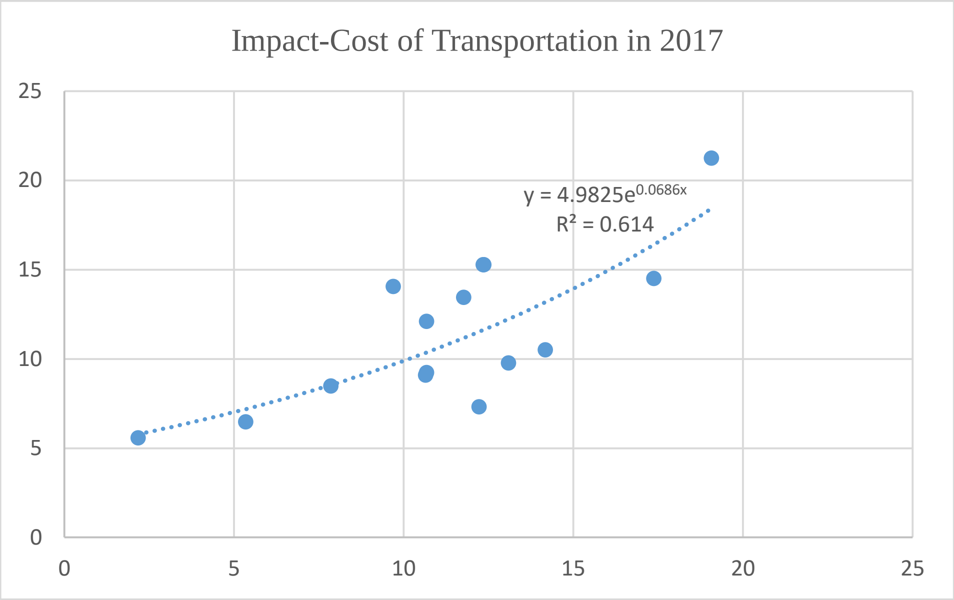
2018			
No.	**Impact category**	**LCA Results of Landfilling**	**Damage Cost of Landfilling**
1	Global warming	18.0961266	15.23142259
2	Stratospheric ozone depletion	3.393317248	6.807759857
3	Ozone formation, Human health	13.30662282	16.23514635
4	Fine particulate matter formation	11.90800604	16.28376306
5	Ozone formation, Terrestrial ecosystems	13.32469381	16.25321734
6	Terrestrial acidification	12.76116083	14.44755978
7	Freshwater eutrophication	8.758704003	9.400557889
8	Marine eutrophication	6.210741533	7.345364259
9	Terrestrial ecotoxicity	18.67561122	20.86053826
10	Freshwater ecotoxicity	13.35395035	10.05440662
11	Marine ecotoxicity	12.40412634	7.519242254
12	Human carcinogenic toxicity	9.979850406	8.438071142
13	Land use	15.61951393	11.97369397
14	Mineral resource scarcity (atmospheric emissions)	11.39079584	12.82588037
15	Mineral resource scarcity (emissions to water - soil)	11.39079584	9.959504115

Natural logarithmic calculation for recycling method in 2018

2018			
No.	**Impact category**	**LCA Results of Recycling**	**Damage Cost of Recycling**
1	Global warming	16.93527059	14.07056658
2	Stratospheric ozone depletion	2.202810204	5.617252813
3	Ozone formation, Human health	12.47638837	15.40491189
4	Fine particulate matter formation	11.64732165	16.02307867
5	Ozone formation, Terrestrial ecosystems	12.4923719	15.42089542
6	Terrestrial acidification	11.71895289	13.40535184
7	Freshwater eutrophication	6.854235877	7.496089763
8	Marine eutrophication	4.294886136	5.429508862
9	Terrestrial ecotoxicity	15.97950792	18.16443497
10	Freshwater ecotoxicity	11.3380374	8.038493671
11	Marine ecotoxicity	10.37650501	5.491620922
12	Human carcinogenic toxicity	7.43392549	5.892146226
13	Land use	10.88024436	7.234424397
14	Mineral resource scarcity (atmospheric emissions)	10.00240866	11.43749318
15	Mineral resource scarcity (emissions to water - soil)	10.00240866	8.571116933

Natural logarithmic calculation for transportation method in 2017

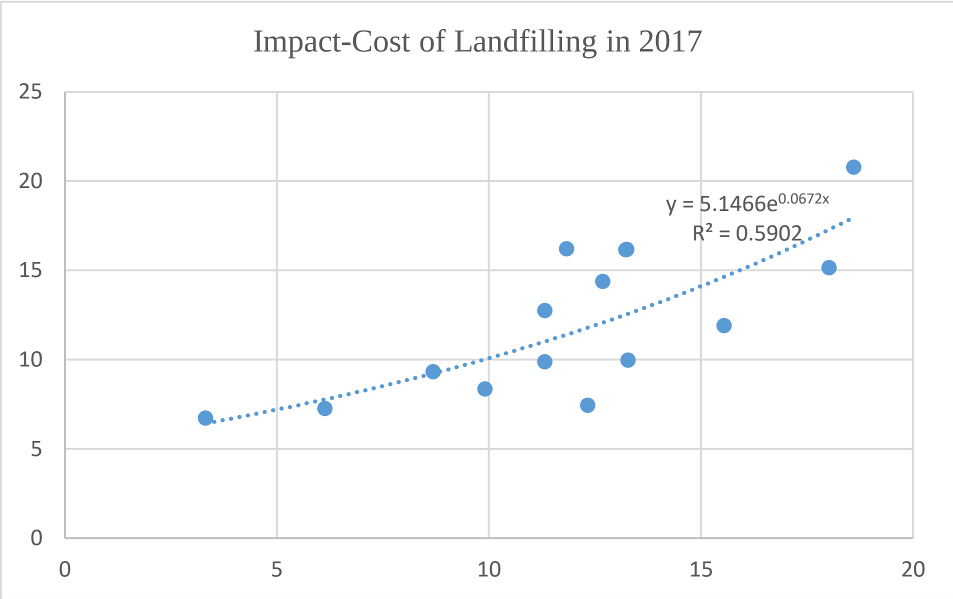
2017			
No.	**Impact category**	**LCA Results of Transportation**	**Damage Cost of Transportation**
1	Global warming	17.3744183	14.50971429
2	Stratospheric ozone depletion	2.164454787	5.578897396
3	Ozone formation, Human health	12.34467435	15.27319787
4	Fine particulate matter formation	9.68961576	14.06537278
5	Ozone formation, Terrestrial ecosystems	12.36004171	15.28856523
6	Terrestrial acidification	11.76926903	13.45566798
7	Freshwater eutrophication	7.851732109	8.493585995
8	Marine eutrophication	5.337515964	6.47213869
9	Terrestrial ecotoxicity	19.06740826	21.25233531
10	Freshwater ecotoxicity	13.0834816	9.783937876
11	Marine ecotoxicity	12.21314523	7.328261137
12	Human carcinogenic toxicity	10.64076238	9.098983114
13	Land use	14.16673234	10.52091238
14	Mineral resource scarcity (atmospheric emissions)	10.67068005	12.10576457
15	Mineral resource scarcity (emissions to water - soil)	10.67068005	9.239388319



The relationship between impact and cost for transportation method in 2017

Natural logarithmic calculation for landfilling method in 2017

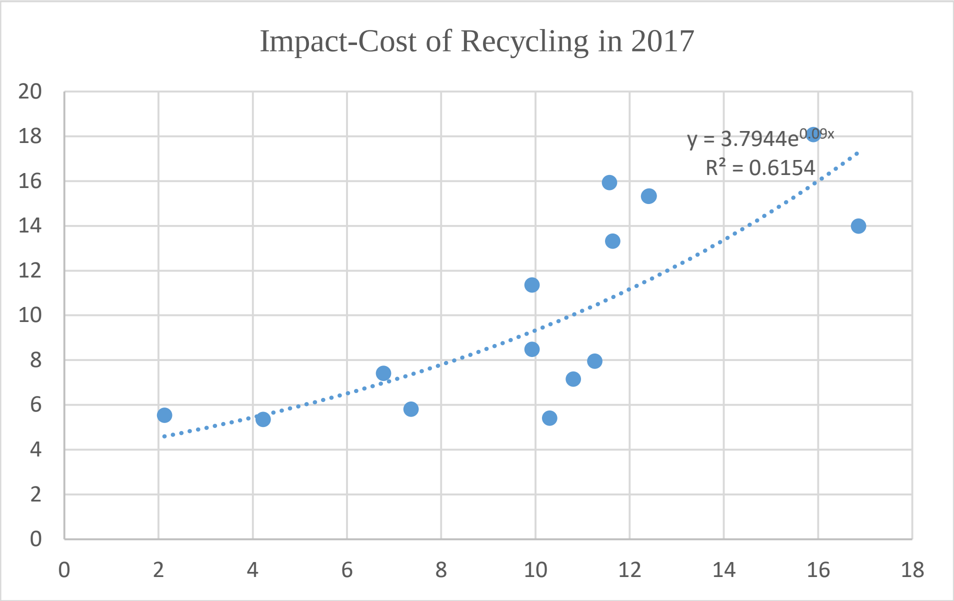
2017			
No.	**Impact category**	**LCA Results of Landfilling**	**Damage Cost of Landfilling**
1	Global warming	18.01916556	15.15446155
2	Stratospheric ozone depletion	3.316356194	6.730798802
3	Ozone formation, Human health	13.22966177	16.1581853
4	Fine particulate matter formation	11.83104503	16.20680205
5	Ozone formation, Terrestrial ecosystems	13.24773276	16.17625629
6	Terrestrial acidification	12.68419978	14.37059874
7	Freshwater eutrophication	8.681742961	9.323596848
8	Marine eutrophication	6.133780486	7.268403212
9	Terrestrial ecotoxicity	18.59865018	20.78357723
10	Freshwater ecotoxicity	13.27698931	9.977445582
11	Marine ecotoxicity	12.32716527	7.442281176
12	Human carcinogenic toxicity	9.902889324	8.36111006
13	Land use	15.54255289	11.89673293
14	Mineral resource scarcity (atmospheric emissions)	11.3138348	12.74891932
15	Mineral resource scarcity (emissions to water - soil)	11.3138348	9.88254307



The relationship between impact and cost for landfilling method in 2017

Natural logarithmic calculation for recycling method in 2017

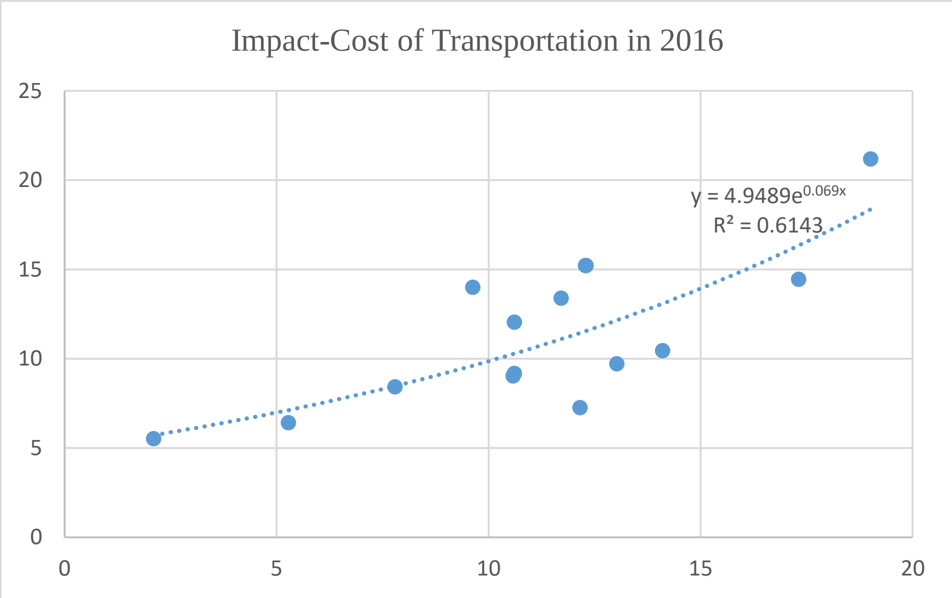
2017			
No.	**Impact category**	**LCA Results of Recycling**	**Damage Cost of Recycling**
1	Global warming	16.85830956	13.99360555
2	Stratospheric ozone depletion	2.125849167	5.540291776
3	Ozone formation, Human health	12.39942732	15.32795085
4	Fine particulate matter formation	11.57036063	15.94611765
5	Ozone formation, Terrestrial ecosystems	12.41541086	15.34393438
6	Terrestrial acidification	11.6419918	13.32839076
7	Freshwater eutrophication	6.777274839	7.419128725
8	Marine eutrophication	4.217925101	5.352547827
9	Terrestrial ecotoxicity	15.90254689	18.08747394
10	Freshwater ecotoxicity	11.26107636	7.961532633
11	Marine ecotoxicity	10.29954397	5.414659882
12	Human carcinogenic toxicity	7.356964444	5.81518518
13	Land use	10.80328333	7.157463363
14	Mineral resource scarcity (atmospheric emissions)	9.925447606	11.36053213
15	Mineral resource scarcity (emissions to water - soil)	9.925447606	8.494155879



The relationship between impact and cost for recycling method in 2017

Natural logarithmic calculation for transportation method in 2016

2016			
No.	**Impact category**	**LCA Results of Transportation**	**Damage Cost of Transportation**
1	Global warming	17.31254289	14.44783888
2	Stratospheric ozone depletion	2.102579385	5.517021994
3	Ozone formation, Human health	12.28279894	15.21132246
4	Fine particulate matter formation	9.627740375	14.0034974
5	Ozone formation, Terrestrial ecosystems	12.29816629	15.22668982
6	Terrestrial acidification	11.70739362	13.39379257
7	Freshwater eutrophication	7.789856721	8.431710607
8	Marine eutrophication	5.27564053	6.410263256
9	Terrestrial ecotoxicity	19.00553283	21.19045988
10	Freshwater ecotoxicity	13.02160621	9.722062478
11	Marine ecotoxicity	12.15126983	7.266385743
12	Human carcinogenic toxicity	10.57888697	9.037107705
13	Land use	14.10485689	10.45903693
14	Mineral resource scarcity (atmospheric emissions)	10.60880463	12.04388915
15	Mineral resource scarcity (emissions to water - soil)	10.60880463	9.177512902

