Monetarization – A Life Cycle Assessment weighting methodology

Monetize environmental impacts of paint with and without modified colloidal silica for AkzoNobel

Master’s thesis in Industrial Ecology

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Department of Energy and Environment
CHALMERS UNIVERSITY OF TECHNOLOGY
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ABSTRACT

It is today widely recognized that human activities give impacts on the environment. These could for instance result in depleted resources and ecosystems, which give a cost burden for present and future societies. An interest has recently arisen to convert these various environmental impacts to single environmental costs. This procedure is called monetarization. One company working explorative with this issue of economic assessment of environmental impacts is AkzoNobel, focusing its businesses on the main areas: decorative paints, performance coatings and specialty chemicals. This study was conducted in collaboration with AkzoNobel´s sustainability department, with the aim to explore the environmental costs of two decorative paints by utilizing monetarization, which is a weighting methodology within Life Cycle Assessment. Environmental impacts from paints are often regarded of high importance to mitigate and a fairly new strategy is to include modified colloidal silica in paint. This improves and gives new functionalities to the paint implying a prolonged life time. A comparison of two paints, one with and one without modified colloidal silica was performed to investigate the environmental benefits of colloidal silica in paint. The three monetary weighting methods EPS, Stepwise and Ecovalue were selected through a literature review and utilized to accomplish this comparison. Results showed that the expected longer durability of the paint with colloidal silica is a decisive factor for the environmental benefits. It was also identified lack of models to fully cover the environmental impacts from colloidal silica and similar materials. Further work therefore require investigations of these life time benefits but also better understanding of the environmental impacts from colloidal silica in the context of Life Cycle Assessment.

Key words:

colloidal silica, environmental cost, Life Cycle Assessment, monetary weighting, paint

Contents

Preface

This master´s thesis report (30 Credits) is written as the final part of the Chalmers program, Industrial Ecology (120 Credits). The project has been carried out from January to June year 2014 and was conducted at the sustainability department at AkzoNobel. The department provides sustainability and environmental services for strategic support internally in order to increase the value of their businesses. The project has been conducted in cooperation with two AkzoNobel business units, Decorative Paints (in Malmö) & Pulp and Performance Chemicals (in Bohus), which have formed a reference group for the project including Michael Persson (Innovation manager, PPC), Caterina Camerani (Sustainability specialist, PPC), Jonas Rothen (Marketing manager for Colloidal silica, PPC), Peter Greenwood (Business development specialist, PPC), Jenny Lundegård (Market manager innovation, Decorative Paints) and Margareta Melander (Manager Raw material and Product Safety/RD&I, Decorative Paints).

The master´s thesis project has been carried out with Jacob Lindberg as a researcher and Dr. Karin Andersson Halldén as supervisor, Johan Widheden as assistant supervisor and Prof. Anne-Marie Tillman as examiner. The project was examined in the *Department of Energy and Environment* at the *Division of Environmental System Analysis* on Chalmers University of Technology.

I would like to express my gratitude to the above mentioned people for their contribution which enabled the realization of this report, with a special thanks to Karin Andersson Halldén.

Göteborg June 2014

Jacob Lindberg

Abbreviations

LCA – Life Cycle Assessment

LCI – Life Cycle Inventory

LCIA – Life Cycle Impact Assessment

EPS[1] – Environmental Priority Strategies

[1] In subsection 3.4.4, the abbreviation EPS stands for Eco-premium solutions but everywhere else EPS means Environmental Priority Strategies

1 Introduction

Every year, millions of people world-wide paint their houses in order to protect the surface and to give the house a refreshing new finish. This creates values, for instance prolonged life time of the facade and an aesthetic value. One drawback is however the rise of environmental impacts in the life cycle of paint, from raw material extraction