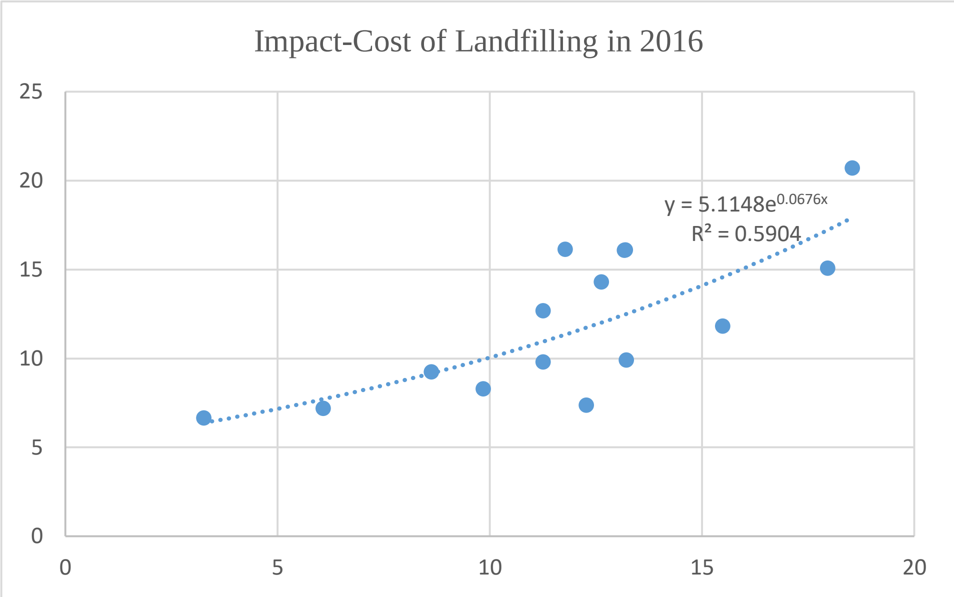


The relationship between impact and cost for transportation method in 2016

The relationship between impact and cost for transportation method in 2016

Natural logarithmic calculation for landfilling method in 2016

2016			
No.	**Impact category**	**LCA Results of Landfilling**	**Damage Cost of Landfilling**
1	Global warming	17.95729016	15.09258615
2	Stratospheric ozone depletion	3.254480817	6.668923425
3	Ozone formation, Human health	13.16778637	16.0963099
4	Fine particulate matter formation	11.76916958	16.1449266
5	Ozone formation, Terrestrial ecosystems	13.18585737	16.11438089
6	Terrestrial acidification	12.6223244	14.30872335
7	Freshwater eutrophication	8.619867553	9.261721439
8	Marine eutrophication	6.071905083	7.20652781
9	Terrestrial ecotoxicity	18.5367748	20.72170185
10	Freshwater ecotoxicity	13.2151139	9.915570174
11	Marine ecotoxicity	12.26528988	7.380405789
12	Human carcinogenic toxicity	9.841013938	8.299234674
13	Land use	15.4806775	11.83485753
14	Mineral resource scarcity (atmospheric emissions)	11.25195939	12.68704391
15	Mineral resource scarcity (emissions to water - soil)	11.25195939	9.820667661

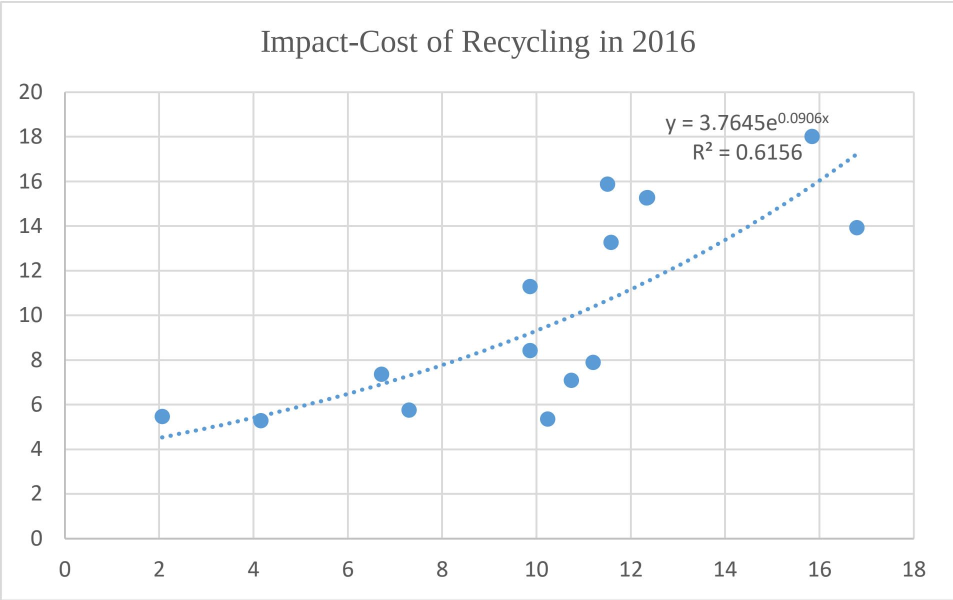


The relationship between impact and cost for landfilling method in 2016

The relationship between impact and cost for landfilling method in 2016

Natural logarithmic calculation for recycling method in 2016

2016			
No.	**Impact category**	**LCA Results of Recycling**	**Damage Cost of Recycling**
1	Global warming	16.79643417	13.93173016
2	Stratospheric ozone depletion	2.063973768	5.478416376
3	Ozone formation, Human health	12.3375519	15.26607543
4	Fine particulate matter formation	11.50848523	15.88424225
5	Ozone formation, Terrestrial ecosystems	12.35353543	15.28205895
6	Terrestrial acidification	11.58011643	13.26651539
7	Freshwater eutrophication	6.715399435	7.357253321
8	Marine eutrophication	4.156049698	5.290672424
9	Terrestrial ecotoxicity	15.84067148	18.02559853
10	Freshwater ecotoxicity	11.19920096	7.899657232
11	Marine ecotoxicity	10.23766856	5.35278447
12	Human carcinogenic toxicity	7.29508907	5.753309806
13	Land use	10.74140792	7.09558796
14	Mineral resource scarcity (atmospheric emissions)	9.86357223	11.29865676
15	Mineral resource scarcity (emissions to water - soil)	9.86357223	8.432280503

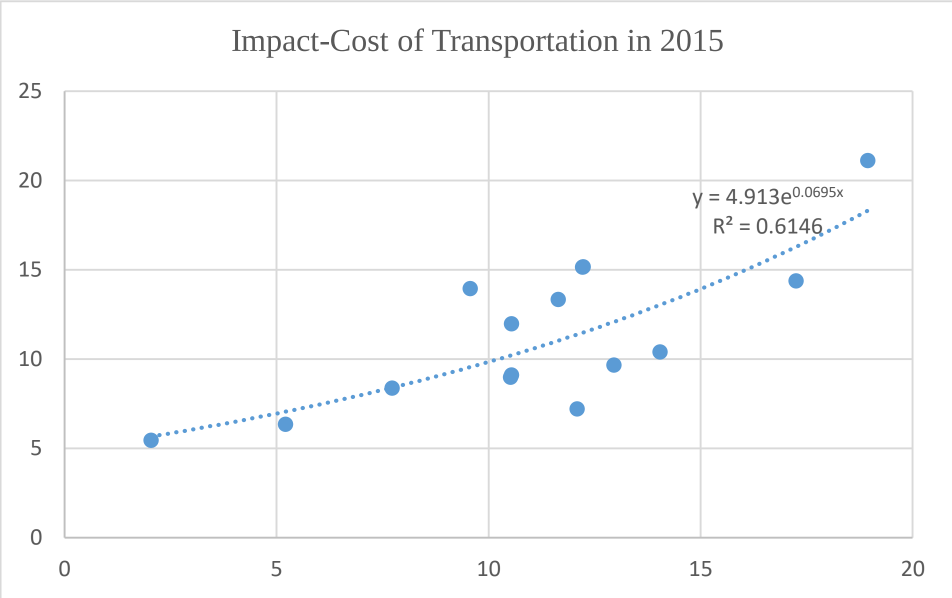


The relationship between impact and cost for recycling method in 2016

The relationship between impact and cost for recycling method in 2016

Natural logarithmic calculation for transportation method in 2015

2015			
No.	**Impact category**	**LCA Results of Transportation**	**Damage Cost of Transportation**
1	Global warming	17.24658494	14.38188093
2	Stratospheric ozone depletion	2.036621419	5.451064028
3	Ozone formation, Human health	12.21684096	15.14536448
4	Fine particulate matter formation	9.561782358	13.93753938
5	Ozone formation, Terrestrial ecosystems	12.23220831	15.16073184
6	Terrestrial acidification	11.64143574	13.32783469
7	Freshwater eutrophication	7.723898771	8.365752657
8	Marine eutrophication	5.209682582	6.344305308
9	Terrestrial ecotoxicity	18.93957489	21.12450194
10	Freshwater ecotoxicity	12.95564824	9.656104516
11	Marine ecotoxicity	12.08531187	7.200427786
12	Human carcinogenic toxicity	10.512929	8.971149732
13	Land use	14.03889896	10.39307899
14	Mineral resource scarcity (atmospheric emissions)	10.54284667	11.9779312
15	Mineral resource scarcity (emissions to water - soil)	10.54284667	9.111554944

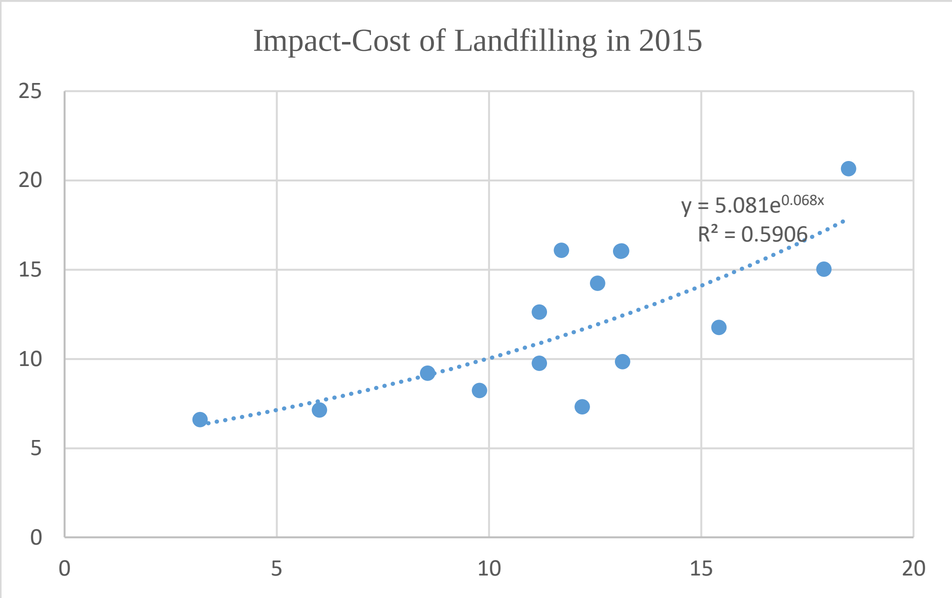


The relationship between impact and cost for transportation method in 2015

The relationship between impact and cost for transportation method in 2015

Natural logarithmic calculation for landfilling method in 2015

2015			
No.	**Impact category**	**LCA Results of Landfilling**	**Damage Cost of Landfilling**
1	Global warming	17.89133219	15.02662818
2	Stratospheric ozone depletion	3.188522839	6.602965447
3	Ozone formation, Human health	13.10182841	16.03035193
4	Fine particulate matter formation	11.70321164	16.07896866
5	Ozone formation, Terrestrial ecosystems	13.1198994	16.04842293
6	Terrestrial acidification	12.55636642	14.24276538
7	Freshwater eutrophication	8.553909598	9.195763485
8	Marine eutrophication	6.005947117	7.140569843
9	Terrestrial ecotoxicity	18.47081676	20.65574381
10	Freshwater ecotoxicity	13.14915593	9.849612202
11	Marine ecotoxicity	12.19933193	7.314447839
12	Human carcinogenic toxicity	9.775055991	8.233276727
13	Land use	15.41471953	11.76889957
14	Mineral resource scarcity (atmospheric emissions)	11.18600143	12.62108595
15	Mineral resource scarcity (emissions to water - soil)	11.18600143	9.7547097

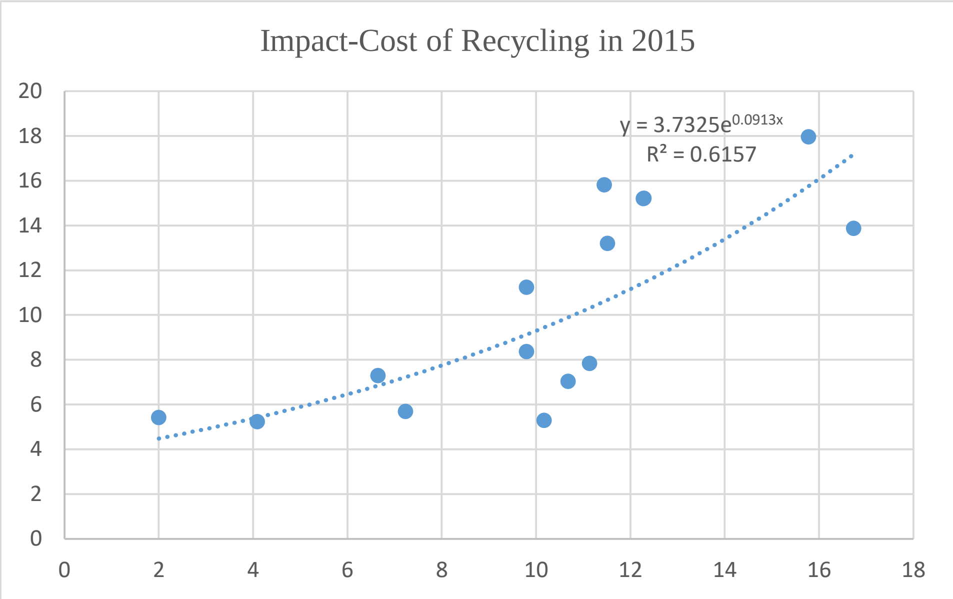


The relationship between impact and cost for landfilling method in 2015

The relationship between impact and cost for landfilling method in 2015

Natural logarithmic calculation for recycling method in 2015

2015			
No.	**Impact category**	**LCA Results of Recycling**	**Damage Cost of Recycling**
1	Global warming	16.73047617	13.86577216
2	Stratospheric ozone depletion	1.998015791	5.412458399
3	Ozone formation, Human health	12.27159396	15.20011749
4	Fine particulate matter formation	11.44252726	15.81828428
5	Ozone formation, Terrestrial ecosystems	12.28757747	15.21610099
6	Terrestrial acidification	11.5141585	13.20055746
7	Freshwater eutrophication	6.649441466	7.291295353
8	Marine eutrophication	4.090091732	5.224714458
9	Terrestrial ecotoxicity	15.77471351	17.95964056
10	Freshwater ecotoxicity	11.13324298	7.833699252
11	Marine ecotoxicity	10.17171058	5.286826494
12	Human carcinogenic toxicity	7.229131064	5.6873518
13	Land use	10.67544996	7.029629992
14	Mineral resource scarcity (atmospheric emissions)	9.797614239	11.23269876
15	Mineral resource scarcity (emissions to water - soil)	9.797614239	8.366322512

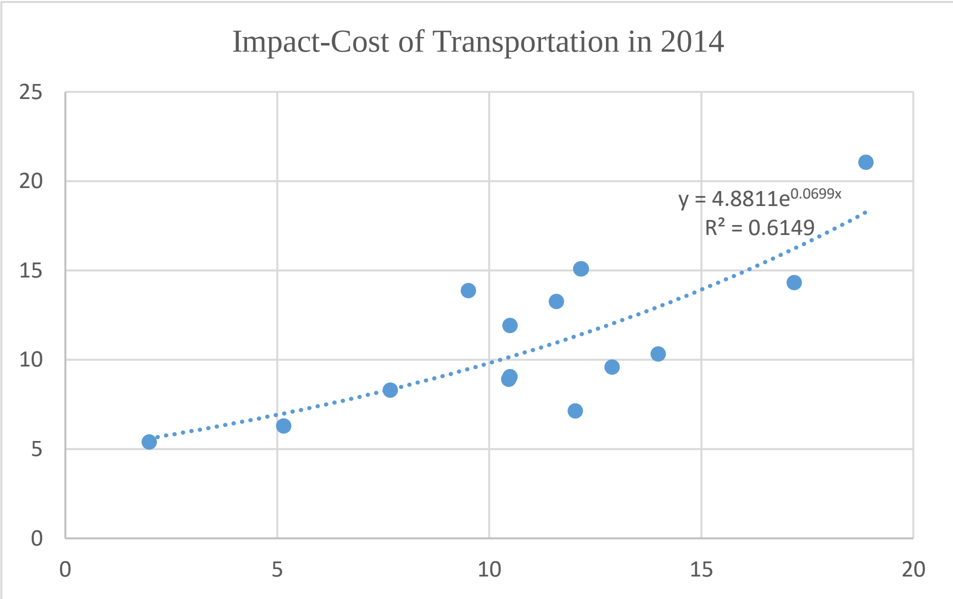


The relationship between impact and cost for recycling method in 2015

The relationship between impact and cost for recycling method in 2015

Natural logarithmic calculation for transportation method in 2014

2014			
No.	**Impact category**	**LCA Results of Transportation**	**Damage Cost of Transportation**
1	Global warming	17.18808872	14.32338471
2	Stratospheric ozone depletion	1.978125212	5.392567821
3	Ozone formation, Human health	12.15834478	15.0868683
4	Fine particulate matter formation	9.503286142	13.87904316
5	Ozone formation, Terrestrial ecosystems	12.17371215	15.10223567
6	Terrestrial acidification	11.58293947	13.26933842
7	Freshwater eutrophication	7.665402527	8.307256413
8	Marine eutrophication	5.151186374	6.2858091
9	Terrestrial ecotoxicity	18.8810787	21.06600575
10	Freshwater ecotoxicity	12.89715204	9.597608312
11	Marine ecotoxicity	12.02681565	7.141931559
12	Human carcinogenic toxicity	10.4544328	8.912653539
13	Land use	13.98040277	10.3345828
14	Mineral resource scarcity (atmospheric emissions)	10.48435047	11.919435
15	Mineral resource scarcity (emissions to water - soil)	10.48435047	9.053058743

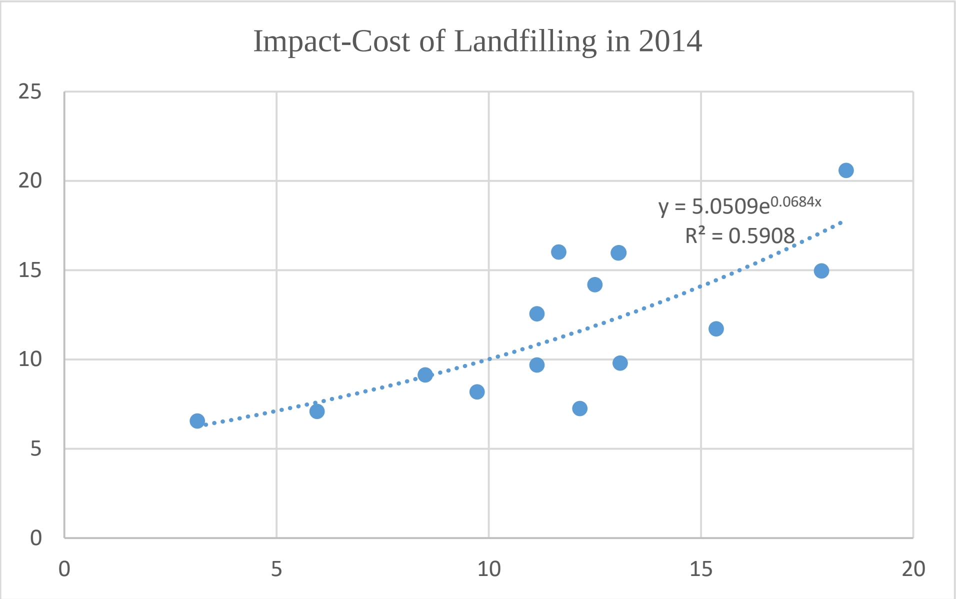


The relationship between impact and cost for transportation method in 2014

The relationship between impact and cost for transportation method in 2014

Natural logarithmic calculation for landfilling method in 2014

2014			
No.	**Impact category**	**LCA Results of Landfilling**	**Damage Cost of Landfilling**
1	Global warming	17.83283598	14.96813197
2	Stratospheric ozone depletion	3.130026644	6.544469252
3	Ozone formation, Human health	13.0433322	15.97185573
4	Fine particulate matter formation	11.64471545	16.02047248
5	Ozone formation, Terrestrial ecosystems	13.06140319	15.98992672
6	Terrestrial acidification	12.4978702	14.18426915
7	Freshwater eutrophication	8.495413384	9.13726727
8	Marine eutrophication	5.947450902	7.082073628
9	Terrestrial ecotoxicity	18.41232059	20.59724763
10	Freshwater ecotoxicity	13.09065973	9.791115998
11	Marine ecotoxicity	12.14083569	7.255951605
12	Human carcinogenic toxicity	9.716559783	8.174780519
13	Land use	15.35622332	11.71040336
14	Mineral resource scarcity (atmospheric emissions)	11.12750522	12.56258974
15	Mineral resource scarcity (emissions to water - soil)	11.12750522	9.696213492

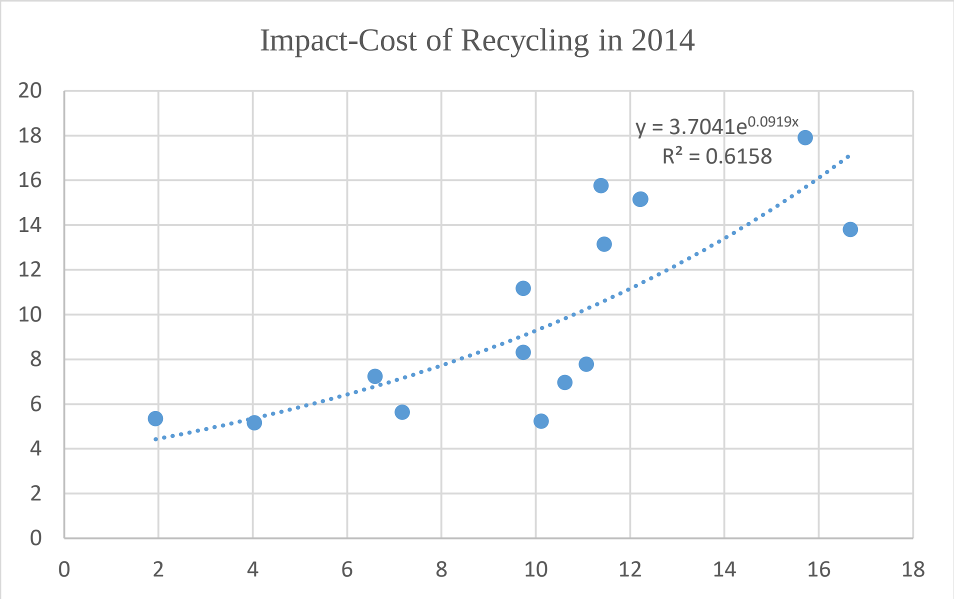


The relationship between impact and cost for landfilling method in 2014

The relationship between impact and cost for landfilling method in 2014

Natural logarithmic calculation for recycling method in 2014

2014			
No.	**Impact category**	**LCA Results of Recycling**	**Damage Cost of Recycling**
1	Global warming	16.67197996	13.80727595
2	Stratospheric ozone depletion	1.939519587	5.353962196
3	Ozone formation, Human health	12.21309776	15.14162129
4	Fine particulate matter formation	11.38403105	15.75978808
5	Ozone formation, Terrestrial ecosystems	12.22908126	15.15760478
6	Terrestrial acidification	11.45566225	13.1420612
7	Freshwater eutrophication	6.590945264	7.23279915
8	Marine eutrophication	4.031595515	5.166218241
9	Terrestrial ecotoxicity	15.7162173	17.90114435
10	Freshwater ecotoxicity	11.07474678	7.775203047
11	Marine ecotoxicity	10.11321438	5.228330292
12	Human carcinogenic toxicity	7.170634872	5.628855608
13	Land use	10.61695375	6.971133782
14	Mineral resource scarcity (atmospheric emissions)	9.739118029	11.17420255
15	Mineral resource scarcity (emissions to water - soil)	9.739118029	8.307826302

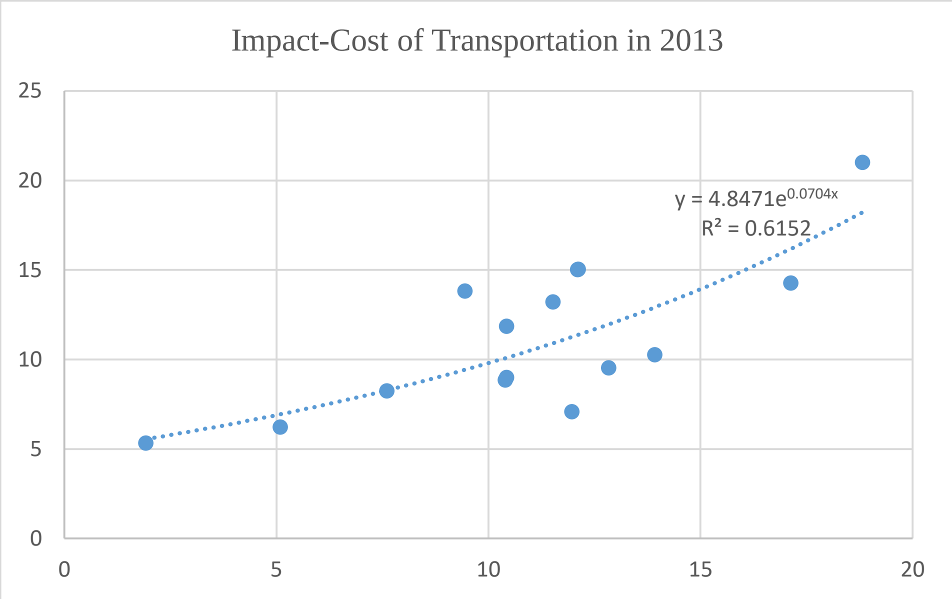


The relationship between impact and cost for recycling method in 2014

The relationship between impact and cost for recycling method in 2014

Natural logarithmic calculation for transportation method in 2013

2013			
No.	**Impact category**	**LCA Results of Transportation**	**Damage Cost of Transportation**
1	Global warming	17.12595692	14.26125291
2	Stratospheric ozone depletion	1.915993431	5.330436039
3	Ozone formation, Human health	12.09621297	15.02473649
4	Fine particulate matter formation	9.44115443	13.81691145
5	Ozone formation, Terrestrial ecosystems	12.11158036	15.04010388
6	Terrestrial acidification	11.52080772	13.20720667
7	Freshwater eutrophication	7.603270753	8.245124639
8	Marine eutrophication	5.089054592	6.223677319
9	Terrestrial ecotoxicity	18.81894687	21.00387392
10	Freshwater ecotoxicity	12.83502026	9.535476533
11	Marine ecotoxicity	11.96468384	7.079799756
12	Human carcinogenic toxicity	10.39230101	8.850521742
13	Land use	13.91827092	10.27245095
14	Mineral resource scarcity (atmospheric emissions)	10.42221867	11.85730319
15	Mineral resource scarcity (emissions to water - soil)	10.42221867	8.990926938

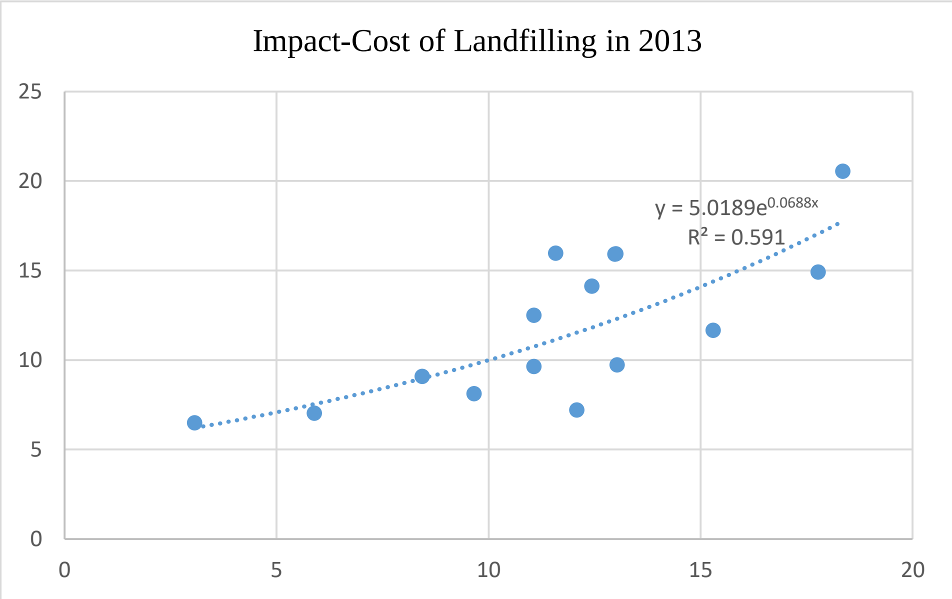


The relationship between impact and cost for transportation method in 2013

The relationship between impact and cost for transportation method in 2013

Natural logarithmic calculation for landfilling method in 2013

2013			
No.	**Impact category**	**LCA Results of Landfilling**	**Damage Cost of Landfilling**
1	Global warming	17.77070422	14.9060002
2	Stratospheric ozone depletion	3.06789483	6.482337438
3	Ozone formation, Human health	12.98120042	15.90972395
4	Fine particulate matter formation	11.5825837	15.95834072
5	Ozone formation, Terrestrial ecosystems	12.99927141	15.92779493
6	Terrestrial acidification	12.43573844	14.12213739
7	Freshwater eutrophication	8.433281594	9.07513548
8	Marine eutrophication	5.885319112	7.019941839
9	Terrestrial ecotoxicity	18.3501888	20.53511585
10	Freshwater ecotoxicity	13.02852795	9.72898422
11	Marine ecotoxicity	12.07870394	7.19381985
12	Human carcinogenic toxicity	9.654428002	8.112648738
13	Land use	15.29409153	11.64827157
14	Mineral resource scarcity (atmospheric emissions)	11.06537344	12.50045796
15	Mineral resource scarcity (emissions to water - soil)	11.06537344	9.634081711

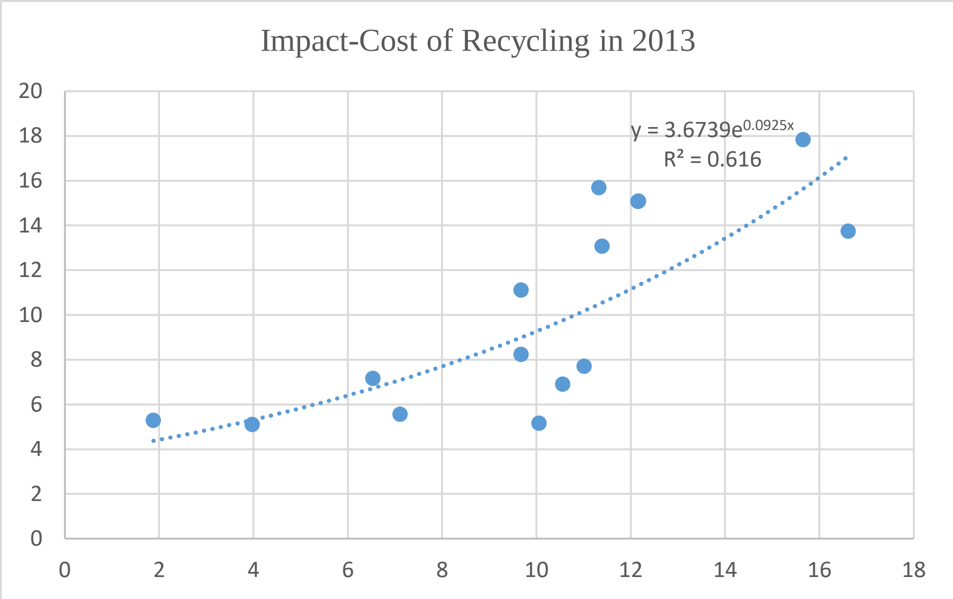


The relationship between impact and cost for landfilling method in 2013

The relationship between impact and cost for landfilling method in 2013

Natural logarithmic calculation for recycling method in 2013

2013			
No.	**Impact category**	**LCA Results of Recycling**	**Damage Cost of Recycling**
1	Global warming	16.60984824	13.74514422
2	Stratospheric ozone depletion	1.87738781	5.291830418
3	Ozone formation, Human health	12.15096598	15.07948951
4	Fine particulate matter formation	11.32189927	15.69765629
5	Ozone formation, Terrestrial ecosystems	12.16694947	15.09547299
6	Terrestrial acidification	11.39353047	13.07992942
7	Freshwater eutrophication	6.528813473	7.170667359
8	Marine eutrophication	3.969463741	5.104086467
9	Terrestrial ecotoxicity	15.65408553	17.83901258
10	Freshwater ecotoxicity	11.012615	7.713071268
11	Marine ecotoxicity	10.05108261	5.166198517
12	Human carcinogenic toxicity	7.108503108	5.566723844
13	Land use	10.55482196	6.909001997
14	Mineral resource scarcity (atmospheric emissions)	9.676986245	11.11207077
15	Mineral resource scarcity (emissions to water - soil)	9.676986245	8.245694518



The relationship between impact and cost for recycling method in 2013

The relationship between impact and cost for recycling method in 2013

Appendix 8

Global Warming Potentials in units (kg CO2-eq/kg) of three cultural perspectives (RIVM Report 2016-0104, p. 29-35)

Name	**Formula**	**Individualist (20 years)**	**Hierarchist (100 years)**	**Egalitarian (1,000 years)**
Carbon dioxide	CO2	1	1	1
Methane	CH4	84	34	4.8
Fossil methane	CH4	85	36	4.9
Nitrous oxide	N2O	264	298	78.8
Chlorofluorocarbons				
CFC-11	CCl3F	6900	5.352	875.4
CFC-12	CCl2F2	10800	11547	2709.4
CFC-13	CClF3	10900	15451	1,2684.1
CFC-113	CCl2FCClF2	6940	6586	1,409.5
CFC-114	CCl2FCClF2	7710	9615	3,492.3
CFC-115	CClF2CF3	5860	8516	8,578.8
Hydrochlorofluoro-carbons				
HCFC-21	CHCl2F	543	179	24.6
HCFC-22	CHClF2	5280	2106	295.9
HCFC-122	CHCl2CF2Cl	218	72	9.9
HCFC-122a	CHFClCFCl2	945	312	43.2
HCFC-123	CHCl2CF3	292	96	13.3
HCFC-123a	CHClFCF2Cl	1350	447	61.9
124	CHClFCF3	1870	635	88.2
HCFC-132c	CH2FCFCl2	1230	409	56.6
HCFC-141b	CH3CCl2F	2550	938	130.9
HCFC-142b	CH3CClF2	5020	2345	332.5
HCFC-225ca	CHCl2CF2CF3	469	155	21.4
HCFC-225cb	CHClFCF2CClF2	1860	633	87.8
(E)-1-Chloro-3,3,3-trifluoroprop-1-ene	trans-CF3CH=CHCl	5	2	0.3
Hydrofluorocarbons				
HFC-23	CHF3	10800	13856	5664.5
HFC-32	CH2F2	2430	817	113.3
HFC-41	CH3F	427	141	19.5
HFC-125	CHF2CF3	6090	3691	546.4
HFC-134	CHF2CHF2	3580	1337	186.4
HFC-134a	CH2FCF3	3710	1549	217.6
HFC-143	CH2FCHF2	1200	397	54.9
HFC-143a	CH3CF3	6940	5508	913.3
HFC-152	CH2FCH2F	60	20	2.8
HFC-152a	CH3CHF2	506	167	23.0

HFC-161	CH3CH2F	13	4	0.6
HFC-227ca	CF3CF2CHF2	5,080	3,077	455.5
HFC-227ea	CF3CHFCF3	5,360	3,860	607.5
HFC-236cb	CH2FCF2CF3	3,480	1,438	202.4
HFC-236ea	CHF2CHFCF3	4,110	1,596	223.5
HFC-236fa	CF3CH2CF3	6,940	8,998	3918.3
HFC-245ca	CH2FCF2CHF2	2,510	863	119.7
HFC-245cb	CF3CF2CH3	6,680	5,298	879.9
HFC-245ea	CHF2CHFCHF2	863	285	39.4
HFC-245eb	CH2FCHFCF3	1,070	352	48.6
HFC-245fa	CHF2CH2CF3	2,920	1,032	143.7
HFC-263fb	CH3CH2CF3	278	92	12.6
HFC-272ca	CH3CF2CH3	530	175	24.1
HFC-329p	CHF2CF2CF2C F3	4,510	2,742	407.1
HFC-365mfc	CH3CF2CH2CF 3	2,660	966	134.7
HFC-43-10mee	CF3CHFCHFCF 2C F3	4,310	1,952	276.6
HFC-1132a	CH2=CF2	0	0	0.0
HFC-1141	CH2=CHF	0	0	0.0
(Z)-HFC-1225ye	CF3CF=CHF(Z)	1	0	0.0
(E)-HFC-1225ye	CF3CF=CHF(E)	0	0	0.0
(Z)-HFC-1234ze	CF3CH=CHF(Z)	1	0	0.0
HFC-1234yf	CF3CF=CH2	1	0	0.1
(E)-HFC-1234ze	trans- CF3CH=CHF	4	1	0.2

(Z)-HFC-1336	CF3CH=CHCF3 (Z)	6	2	0.3
HFC-1243zf	CF3CH=CH2	1	0	0.0
HFC-1345zfc	C2F5CH=CH2	0	0	0.0
3,3,4,4,5,5,6,6,6- Nonafluorohex-1-ene 3,3,4,4,5,5,6,6,7,7,8,8,8	C4F9CH=CH2 C6F13CH=CH2	1 0	0 0	0.0 0.0
-Tridecafluorooct-1-ene 3,3,4,4,5,5,6,6,7,7,8,8,9	C8F17CH=CH2	0	0	0.0
3,3,4,4,5,5,6,6,7,7,8,8,9,9,10,10,10- Heptadecafluorodec-1-ene				
Chlorocarbons and hydrochlorocarbons				
Methyl chloroform	CH3CCl3	578	193	26.8
Carbon tetrachloride	CCl4	3,480	2,019	296.0
Methyl chloride	CH3Cl	45	15	2.0
Methylene chloride	CH2Cl2	33	11	1.5
Chloroform	CHCl3	60	20	2.7
1,2-Dichloroethane	CH2ClCH2Cl	3	1	0.2
Bromocarbons, hyrdobromocarbons and Halons				
Methyl bromide	CH3Br	9	3	0.4
Methylene bromide	CH2Br2	4	1	0.2
Halon-1201	CHBrF2	1,350	454	62.9
Halon-1202	CBr2F2	848	280	38.7
Halon-1211	CBrClF2	4,590	2,070	293.3

Halon-1301	CBrF3	7,800	7,154	1342.2
Halon-2301	CH2BrCF3	635	210	29.1
Halon-2311/Halothane	CHBrClCF3	151	50	6.9
Halon-2401	CHFBrCF3	674	223	30.7
Halon-2402	CBrF2CBrF2	3,440	1,734	248.0
Fully Fluorinated Species				
Nitrogen trifluoride	NF3	12,800	17,885	12,816.7
Sulphur hexafluoride	SF6	17,500	26,087	34,368.5
(Trifluoromethyl)sulfur pentafluoride	SF5CF3	13,500	19,396	17,724.5
Sulfuryl fluoride	SO2F2	6,840	4,732	731.9
PFC-14	CF4	4,880	7,349	11,009.8
PFC-116	C2F6	8,210	12,340	17,810.2
PFC-c216	c-C3F6	6,850	10,208	13,315.3
PFC-218	C3F8	6,640	9,878	12,611.8
PFC-318	c-C4F8	7,110	10,592	13,921.4
PFC-31-10	C4F10	6,870	10,213	13,018.1
Perfluorocyclopentene	c-C5F8	7	2	0.3
PFC-41-12	n-C5F12	6,350	9,484	12,838.0
PFC-51-14	n-C6F14	5,890	8,780	11,504.8
PFC-61-16	n-C7F16	5,830	8,681	11,301.3
PFC-71-18	C8F18	5,680	8,456	11,042.5
PFC-91-18	C10F18	5,390	7,977	9,686.2
Perfluorodecalin(cis)	Z-C10F18	5,430	8,033	9,759.0
Perfluorodecalin(trans)	E-C10F18	4,720	6,980	8,505.2
PFC-1114	CF2=CF2	0	0	0.0
PFC-1216	CF3CF=CF2	0	0	0.0
Perfluorobuta-1,3-diene	CF2=CFCF=CF2	0	0	0.0
Perfluorobut-1-ene	CF3CF2CF=CF2	0	0	0.0
Perfluorobut-2-ene	CF3CF=CFCF3	6	2	0.3
Halogenated alcohols and ethers				
HFE-125	CHF2OCF3	12,400	13,951	3,657.5
HFE-134 (HG-00)	CHF2OCHF2	11,600	6,512	946.2
HFE-143a	CH3OCF3	1,890	632	87.5
HFE-227ea	CF3CHFOCF3	8,900	7,377	1,261.5
HCFE-235ca2(enflurane)	CHF2OCF2CHFCl	2,120	705	97.6

HCFE-235da2(isoflurane)	CHF2OCHClCF3	1,800	595	82.2
HFE-236ca	CHF2OCF2CHF2	9,710	4,990	715.3
HFE-236ea2(desflurane)	CHF2OCHFCF3	5,550	2,143	300.1
HFE-236fa	CF3CH2OCF3	3,350	1,177	163.8
HFE-245cb2	CF3CF2OCH3	2,360	790	109.5
HFE-245fa1	CHF2CH2OCF3	2,900	997	138.5
HFE-245fa2	CHF2OCH2CF3	2,910	981	135.9
2,2,3,3,3-	CF3CF2CH2OH	69	23	3.1
Pentafluoropropane-1-ol				
HFE-254cb1	CH3OCF2CHF2	1,110	365	50.4
HFE-263fb2	CF3CH2OCH3	5	2	0.2
HFE-263m1	CF3OCH2CH3	108	36	4.9
3,3,3-Trifluoropropan-1-ol	CF3CH2CH2OH	1	0	0.1
HFE-329mcc2	CHF2CF2OCF2CF	6,720	3,598	519.8

HFE-338mmz1	(CF3)2CHOCHF2	5,940	3,081	442.1
HFE-338mcf2	CF3CH2OCF2CF3	3,180	1,118	155.5
Sevoflurane (HFE-347mmz1)	(CF3)2CHOCH2F	795	262	36.1
HFE-347mcc3 (HFE-7000)	CH3OCF2CF2CF3	1,910	641	88.8
HFE-347mcf2	CHF2CH2OCF2CF3	2,990	1,028	142.9
HFE-347pcf2	CHF2CF2OCH2CF3	3,150	1,072	148.7
HFE-347mmy1	(CF3)2CFOCH3	1,330	440	60.8
HFE-356mec3	CH3OCF2CHFCF3	1,410	468	64.8
HFE-356mff2	CF3CH2OCH2CF3	62	20	2.8
HFE-356pcf2	CHF2CH2OCF2CHF2	2,560	867	120.3
HFE-356pcf3	CHF2OCH2CF2CHF2	1,640	540	74.7
HFE-356pcc3	CH3OCF2CF2CHF2	1,510	500	69.2
HFE-356mmz1	(CF3)2CHOCH3	50	17	2.3
HFE-365mcf3	CF3CF2CH2OCH3	3	1	0.2
HFE-365mcf2	CF3CF2OCH2CH3	215	71	9.8
HFE-374pc2	CHF2CF2OCH2CH3	2,260	758	105.0
4,4,4-Trifluorobutan-1-ol	CF3(CH2)2CH2OH	0	0	0.0

2,2,3,3,4,4,5,5-Octafluorocyclopentaonl	(CF2)4CH(OH)	47	16	2.2
HFE-43-10pccc124(H-Galden 1040x,HG-11)	CHF2OCF2OC2F4OCHF2	8,010	3,353	471.7
HFE-449s1 (HFE-7100)	C4F9OCH3	1,530	509	70.4
n-HFE-7100	n-C4F9OCH3	1,760	587	81.2
i-HFE-7100	i-C4F9OCH3	1,480	492	68.1
HFE-569sf2 (HFE-7200)	C4F9OC2H5	209	69	9.5
n-HFE-7200	n-C4F9OC2H5	237	79	10.8
i-HFE-7200	i-C4F9OC2H5	163	54	7.4
HFE-236ca12 (HG-10)	CHF2OCF2OCHF2	11,000	6,260	912.0
HFE-338pcc13 (HG-01)	CHF2OCF2CF2OCHF2	8,430	3,466	486.9
1,1,1,3,3,3-Hexafluoropropane-2-ol	(CF3)2CHOH	668	221	30.5
HG-02	HF2C–(OCF2CF2)2–OCF2H	7,900	3,250	456.4
HG-03	HF2C–(OCF2CF2)3–OCF2H	8,270	3,400	477.7
HG-20	HF2C (OCF2)2–OCF2H	10,900	6,201	904.1

HG-21	HF2C–	11,100	4,628	651.9
	OCF2CF2OCF2OC			
	F2O–CF2H			
HG-30	HF2C–(OCF2)3–	15,100	8,575	1,250.2
	OCF2H			
1-Ethoxy-1,1,2,2,3,3,3-	CF3CF2CF2OCH2	223	74	10.1
heptafluoropropane	CH3			
Fluoroxene	CF3CH2OCH=CH	0	0	0.0
	2			
1,1,2,2-Tetrafluoro-1-	CH2FOCF2CF2H	3,080	1,051	145.9
(fluoromethoxy)ethane				
2-Ethoxy-3,3,4,4,5-	C12H5F19O2	204	68	9.3
pentafluorotetrahydro-				
2,5-bis[1,2,2,2-				
tetrafluoro-1-				

(trifluoromethyl)e thyl]-				
furan				
Fluoro(methoxy) methane	CH3OCH2F	46	15	2.1
Difluoro(methoxy)methane	CH3OCHF2	528	175	24.1
Fluoro(fluoromet hoxy)-	CH2FOCH2F	479	159	21.9
methane				
Difluoro(fluorom ethoxy)-	CH2FOCHF2	2,260	748	103.3
methane				
Trifluoro(fluorom ethoxy)-	CH2FOCF3	2,730	909	125.8
methane				
HG'-01	CH3OCF2CF2O C	815	269	37.0
	H3			
HG'-02	CH3O(CF2CF2 O)	868	287	39.4
	2CH3			
HG'-03	CH3O(CF2CF2 O)	812	268	37.0
	3CH3			
HFE-329me3	CF3CFHCF2OC F3	7,170	5,241	829.6
3,3,4,4,5,5,6,6,7,7 ,7-	CF3(CF2)4CH2 C	2	1	0.1
Undecafluorohept an-1-ol	H2OH			
3,3,4,4,5,5,6,6,7,7 ,8,8,9	CF3(CF2)6CH2 C	1	0	0.1
,9,9-	H2OH			
Pentadecafluoron onan-1-				
ol				
3,3,4,4,5,5,6,6,7,7 ,8,8,9	CF3(CF2)8CH2 C	1	0	0.0
,9,10,10,11,11,11-	H2OH			
Nonadecafluorou ndecan-				
1-ol				
2-Chloro-1,1,2- trifluoro-	CH3OCF2CHF Cl	449	149	20.4
1-methoxyethane				
PFPMIE(perfluor opoly-	CF3OCF(CF3)C F2	7,500	10,789	9,861.9

methylisopropyl ether)	OCF2OCF3			
HFE-216	CF3OCF=CF2	1	0	0.0
Trifluoromethylformate	HCOOCF3	2,150	712	98.3
Perfluoroethylformate	HCOOCF2CF3	2,130	703	97.1
Perfluoropropylformate	HCOOCF2CF2CF3	1,380	456	63.0

Perfluorobutylformate	HCOOCF2CF2CF2CF3	1,440	475	65.6
2,2,2-Trifluoroethylformate	HCOOCH2CF3	123	41	5.6
3,3,3-Trifluoropropylformate	HCOOCH2CH2CF3	64	21	2.9
1,2,2,2-Tetrafluoroethylformate	HCOOCHFCF3	1,720	569	78.6
1,1,1,3,3,3-Hexafluoropropan-2-ylformate	HCOOCH(CF3)2	1,220	403	55.7
Perfluorobutylacetate	CH3COOCF2CF2CF2CF3	6	2	0.3
Perfluoropropylacetate	CH3COOCF2CF2CF3	6	2	0.3
Perfluoroethylacetate	CH3COOCF2CF3	8	3	0.3
Trifluoromethylacetate	CH3COOCF3	8	3	0.3
Methylcarbonofluoridate	FCOOCH3	350	116	15.9
1,1-Difluoroethylcarbonofluor idate	FCOOCF2CH3	99	33	4.5
1,1-Difluoroethyl2,2,2-trifluoroacetate	CF3COOCF2CH3	113	38	5.2
Ethyl 2,2,2-trifluoroacetate	CF3COOCH2CH3	5	2	0.2
2,2,2-Trifluoroethyl2,2,2-trifluoroacetate	CF3COOCH2CF3	25	8	1.1
Methyl 2,2,2-trifluoroacetate	CF3COOCH3	192	64	8.8
Methyl 2,2-difluoroacetate	HCF2COOCH3	12	4	0.5

Difluoromethyl 2,2,2-	CF3COOCHF2	99	33	4.5
trifluoroacetate				
2,2,3,3,4,4,4-Heptafluorobutan-1-ol	C3F7CH2OH	124	41	5.7
1,1,2-Trifluoro-2-(trifluorom ethoxy)- ethane 1-Ethoxy-1,1,2,3,3,3-	CHF2CHFOCF3 CF3CHFCF2OC H2	3,970 86	1,489 28	207.9 3.9
hexafluoropropan e	CH3			
1,1,1,2,2,3,3-Heptafluoro-3-(1,2,2,2-tetrafluoroetho xy)- propane 2,2,3,3-Tetrafluoro-1-	CF3CF2CF2OC HF CF3 CHF2CF2CH2O H	7,940 48	7,371 16	1,400.4 2.2
propanol 2,2,3,4,4,4-Hexafluoro-	CF3CHFCF2CH 2O	63	21	2.8
1-butanol	H			
2,2,3,3,4,4,4-Heptafluoro-1-butanol	CF3CF2CF2CH 2O H	60	20	2.7

1,1,2,2-Tetrafluoro-3-methoxy-propane	CHF2CF2CH2 OC H3	2	1	0.1
perfluoro-2-methyl-3-pentanone	CF3CF2C(O)CF (C F3)2	0	0	0.0
3,3,3-Trifluoropropanal	CF3CH2CHO	0	0	0.0
2-Fluoroethanol	CH2FCH2OH	3	1	0.1
2,2-Difluoroethanol	CHF2CH2OH	11	4	0.5
2,2,2-Trifluoroethanol	CF3CH2OH	73	24	3.3
1,1'-Oxybis[2-(difluoromethoxy)-1,1,2,2-tetrafluoroethane	HCF2O(CF2CF 2O)2CF2H	9,910	5,741	840.5
1,1,3,3,4,4,6,6,7,7,9,9,1	HCF2O(CF2CF 2O	9,050	5,245	768.4
0,10,12,12-hexadecafluoro-2,5,8,11-Tetraoxadodecan e 1,1,3,3,4,4,6,6,7,7,9,9,1	)3CF2H HCF2O(CF2CF 2O	7,320	4,240	621.6
0,10,12,12,13,13,15,15-	)4CF2H			

eicosafluoro-2,5,8,11,14-Pentaoxapentadecane				

Appendix 9

Midpoint characterisation factors in units (kg CFC-11 equivalents/kg) for 21 ODSs related to three perspectives (RIVM Report 2016-0104, p. 41)

	Individualist (20 year)	**Hierarchist (100 year)**	**Egalitarian (infinite)**
Annex A-I			
CFC-11	1	1	1
CFC-12	0.421	0.587	0.820
CFC-113	0.504	0.664	0.850
CFC-114	0.165	0.270	0.580
CFC-115	0.032	0.061	0.570
Annex A-II			
Halon-1301	11.841	14.066	15.900
Halon-1211	15.053	8.777	7.900
Halon-2402	22.200	14.383	13.000
Annex B-II			
CCl_4	1.203	0.895	0.820
Annex B-III			
CH_3CCl_3	0.396	0.178	0.160
Annex C-I			
HCFC-22	0.085	0.045	0.040
HCFC-123	0.025	0.011	0.010
HCFC-124	0.049	0.022	0.020
HCFC-141b	0.275	0.134	0.120
HCFC-142b	0.111	0.067	0.060
HCFC-225ca	0.050	0.022	0.020
HCFC-225cb	0.073	0.033	0.030
Annex E			
CH_3Br	1.649	0.734	0.660
Others			
Halon-1202	4.247	1.892	1.700
CH_3Cl	0.050	0.022	0.020
N_2O*	0.007	0.011	0.017

* ODPs for N2O should be considered preliminary, because action mode is different from the other ODSs and ODPs remain infinite as more uncertain.

Appendix 10

Characterisation factors of midpoint in kBq Co-60 to air eq/kBq per emission compartment (RIVM Report 2016-0104, pp. 47-48)

Radionuclide	Individualist	Hierarchist	Egalitarian
Emissions to air			
Am-241	5.45E+01	5.45E+01	5.55E+01
C-14	6.14E-01	1.15E+00	1.29E+01
Co-58	2.55E-02	2.55E-02	2.55E-02
Co-60	1.00E+00	1.00E+00	1.00E+00
Cs-134	7.18E-01	7.18E-01	7.18E-01
Cs-137	1.27E+00	1.64E+00	1.64E+00
H-3	8.55E-04	8.56E-04	8.56E-04
I-129	8.32E+00	1.05E+01	2.07E+02
I-131	9.09E-03	9.09E-03	9.09E-03
I-133	5.64E-04	5.64E-04	5.64E-04
Kr-85	6.03E-06	8.48E-06	8.48E-06
Pb-210	-	9.09E-02	9.09E-02
Po-210	9.09E-02	9.09E-02	9.09E-02
Pu alpha[a]	-	-	5.00E+00
Pu-238	-	-	4.00E+00
Pu-239	3.18E+01	3.18E+01	3.18E+01
Ra-226	-	-	5.45E-02
Rn-222	1.45E-03	1.45E-03	1.45E-03
Ru-106	1.00E-01	1.00E-01	1.00E-01
Sr-90	1.52E+00	2.45E+00	2.45E+00
Tc-99	7.57E-01	1.18E+00	1.18E+00
Th-230	-	-	2.73E+00
U-234[a]	-	-	5.82E+00
U-235[a]	-	-	1.27E+00
U-238[a]	-	-	4.91E-01
Xe-133	8.55E-06	8.55E-06	8.55E-06
Emissions to fresh water (rivers and lakes)			
Ag-110m	3.00E-02	3.00E-02	3.00E-02
Am-241	3.36E-03	3.45E-03	3.64E-03
C-14	3.45E-03	6.09E-03	1.27E-02
Co-58	2.45E-03	2.45E-03	2.45E-03
Co-60	2.64E+00	2.64E+00	2.64E+00
Cs-134	8.64E+00	8.64E+00	8.64E+00
Cs-137	9.09E+00	1.00E+01	1.00E+01
H-3	4.07E-05	4.12E-05	4.12E-05
I-129	2.52E-01	2.87E-01	1.55E+02
I-131	3.00E-02	3.00E-02	3.00E-02
Mn-54	1.91E-02	1.91E-02	1.91E-02
Pu-239	3.45E-04	3.73E-04	4.18E-04
Ra-226[a]	-	-	7.73E-03
Ru-106	2.36E-04	2.36E-04	2.36E-04
Sb-124	4.91E-02	4.91E-02	4.91E-02

Radionuclide	Individualist	Hierarchist	Egalitarian
Sr-90	1.27E-02	2.45E-02	2.82E-02
Tc-99	7.55E-03	3.09E-02	3.09E-02
U-234[a]	-	-	1.45E-01
U-235[a]	-	-	1.36E-01
U-238[a]	-	-	1.36E-01
Emissions to the marine environment			
Am-241	4.73E-02	4.82E-02	4.82E-02

C-14	2.73E-02	2.73E-02	2.73E-02
Cm alpha[a]	-	-	3.45E+00
Co-60	2.36E-02	2.36E-02	2.36E-02
Cs-134	4.73E-03	4.73E-03	4.73E-03
Cs-137	5.82E-03	5.82E-03	5.82E-03
H-3	3.60E-06	4.05E-06	4.05E-06
I-129	2.22E-02	3.00E-02	1.55E+02
Pu alpha[a]	-	-	4.45E+00
Pu-239	5.27E-03	5.36E-03	5.73E-03
Ru-106	1.09E-03	1.09E-03	1.09E-03
Sb-125	8.91E-04	8.91E-04	8.91E-04
Sr-90	4.55E-04	4.55E-04	4.55E-04
Tc-99	7.82E-05	7.91E-05	1.09E-04
U-234[a]	-	-	1.36E-03
U-235[a]	-	-	2.03E-03
U-238[a]	-	-	8.33E-04

Appendix 11

Midpoint characterisation factors for individual NMVOCs (RIVM Report 2016-0104, pp. 135-138)

CAS nr	**Substance name**	**HOFP (NO_X-K)**
000074-84-0	Ethane	0.0
000074-98-6	Propane	0.05
000106-97-8	Butane	0.11
000075-28-5	i-Butane	0.10
000109-66-0	Pentane	0.15
000078-78-4	i-Pentane	0.12
000463-82-1	Neopentane	0.07
000110-54-3	Hexane	0.15
000107-83-5	2-Methylpentane	0.15
000096-14-0	3-Methylpentane	0.16
000075-83-2	2,2-Dimethylbutane	0.08
000079-29-8	2,3-Dimethylbutane	0.18
000142-82-5	Heptane	0.13
000591-76-4	2-Methylhexane	0.12
000589-34-4	3-Methylhexane	0.15
000111-65-9	Octane	0.12
000592-27-8	2-Methylheptane	0.12
000589-81-1	3-Methylheptane	0.13
000111-84-2	Nonane	0.12

003221-61-2	2-Methyloctane	0.12
002216-33-3	3-Methyloctane	0.12
002216-34-4	4-Methyloctane	0.13
000922-28-1	3,4-Dimethylheptane	0.13
000124-18-5	Decane	0.13
000871-93-0	2-Methylnonane	0.13
005911-04-6	3-Methylnonane	0.14
017301-94-9	4-Methylnonane	0.13
015869-89-3	2,5-Dimethyloctane	0.14
002051-30-1	2,6-Dimethyloctane	0.13
014676-29-0	2-Methyl-3-ethylheptane	0.12
013475-81-5	2,2-Dimethyl-3,3-dimethylhexane	0.07
001120-21-4	Undecane	0.13
006975-98-0	2-Methyldecane	0.12
013151-34-3	3-Methyldecane	0.13
002847-72-5	4-Methyldecane	0.13
013151-35-4	5-Methyldecane	0.13
000112-40-3	Dodecane	0.12
000629-50-5	Tridecane	0.15
000629-59-4	Tetradecane	0.17

CAS nr	Substance name	HOFP (NO_X-K)
000096-37-7	Methylcyclopentane	0.18
000110-82-7	Cyclohexane	0.10
000108-87-2	Methylcyclohexane	0.24
001678-91-7	Ethylcyclohexane	0.23
001678-92-8	Propylcyclohexane	0.22
001678-97-3	1,2,3-Trimethylcyclohexane	0.21
000696-29-7	i-Propylcyclohexane	0.22
001678-93-9	Butylcyclohexane	0.21
001678-98-4	i-Butylcyclohexane	0.21
004291-80-9	1-Methyl-3-propylcyclohexane	0.22
004291-81-0	1-Methyl-4-propylcyclohexane	0.20
004292-92-6	Pentylcyclohexane	0.20
004292-75-5	Hexylcyclohexane	0.20
000074-85-1	Ethylene	0.36
000115-07-1	Propylene	0.42
000106-98-9	But-1-ene	0.38
000590-18-1	Cis-but-2-ene	0.41
000624-64-6	Trans-but-2-ene	0.42
000106-98-9	Butylene	0.23
000106-99-0	1,3-Butadiene	0.32
000627-20-3	Cis-pent-2-ene	0.40
000646-04-8	Trans-pent-2-ene	0.40
000109-67-1	1-Pentene	0.34
000563-46-2	2-Methylbut-1-ene	0.27

000563-45-1	3-Methylbut-1-ene	0.26
000513-35-9	2-Methylbut-2-ene	0.30
000078-79-5	Isoprene	0.41
000592-41-6	Hex-1-ene	0.32
007688-21-3	Cis-hex-2-ene	0.38
009016-80-2	Trans-hex-2-ene	0.37
000080-56-8	Alpha-pinene	0.25
000127-91-3	Beta-pinene	0.12
000138-86-3	Limonene	0.26
004516-90-9	2-Methyl-3-butenol	-0.01
000071-43-2	Benzene	0.04
000108-88-3	Toluene	0.16
000095-47-6	o-Xylene	0.28
000108-38-3	m-Xylene	0.31
000106-42-3	p-Xylene	0.26
000100-41-4	Ethylbenzene	0.17
000103-65-1	Propylbenzene	0.14
000098-82-8	i-Propylbenzene	0.12

CAS nr	**Substance name**	**HOFP (NO_X-K)**
000526-73-8	1,2,3-Trimethylbenzene	0.38
000095-63-6	1,2,4-Trimethylbenzene	0.40
000108-67-8	1,3,5-Trimethylbenzene	0.39
000611-14-3	o-Ethyltoluene	0.26
000620-14-4	m-Ethyltoluene	0.28

000622-96-8	p-Ethyltoluene	0.23
029224-55-3	3,5-Dimethylethylbenzene	0.38
025550-13-4	3,5-Diethyltoluene	0.36
000527-53-7	1,2,3,5-Tetramethylbenzene	0.38
000095-93-2	1,2,4,5-Tetramethylbenzene	0.36
000099-87-6	1-Methyl-4-i-propylbenzene	0.27
000535-77-3	1-Methyl-3-i-propylbenzene	0.32
000100-42-5	Styrene	0.02
000050-00-0	Formaldehyde	0.17
000075-07-0	Acetaldehyde	0.20
000123-38-6	Propionaldehyde	0.26
-	i-Propionaldehyde	0.18
000123-72-8	Butyraldehyde	0.25
000100-62-3	Pentanal	0.26
000590-86-3	3-Methylbutanal	0.15
000100-52-7	Benzaldehyde	-0.07
000529-20-4	2-Methylbenzaldehyde	-0.10
000620-23-5	3-Methylbenzaldehyde	-0.07
000104-87-0	4-Methylbenzaldehyde	0.02
000067-56-1	Methanol	0.05
000064-17-5	Ethanol	0.12
000071-23-8	Propanol	0.17
000067-63-0	i-Propanol	0.07
000071-36-3	Butanol	0.19
000078-83-1	i-Butanol	0.13

000078-92-2	sec-butanol	0.15
000075-65-0	t-Butanol	0.01
000123-51-3	3-Methyl-1-butanol	0.16
000108-95-2	Phenol	-0.02
000095-48-7	o-Cresol	0.07
000095-87-4	2,5-Xylenol	0.20
000105-67-9	2,4-Xylenol	0.20
000526-75-0	2,3-Xylenol	0.12
000108-93-0	Cyclohexanol	0.16
000123-42-2	Diacetone alcohol	0.11
000067-64-1	Acetone	0.02
000078-93-3	Methylethylketone	0.12

CAS nr	**Substance name**	**HOFP (NO_X-K)**
000108-10-1	Methyl-i-butylketone	0.19
000108-94-1	Cyclohexanone	0.11
000107-87-9	Methylpropylketone	0.00
000107-31-3	Methyl formate	0.01
000079-20-9	Methyl acetate	0.03
000141-78-6	Ethyl acetate	0.07
000108-21-4	i-Propyl acetate	0.08
000123-86-4	Butyl acetate	0.09
000109-60-4	n-Propyl acetate	0.09
000064-18-6	Formic acid	0.01
000064-19-7	Acetic acid	0.03
000079-09-4	Propanoic acid	0.05

000115-10-6	Dimethylether	0.07
000060-29-7	Diethylether	0.17
000108-20-3	Di-i-propylether	0.16
000107-21-1	Ethylene glycol	0.12
000057-55-6	Propylene glycol	0.14
000111-76-2	2-Butoxyethanol	0.16
000107-98-2	1-Methoxy-2-propanol	0.12
000109-86-4	2-Methoxyethanol	0.11
000110-80-5	2-Ethoxyethanol	0.13
000107-02-8	Acrolein	0.20
000078-85-3	Methacrolein	0.33
000107-22-2	Glyoxal	0.08
000078-98-8	Methylglyoxal	0.37
000074-86-2	Acetylene	0.03
000074-99-7	Propyne	0.26
000075-09-2	Methylene dichloride	0.01
000075-00-3	Ethyl chloride	0.04
000127-18-4	Tetrachloroethylene	0.00
000079-01-6	Trichloroethylene	0.11
000075-34-3	Ethylidene dichloride	0.20
000071-55-6	Methyl chloroform	0.00
000074-87-3	Methyl chloride	0.00
000156-59-2	Cis-dichloroethylene	0.00
000156-60-5	Trans-dichloroethylene	0.00
000067-66-3	Chloroform	0.00

Appendix 12

Additions to USEtox organic and inorganic database (RIVM Report 2016-0104, p. 170)

Name	**Unit**	**Source**
Dimensionless plant/air partition coefficient vegetation	m^3/m^3	Extracted from the original USES-LCA 2.0 substance database for organics
OVERALL MASS TRANSFER COEFFICIENT air/plant interface	m/s	Extracted from the original USES-LCA 2.0 substance database for organics
Root/soil PARTITION COEFFICIENT	kg(wwt)/kg(wwt)	Extracted from the original USES-LCA 2.0 substance database for organics
Leaf/soil PARTITION COEFFICIENT	kg(wwt)/kg(wwt)	Extracted from the original USES-LCA 2.0 substance database for organics
Transpiration Stream Concentration Factor	-	Extracted from the original USES-LCA 2.0 substance database for organics. Additionally, the plant uptake model of Trapp (2009) is included.
Root Concentration Factor	l/kg wwt	Extracted from the original USES-LCA 2.0 substance database for organics. Additionally, the plant uptake model of Trapp (2009) is included.
Bioaccumulation factor for meat	d/kg(food)	Calculated (for both neutral and dissociating organics) using the regressions of et al. Hendriks, (2007)

Bioaccumulation factor for milk	d/kg(food)	Calculated (for both neutral and dissociating organics) using the regressions of et al.Hendriks, (2007)
Fish/water PARTITION COEFFICIENT	l/kg	Calculated for dissociating organics using the regressions of et al. Fu, (2009)
Bioavailability for oral uptake	-	Extracted from the original USES-LCA 2.0 substance database for both organics and metals
Bioavailability for inhalation	-	Extracted from the original USES-LCA 2.0 substance database for both organics and
IARC classification	-	(IARC 2004)
FRACTION in gas phase air (METAL/INORGANIC)	-	Extracted from the original USES-LCA 2.0 substance database for metals
Gas WASHOUT (METAL/INORGANIC)	$m.s^{-1}$	Extracted from the original USES-LCA 2.0 substance database for metals
Aerosol COLLECTION EFFICIENCY	-	Extracted from the original USES-LCA 2.0 substance database for metals

Appendix 13

Cultural perspectives and four emission compartments of ETPs and HTPs (RIVM Report 2016-0104, p. 77-79)

Substance	Emission compartment	Individualist	Hierarchist	Egalitarian
Freshwater ecotoxicity				
1,4-Dichlorobenzene	Urban air	1.3E-03	1.3E-03	1.3E-03
1,4-Dichlorobenzene	Fresh water	**1**	**1**	**1**
1,4-Dichlorobenzene	Seawater	5.5E-04	5.5E-04	5.5E-04
1,4-Dichlorobenzene	Industrial soil	3.2E-02	3.2E-02	3.2E-02
Nickel	Urban air	6.8E-01	2.2E+00	1.6E+01
Nickel	Fresh water	4.2E+01	4.6E+01	4.6E+01
Nickel	Seawater	0.0E+00	0.0E+00	0.0E+00
Nickel	Industrial soil	4.6E-01	3.2E+00	4.2E+01
Marine ecotoxicity				
1,4-Dichlorobenzene	Urban air	1.5E-01	1.5E-01	1.8E-01
1,4-Dichlorobenzene	Fresh water	1.8E-01	1.8E-01	1.8E-01
1,4-Dichlorobenzene	Seawater	**1**	**1**	**1**
1,4-Dichlorobenzene	Industrial soil	8.2E-02	8.2E-02	8.2E-02
Nickel	Urban air	3.1E+01	1.1E+02	5.5E+04
Nickel	Fresh water	1.3E+01	5.7E+01	2.5E+04
Nickel	Seawater	9.8E+01	3.2E+02	1.3E+05
Nickel	Industrial soil	9.4E-02	2.3E+00	2.3E+04

Terrestrial ecotoxicity				
1,4-Dichlorobenzene	Urban air	6.3E-03	6.3E-03	6.3E-03
1,4-Dichlorobenzene	Fresh water	5.7E-03	5.7E-03	5.7E-03
1,4-Dichlorobenzene	Seawater	2.7E-03	2.7E-03	2.7E-03
1,4-Dichlorobenzene	Industrial soil	**1**	**1**	**1**
Nickel	Urban air	2.1E+01	5.4E+01	2.1E+02
Nickel	Fresh water	0.0E+00	0.0E+00	0.0E+00
Nickel	Seawater	0.0E+00	0.0E+00	0.0E+00
Nickel	Industrial soil	7.6E+00	3.7E+01	4.5E+02
Human toxicity (carcinogenic)				
1,4-Dichlorobenzene	Urban air	**1**	**1**	**1**
1,4-Dichlorobenzene	Fresh water	6.9E-01	6.9E-01	6.9E-01
1,4-Dichlorobenzene	Seawater	1.9E-01	1.9E-01	1.9E-01
1,4-Dichlorobenzene	Industrial soil	2.3E-01	2.3E-01	2.3E-01
Nickel	Urban air	3.1E+01	3.7E+02	9.1E+02
Nickel	Fresh water	3.4E+00	2.3E+01	2.5E+02
Nickel	Seawater	0.0E+00	3.5E+00	1.2E+03
Nickel	Industrial soil	2.1E+00	1.2E+01	3.6E+02
Human toxicity (non-carcinogenic)				
1,4-Dichlorobenzene	Urban air	**1**	**1**	**1**
1,4-Dichlorobenzene	Fresh water	8.5E-01	8.5E-01	8.5E-01
1,4-Dichlorobenzene	Seawater	1.9E-01	1.9E-01	1.9E-01

1,4-Dichlorobenzene	Industrial soil	2.3E-01	2.3E-01	2.3E-01
Nickel	Urban air	5.8E+00	7.1E+01	1.7E+02
Nickel	Fresh water	6.5E+01	4.4E+00	4.8E+01
Nickel	Seawater	0.0E+00	6.7E-01	2.3E+02
Nickel	Industrial soil	4.1E-01	2.4E+00	6.8E+01

Appendix 14

Supporting data for land use (RIVM Report 2016-0104, p. 189)

LAND USE	CFm_{relax} (annual crop eq)			
	Mammals	**Birds**	**Arthropods**	**Vascular plants**
Pasture and Meadow	0.55	0.33	0.42	0.18
Annual crops	0.75	0.88	1.08	0.70
Permanent Crops	0.45	1.03	0.93	0.47
Mosaic Agriculture	-0.23	0.37	0.07	0.62
Artificial areas	-	-	-	-0.70

Supporting data for land use (RIVM Report 2016-0104, p. 189)

LAND USE	t_{rel} (year)											
	Mammals		Birds		Insects		Invertebrates		Plants		Trees	
Type	Forest	Open	Forest	Open	Forest	Open	Forest	Open	Forest	Open	Forest	Open
Pasture and Meadow	69.4	7.2	68.6	6.3	69.2	7.0	79.5	8.0	73.5	7.3	102.2	10.4
Annual crops	69.4	7.2	68.6	6.3	69.2	7.0	79.5	8.0	73.5	7.3	102.2	10.4
Permanent Crops	69.4	7.2	68.6	6.3	69.2	7.0	79.5	8.0	73.5	7.3	102.2	10.4
Mosaic Agriculture	69.4	7.2	68.6	6.3	69.2	7.0	79.5	8.0	73.5	7.3	102.2	10.4
Artificial areas	69.4	7.2	68.6	6.3	69.2	7.0	79.5	8.0	73.5	7.3	102.2	10.4

Supporting data for land use (RIVM Report 2016-0104, p. 190)

LAND USE	CFm_{relax} (annual crop eq·yr)											
	Mammals		Birds		Insects		Invertebrates		Plants		Trees	
Type	Forest	Open	Forest	Open	Forest	Open	Forest	Open	Forest	Open	Forest	Open
Pasture and Meadow	19.1	2.0	11.4	1.1	14.4	1.5	16.6	1.7	6.7	0.7	9.4	1.0
Annual crops	26.0	2.7	30.3	2.8	37.5	3.8	43.1	4.3	25.7	2.5	35.8	3.6
Permanent Crops	15.6	1.6	35.5	3.3	32.3	3.3	37.1	3.7	17.2	1.7	23.8	2.4
Mosaic Agriculture	-8.1	-0.8	12.6	1.2	2.3	0.2	2.7	0.3	22.7	2.2	31.5	3.2
Artificial areas	-	-	-	-	-	-	-	-	25.7	2.5	-35.8	-3.6

Appendix 15

Applicable land occupation categories in EcoInvent with ReCiPe (RIVM Report 2016-0104, p. 192)

Name in EcoInvent	**Name in ReCiPe**
Occupation, pasture, man-made, intensive	Pasture
Occupation, permanent crop, non-irrigated, intensive	permanent crops
Occupation, mineral extraction site	artificial area
Occupation, annual crop, greenhouse	artificial area
Occupation, permanent crop, irrigated, intensive	permanent crops
Occupation, industrial area	artificial area
Occupation, construction site	artificial area
Occupation, annual crop, non-irrigated, intensive	annual crops
Occupation, traffic area, road network	artificial area
Occupation, annual crop, irrigated, intensive	annual crops
Occupation, dump site	artificial area
Occupation, river, artificial	-
Occupation, annual crop	annual crops
Occupation, lake, artificial	-
Occupation, annual crop, non-irrigated, extensive	annual crops
Occupation, traffic area, rail/road embankment	artificial area
Occupation, pasture, man-made, extensive	pasture
Occupation, pasture, man-made, extensive	-
managed forest Occupation, forest, intensive	-
managed forest Occupation, permanent crop	-
permanent crops Occupation, traffic area, rail network	-
artificial area Occupation, seabed, infrastructure	-
Occupation, seabed, drilling and mining	-
Occupation, annual crop, non-irrigated	-
Occupation, shrub land, sclerophyllous	annual crops
Occupation, annual crop, irrigated	managed forest
Occupation, urban, discontinuously built	annual crops
Occupation, pasture, man-made	artificial area
Occupation, grassland, natural (non-use)	pasture
Occupation, urban/industrial fallow (non-use	-
Occupation, pasture, man-made, intensive	artificial area
Occupation, grassland, natural (non-use)	Pasture
Occupation, urban/industrial fallow (non-use)	permanent crops

Appendix 16

Data about parameters alpha and beta for every covered mineral resource (RIVM Report 2016-0104, p. 194)

	Cumulative grade-tonnage regression parameters			**Cumulative mineral resource extracted (CME in kg *x*)**	**Reserves (R in kg *x*)**	**Ultimate recoverable resource (URR in kg *x*)**
Mineral resource	**Scale α**	**Shape β**	**R^2**			
Aluminium	-1.35	0.10	0.91	1.04E+12	1.48E+13	1.34E+16
Antimony	-2.06	0.42	0.85	6.79E+09	1.80E+09	6.61E+10
Chromium	-1.15	0.12	0.87	2.06E+11	1.48E+11	1.52E+13
Cobalt	-4.86	0.17	0.95	2.28E+09	7.20E+09	2.86E+12
Copper	-3.61	0.17	0.79	5.92E+11	6.90E+11	4.36E+12
Gold	-11.9	0.20	0.86	1.44E+08	5.40E+07	7.20E+07
Iron	-0.57	0.13	0.93	3.41E+13	8.10E+13	6.46E+15
Lead	-2.61	0.21	0.85	2.35E+11	8.90E+10	2.81E+12
Lithium	-4.95	0.11	0.72	9.81E+09	1.30E+10	3.47E+12
Manganese	-1.19	0.08	0.77	5.80E+11	5.70E+11	1.27E+14
Molybdenum	-6.34	0.27	0.94	6.62E+09	1.10E+10	1.82E+11
Nickel	-4.26	0.16	0.93	5.53E+10	7.40E+10	7.76E+12
Niobium	-4.39	0.27	0.70	1.07E+09	4.30E+09	4.80E+11
Phosphorus	-2.14	0.10	0.93	9.78E+11	2.18E+12	1.90E+13
Silver	-8.08	0.26	0.73	1.13E+09	5.20E+08	2.00E+10
Tin	-4.95	0.21	0.79	2.00E+10	4.70E+09	2.20E+11
Uranium	-5.54	0.50	0.86	2.71E+09	2.52E+09	4.30E+11
Zinc	-1.62	0.15	0.70	4.58E+11	2.50E+11	1.11E+13

Appendix 17

Midpoint characterisation factors SOPs (kg Cu-eq/kg) for 70 mineral resources (RIVM Report 2016-0104, p. 99-100)

Mineral resource	**Chemical element**	**Individualist**	**Hierarchist**	**Egalitarian**
Aluminium	Al	1.01E-01	1.69E-01	1.69E-01
Antimony	Sb	1.03E+00	5.72E-01	5.72E-01
Arsenic*	As	8.89E-02	1.31E-01	1.31E-01
Ball clay*		3.86E-03	7.09E-03	7.09E-03
Barite*		1.36E-02	2.28E-02	2.28E-02
Bauxite*		2.41E-03	4.58E-03	4.58E-03
Bentonite clay*		6.07E-03	1.08E-02	1.08E-02
Beryllium*	Be	8.42E+01	7.67E+01	7.67E+01
Bismuth*	Bi	2.77E+00	3.20E+00	3.20E+00
Boron*	B	7.77E-02	1.16E-01	1.16E-01
Cadmium	Cd	2.32E-01	3.20E-01	3.20E-01
Caesium	Ce	1.90E+04	1.18E+04	1.18E+04
Chromium	Cr	5.57E-02	9.51E-02	9.51E-02
Chrysotile*		2.21E-01	3.05E-01	3.05E-01
clay				
unspecified*		5.85E-03	1.04E-02	1.04E-02
Cobalt	co	4.01E+00	6.57E+00	6.57E+00
Copper	Cu	1.00E+00	1.00E+00	1.00E+00
Diamond				
Industrial	c	1.02E+02	9.15E+01	9.15E+01
Diatomite		3.07E-02	4.88E-02	4.88E-02
Feldspar*		8.90E-03	1.54E-02	1.54E-02
Fire clay*		1.95E-03	3.76E-03	3.76E-03

Fuller's earth*		8.61E-03	1.50E-02	1.50E-02
Gallium*		9.28E+01	8.38E+01	8.38E+01
Germanium*		3.89E+02	3.17E+02	3.17E+02
Gold		5.12E+03	3.73E+03	3.73E+03
Graphite*		1.34E-01	1.92E-01	1.92E-01
Gypsum*		1.44E-03	2.83E-03	2.83E-03
Hafnium*		1.08E+02	9.67E+01	9.67E+01
Ilmenite*		2.40E-02	3.88E-02	3.88E-02
Indium*	In	1.15E+02	1.03E+02	1.03E+02
Iodine*	I	6.51E+00	7.09E+00	7.09E+00
Iron	Fe	3.82E-02	6.19E-02	6.19E-02
Iron ore*		1.02E-02	1.75E-02	1.75E-02
Kaolin*		1.46E-02	2.45E-02	2.45E-02
Kyanite*		3.15E-02	5.00E-02	5.00E-02
Lead	Pb	4.83E-01	4.91E-01	4.91E-01
Lime*		1.19E-02	2.02E-02	2.02E-02
Lithium	Li	2.42E+00	4.86E+00	4.86E+00
Magnesium*	Mg	6.14E-01	7.90E-01	7.90E-01
Manganese	Mn	3.76E-02	8.23E-02	8.23E-02
Mercury*	Hg	8.37E+00	8.96E+00	8.96E+00
Molybdenum	Mo	2.90E+01	2.92E+01	2.92E+01
Nickel	Ni	1.85E+00	2.89E+00	2.89E+00
Niobium	Nb	4.46E+00	5.20E+00	5.20E+00
Palladium*	Pd	6.37E+03	4.28E+03	4.28E+03
Perlite*		5.08E-03	9.16E-03	9.16E-03
Phosphorus	P	1.40E-01	1.67E-01	1.67E-01
Platinum*	Pt	1.38E+04	8.77E+03	8.77E+03

*ASOPs extrapolated from price data.

Appendix 18

Fossil fuel potential (FFP) for five fossil resources (RIVM Report 2016-0104, p. 104)

Fossil resource	**Unit**	**Characterisation factor**
Crude oil	oil-eq/kg	1
Natural gas	oil-eq/Nm^3	0.84
Hard coal	oil-eq/kg	0.42
Brown coal	oil-eq/kg	0.22
Peat	oil-eq/kg	0.22

Appendix 19

Equation data of cumulative cost-tonnage parameters a and b for every fossil resource (RIVM Report 2016-0104, p. 197)

	Cumulative cost-tonnage regression parameters		**Surplus cost potential**		
Fossil resource	**Intercept** ***a***	**Slope** ***b***	**Cum. fossil extracted (CFE in kg or Nm^3 *x*)**	**Current cost (C in USD2008/kg or Nm3 *x*)**	**Reserves (R in kg or Nm^3 *x*)**
Crude oil	40.0	-4.45	1.61E+14	7.33E-02	8.05E+14
Hard coal	36.6	-5.19	NA	2.85E-02	7.19E+14
Natural gas	23.2	-2.42	1.01E+14	6.85E-03	7.99E+14

Supporting data for derived FFPs (Jungbluth & Frischknecht 2010), (RIVM Report 2016-0104, p. 197)

Name	**Unit**	**Higher heating value (HHV)**
Brown coal	MJ-eq/kg	9.9
Crude oil	MJ-eq/kg	45.8
Hard coal	MJ-eq/kg	19.1
Natural gas	MJ-eq/Nm^3	38.3
Peat	MJ-eq/kg	9.9

Appendix 20

Five fossil resources for endpoint characterisation factors in USD2013/unit of resource (RIVM Report 2016-0104, p. 105)

Fossil resource	**Unit**	**Individualist**	**Hierarchist**	**Egalitarian**
Crude oil	USD2013/kg	0.457	0.457	0.457
Hard coal	USD2013/kg	0.034	0.034	0.034
Natural gas	USD2013/Nm^3	0.301	0.301	0.301
Brown coal*	USD2013/kg			0.034
Peat*	USD2013/kg			0.034

*No characterisation factors were calculated clearly for brown coal and peat, because of a lack of information for production and cost. Moreover, the egalitarian perspective from a precautionary viewpoint is provided as a proxy as a characterisation factor of hard coal.

Appendix 21

Total global emissions and derived weights per substance and emission compartment (RIVM Report 2016-0104, p. 107-110)

Substance	Emission air	Weight air	Emission water	Weight water	Emission soil	Weight soil
Aldehydes, unspecified (kg)						
2-Butenal	2.32E+04	0.0002	3.46E+04	0.0019	-	-
Acetalde-hyde	2.73E+07	0.2741	7.22E+06	0.4021	-	-
Benzalde-hyde	5.64E+05	0.0057	1.65E+06	0.0919	-	-
Formalde-hyde	7.17E+07	0.7200	9.05E+06	0.5040	-	-
Anthracene	3.77E+04	0.0084	9.55E+02	0.0196	1.85E+05	1
Benzo(a)-pyrene	1.10E+06	0.2459	4.28E+01	0.0009	-	-
Fluoran-thene	2.01E+05	0.0449	8.67E+01	0.0018	-	-
Naphthalene	2.41E+06	0.5388	4.67E+04	0.9561	-	-
Phenan- threne	5.56E+05	0.1243	1.02E+03	0.0209	-	-

Substance	Emission air	Weight air	Emission water	Weight water	Emission soil	Weight soil
Pyrene	1.68E+05	0.0376	3.98E+01	0.0008	-	-
Actinides, unspecified (kBq)						
Americium- 241	1.40E+06	0.0290	9.35E+08	0.0091	-	-
Uranium-234	1.94E+07	0.4020	9.22E+08	0.0090	-	-
Uranium-235	8.42E+05	0.0174	4.05E+07	0.0004	-	-
Uranium-238	1.82E+07	0.3771	1.16E+09	0.0113	-	-
Plutonium-241	8.42E+06	0.1745	9.98E+10	0.9703	-	-

Carboxylic acids (kg)						
Formic acid	7.24E+05	0.4254	2.62E+05	0.7973	-	-
Acrylic acid	9.78E+05	0.5746	6.66E+04	0.2027	-	-
Hydrocarbons, chlorinated (kg)						
Ethane, 1,1,1-trichloro-,HCFC-140	3.57E+05	3.70E-03	7.41E+04	4.10E-02	-	-
Methane, tetrachloro-, CFC-10	4.17E+05	4.32E-03	-	-	-	-
Ethane,1,1,1,2- tetrachloro-	4.80E+03	4.97E-05	-	-	-	-
Ethane, 1,1,2,2- tetrachloro-	4.59E+03	4.76E-05	1.22E+01	6.76E-06	-	-
Ethane, 1,1,2- trichloro-	5.75E+05	5.96E-03	2.40E+05	1.33E-01	-	-
Benzene, 1,2,4- trichloro-	1.49E+05	1.54E-03	1.12E+02	6.20E-05	-	-
Ethane, 1,2- dichloro-	2.43E+05	2.52E-03	5.91E+04	3.27E-02	-	-
Ethene, 1,2- dichloro-	9.33E+03	9.67E-05	1.52E+04	8.42E-03	-	-
Propane, 1,2- dichloro-	4.76E+06	4.93E-02	2.25E+03	1.25E-03	-	-
Benzene, 1,3- dichloro-	5.71E+03	5.92E-05	1.97E+02	1.09E-04	-	-
Propene, 1,3- dichloro-	1.73E+04	1.79E-04	2.77E+03	1.53E-03	-	-
Allyl chloride	5.83E+05	6.04E-03	1.21E+03	6.70E-04	-	-
Benzotrichloride	1.74E+03	1.80E-05	-	-	-	-
Benzyl chloride	1.95E+04	2.02E-04	3.88E+02	2.15E-04	-	-
Benzene, chloro-	1.72E+06	1.78E-02	6.66E+04	3.69E-02	-	-
Ethane, chloro-	5.30E+06	5.49E-02	9.42E+02	5.22E-04	-	-
Chloroform	4.20E+06	4.35E-02	5.56E+05	3.08E-01	-	-

Substance	Emission air	Weight air	Emission water	Weight water	Emission soil	Weight soil
Propene, 1- chloro-1-	9.16E+05	9.49E-03	0.00E+00	0.00E+00	-	-
Methane, dichloro-, HCC-30	1.42E+07	1.47E-01	2.73E+05	1.51E-01	-	-
Butadiene, hexachloro-	4.00E+03	4.14E-05	0.00E+00	0.00E+00	-	-
Benzene, hexachloro-	2.60E+04	2.69E-04	2.17E+02	1.20E-04	-	-
Cyclopenta- diene, hexachloro-	1.23E+03	1.27E-05	-	-	-	-
Ethane, hexachloro-	3.12E+04	3.23E-04	4.70E+00	2.60E-06	-	-
Methane, monochloro-, R-40	1.20E+07	1.24E-01	7.31E+03	4.05E-03	-	-
Toluene, 2- chloro-	4.76E+04	4.93E-04	8.27E+02	4.58E-04	-	-
Benzene, 1,2- dichloro-	4.31E+05	4.47E-03	1.05E+04	5.82E-03	-	-
Benzene, 1,4- dichloro-	2.08E+07	2.16E-01	2.78E+05	1.54E-01	-	-
Benzene, pentachloro-	1.53E+02	1.59E-06	1.64E+02	9.09E-05	-	-
Ethane, pentachloro-	1.29E+03	1.34E-05	-	-	-	-
Ethene, tetrachloro-	6.92E+06	7.17E-02	7.78E+04	4.31E-02	-	-
Ethene, dichloro- (trans)	4.12E+04	4.27E-04	2.07E+01	1.15E-05	-	-
Butene, 1,4- dichloro-2- (trans)	2.40E+02	2.49E-06	-	-	-	-
Ethene, trichloro-	2.03E+07	2.10E-01	1.06E+05	5.87E-02	-	-
Ethene, chloro-	2.43E+06	2.52E-02	3.24E+04	1.79E-02	-	-

Substance	Emission air	Weight air	Emission water	Weight water	Emission soil	Weight soil
Hydrocarbons, aliphatic, alkanes, cyclic						
Cyclohexane	7.73E+06	0.9418	1.92E+04	0.5310	-	-
Cyclohexa-nol	1.38E+05	0.0168	4.68E+03	0.1294	-	-
Cyclohexyla- mine	5.70E+04	0.0069	3.76E+02	0.0104	-	-
Dicyclo- pentadiene	2.83E+05	0.0345	1.19E+04	0.3291	-	-
Hydrocarbons, aromatic						
Benzene, 1,2,4- trimethyl-	8.52E+06	0.0136	7.88E+03	0.0127	-	-
Benzene, 1,3,5- trimethyl-	8.23E+06	0.0131	7.92E+02	0.0013	-	-
trimethyl- Benzene	1.48E+07	0.0236	8.53E+04	0.1371	-	-
Benzene, ethyl-	4.97E+07	0.0794	3.23E+04	0.0519	-	-
Toluene	5.45E+08	0.8703	4.96E+05	0.7971	-	-
Noble gases, radioactive						
Krypton-85	2.31E+15	0.9251	-	-	-	-
Argon-41	1.87E+14	0.0749	-	-	-	-
Radon-222	6.86E+10	0.0000	-	-	-	-

Appendix 22

CFs per substance group, impact category and sub-compartment as per Individualist perspective, (RIVM Report 2016-0104, p. 111-112)

Substance group	**(Sub)-compartment**	**Global warming** kg CO_2-eq	**Ozone depletion** kg CFC11-eq	**Ionizing radiation** kg Co60to air-eq	**Photo-Chemical ozone (human)** kg NO_X-eq	**Photo-Chemical ozone (eco)** kg NO_X-eq	**Freshwater ecotox** kg 1,4DCB to freshwater- eq	**Marine ecotox** kg 1,4DCB to saltwater- eq	**Terres-trial ecotox** kg 1,4DC to ind soil-eq	**Human tox (cancer)** kg 1,4DCB to urban air-eq	**Human tox (non-cancer)** kg 1,4DCB to urban air-eq
Aldehydes, unspecified	Urban air	-	-	-	1.74E-01	2.81E-01	2.28E-02	1.96E-02	7.30E+00	4.14E+01	3.36E+00
	Rural air	-	-	-	1.74E-01	2.81E-01	1.03E-02	2.91E-02	1.11E+01	1.35E+00	7.23E-01
	Fresh water	-	-	-	-	-	5.85E-01	1.94E-02	6.50E-01	8.16E-02	4.49E-01
	Seawater	-	-	-	-	-	9.66E-05	1.03E-01	6.87E-02	4.46E-03	2.42E-02
PAH, polycyclic aromatic hydro-carbons	Urban air	-	-	-	-	-	2.03E-01	8.53E+00	3.24E+01	3.25E+01	6.01E+00
	Rural air	-	-	-	-	-	3.44E-01	1.56E+01	6.15E+01	3.14E+01	2.48E-01
	Fresh water	-	-	-	-	-	1.30E+01	1.07E+00	3.34E+00	3.20E-02	1.05E+00
	Seawater	-	-	-	-	-	4.86E-03	1.10E+01	1.35E+00	2.30E-03	1.72E-01
	Industrial	-	-	-	-	-	6.02E+00	5.43E+00	3.81E+01	-	1.38E-02
	soil*										

	Agricultural	-	-	-	-	-	8.59E-01	5.05E+00	3.74E+01	-	4.16E-03
	soil*										
Actinides, unspecified	Air	-	-	1.58E+00	-	-	-	-	-	-	-
	Fresh water	-	-	3.05E-05	-	-	-	-	-	-	-
	Seawater	-	-	4.30E-04	-	-	-	-	-	-	-
Carboxylic	Urban air	-	-	-	4.63E-03	7.46E-03	2.19E-02	3.21E-03	5.47E+01	-	4.06E+02
acids	Rural air	-	-	-	4.63E-03	7.46E-03	6.39E-03	4.49E-03	8.18E+01	-	1.45E+01
	Fresh water	-	-	-	-	-	9.48E-02	1.33E-03	1.59E-03	-	1.52E-02
	Seawater	-	-	-	-	-	6.32E-10	9.40E-03	8.00E-06	-	3.52E-06
Hydro-	Urban air	3.02E	1.29E-	-	2.66E-02	4.29E-02	3.60E-04	5.17E-02	1.71E-01	1.87E+00	3.96E+01
carbons,		+01	02								
chlorinated	Rural air	3.02E+01	1.29E-02	-	2.66E-02	4.29E-02	3.63E-04	5.21E-02	1.74E-01	1.26E+00	2.27E+01
	Fresh water	-	-	-	-	-	3.35E-01	6.33E-02	3.80E-01	6.72E-01	9.26E+00
	Seawater	-	-	-	-	-	1.65E-04	3.45E-01	1.77E-01	2.18E-01	4.09E+00
Hydro-	Urban air	-	-	-	9.84E-02	1.59E-01	6.16E-04	2.46E-04	5.75E-01	-	1.86E-01

carbons, aliphatic, alkanes, cyclic	Rural air	-	-	-	9.84E-02	1.59E-01	2.58E-04	3.04E-04	7.31E-01	-	1.37E-02
	Fresh water	-	-	-	-	-	3.02E-01	5.52E-03	9.55E-02	-	2.73E-02
	Seawater	-	-	-	-	-	1.50E-06	1.71E-01	7.95E-03	-	2.18E-03
Hydro- carbons, aromatic	Urban air										
	Rural air										
	Fresh water Seawater										
Noble gases, radioactive, unspecified	Air	-	-	5.62E-06	-	-	-	-	-	-	-

* CF based on Anthracene only

CFs per substance group, impact category and sub-compartment as Hierarchist perspective (RIVM Report 2016-0104, p. 113-115)

Substance group	**(Sub)-compartment**	**Global warming** kg CO_2-eq	**Ozone depletion** kg CFC11-eq	**Ionizing radiation** kg Co60to air-eq	**Photo-Chemical ozone (human)** kg NO_X-eq	**Photo-Chemical ozone (eco)** kg NO_X-eq	**Freshwater ecotox** kg 1,4DCB to freshwater- eq	**Marine ecotox** kg 1,4DCB to saltwater- eq	**Terres-trial ecotox** kg 1,4DC to ind soil-eq	**Human tox (cancer)** kg 1,4DCB to urban air-eq	**Human tox (non-cancer)** kg 1,4DCB to urban air-eq
Aldehydes, unspecified	Urban air	-	-	-	1.74E-01	2.81E-01	2.28E-02	1.95E-02	7.30E+00	4.14E+01	3.36E+00
	Rural air	-	-	-	1.74E-01	2.81E-01	1.03E-02	2.91E-02	1.11E+01	1.35E+00	7.23E-01
	Fresh water	-	-	-	-	-	5.85E-01	1.94E-02	6.50E-01	8.16E-02	4.49E-01
	Seawater	-	-	-	-	-	9.66E-05	1.03E-01	6.87E-02	4.46E-03	2.42E-02
PAH, polycyclic aromatic hydro-carbons	Urban air	-	-	-	-	-	2.03E-01	8.56E+00	3.24E+01	3.38E+01	6.01E+00
	Rural air	-	-	-	-	-	3.45E-01	1.56E+01	6.15E+01	3.14E+01	2.48E-01
	Fresh water	-	-	-	-	-	1.30E+01	1.06E+00	3.34E+00	8.95E-01	1.05E+00
	Seawater	-	-	-	-	-	4.85E-03	1.10E+01	1.35E+00	6.84E-02	1.72E-01
	Industrial	-	-	-	-	-	6.00E+00	5.40E+00	3.81E+01	-	1.38E-02
	soil*										
	Agricultural	-	-	-	-	-	8.57E-01	5.03E+00	3.74E+01	-	4.16E-03

	soil*										
Actinides, unspecified	Air	-	-	1.58E+00	-	-	-	-	-	-	-
	Fresh water	-	-	3.14E-05	-	-	-	-	-	-	-
	Seawater	-	-	4.38E-04	-	-	-	-	-	-	-
Carboxylic	Urban air	-	-	-	4.63E-03	7.46E-03	2.19E-02	3.21E-03	5.47E+01	-	4.06E+02
acids	Rural air	-	-	-	4.63E-03	7.46E-03	6.39E-03	4.49E-03	8.18E+01	-	1.45E+01
	Fresh water	-	-	-	-	-	9.48E-02	1.33E-03	1.59E-03	-	1.52E-02
	Seawater	-	-	-	-	-	6.33E-10	9.41E-03	8.00E-06	-	3.52E-06
Hydro-	Urban air	1.38E	7.26E-	-	2.66E-02	4.29E-02	3.62E-04	5.27E-02	1.78E-01	2.27E+0	4.07E+01
carbons,		+01	03								
chlorinated	Rural air	1.38E+01	7.26E-03	-	2.66E-02	4.29E-02	3.65E-04	5.33E-02	1.80E-01	1.39E+0	2.38E+01
	Fresh water	+01	-	-	-	-	3.35E-01	6.34E-02	3.80E-01	9.59E-01	9.26E+00
	Seawater	-	-	-	-	-	1.65E-04	3.46E-01	1.78E-01	3.41E-01	4.09E+00
Hydro-	Urban air	-	-	-	9.84E-02	1.59E-01	6.16E-04	2.45E-04	5.75E-01	-	1.86E-01
carbons,	Rural air	-	-	-	9.84E-02	1.59E-01	2.58E-04	3.03E-04	7.31E-01	-	1.37E-02

aliphatic,	Fresh water	-	-	-	-	-	3.02E-01	5.53E-03	9.55E-02	-	2.73E-02
alkanes, cyclic	Seawater	-	-	-	-	-	1.50E-06	1.71E-01	7.95E-03	-	2.18E-03
Hydro- carbons, aromatic	Urban air				1.63E-01	2.63E-01	1.54E-05	3.88E-04	3.00E-02	1.72E-01	7.83E-01
	Rural air				1.63E-01	2.63E-01	1.86E-05	4.72E-04	3.66E-02	1.73E-02	6.57E-02
	Fresh water Seawater						1.55E-01	3.70E-03	3.87E-02	5.53E-02	1.04E-01
	Seawater						6.68E-06	1.06E-01	1.43E-02	2.34E-02	3.32E-02
Noble gases, radioactive, unspecified	Air	-	-	7.88E-06	-	-	-	-	-	-	-

* CF based on Anthracene only

CFs per substance group, impact category and sub-compartment as Egalitarian perspective (RIVM Report 2016-0104, p. 116-117)

Substance group	**(Sub)-compartment**	**Global warming** kg CO_2-eq	**Ozone depletion** kg CFC11-eq	**Ionizing radiation** kg Co60to air-eq	**Photo-Chemical ozone (human)** kg NO_X-eq	**Photo-Chemical ozone (eco)** kg NO_X-eq	**Freshwater ecotox** kg 1,4DCB to freshwater- eq	**Marine ecotox** kg 1,4DCB to saltwater- eq	**Terres-trial ecotox** kg 1,4DC to ind soil-eq	**Human tox (cancer)** kg 1,4DCB to urban air-eq	**Human tox (non-cancer)** kg 1,4DCB to urban air-eq
Aldehydes, unspecified	Urban air	-	-	-	1.74E-01	2.81E-01	2.28E-02	1.95E-02	7.30E+00	4.14E+01	3.36E+00
	Rural air	-	-	-	1.74E-01	2.81E-01	1.03E-02	2.91E-02	1.11E+01	1.35E+00	7.23E-01
	Fresh water	-	-	-	-	-	5.85E-01	1.94E-02	6.50E-01	8.26E-02	4.49E-01
	Seawater	-	-	-	-	-	9.66E-05	1.03E-01	6.87E-02	4.50E-03	2.42E-02
PAH, polycyclic aromatic hydro-carbons	Urban air	-	-	-	-	-	2.03E-01	8.56E+00	3.24E+01	3.38E+01	6.01E+00
	Rural air	-	-	-	-	-	3.45E-01	1.56E+01	6.15E+01	3.14E+01	2.48E-01
	Fresh water	-	-	-	-	-	1.30E+01	1.06E+00	3.34E+00	8.95E-01	1.05E+00
	Seawater	-	-	-	-	-	4.85E-03	1.10E+01	1.35E+00	6.84E-02	1.72E-01
	Industrial soil*	-	-	-	-	-	6.00E+00	5.40E+00	3.81E+01	-	1.38E-02
	Agricultural	-	-	-	-	-	8.57E-01	5.03E+00	3.74E+01	-	4.16E-03

	soil*										
Actinides, unspecified	Air	-	-	4.16E+00	-	-	-	-	-	-	-
	Fresh water	-	-	2.92E-03	-	-	-	-	-	-	-
	Seawater	-	-	4.61E-04	-	-	-	-	-	-	-
Carboxylic acids	Urban air	-	-	-	4.63E-03	7.46E-03	2.19E-02	3.21E-03	5.47E+01	-	4.06E+02
	Rural air	-	-	-	4.63E-03	7.46E-03	6.39E-03	4.49E-03	8.18E+01	-	1.45E+01
	Fresh water	-	-	-	-	-	9.48E-02	1.33E-03	1.59E-03	-	1.52E-02
	Seawater	-	-	-	-	-	6.33E-10	9.41E-03	8.00E-06	-	3.52E-06
Hydro-carbons, chlorinated	Urban air	1.97E+0	6.62E-03	-	2.66E-02	4.29E-02	3.62E-04	5.27E-02	1.78E-01	2.27E+0	4.07E+01
	Rural air	1.97E+00	6.62E-03	-	2.66E-02	4.29E-02	3.65E-04	5.33E-02	1.80E-01	1.39E+0	2.38E+01
	Fresh water	+01	-	-	-	-	3.35E-01	6.34E-02	3.80E-01	9.59E-01	9.26E+00
	Seawater	-	-	-	-	-	1.65E-04	3.46E-01	1.78E-01	3.41E-01	4.09E+00
Hydro-	Urban air	-	-	-	9.84E-02	1.59E-01	6.16E-04	2.45E-04	5.75E-01	-	1.86E-01

carbons,	Rural air	-	-	-	9.84E-02	1.59E-01	2.58E-04	3.03E-04	7.31E-01	-	1.37E-02
aliphatic,	Fresh water	-	-	-	-	-	3.02E-01	5.53E-03	9.55E-02	-	2.73E-02
alkanes, cyclic	Seawater	-	-	-	-	-	1.50E-06	1.71E-01	7.95E-03	-	2.18E-03
Hydro-	Urban air				1.63E-01	2.63E-01	1.54E-05	3.88E-04	3.00E-02	1.72E-01	7.83E-01
carbons,	Rural air				1.63E-01	2.63E-01	1.86E-05	4.72E-04	3.66E-02	1.73E-02	6.57E-02
aromatic	Fresh water Seawater						1.55E-01	3.70E-03	3.87E-02	5.53E-02	1.04E-01
	Seawater						6.68E-06	1.06E-01	1.43E-02	2.34E-02	3.32E-02
Noble gases, radioactive, unspecified	Air	-	-	7.88E-06	-	-	-	-	-	-	-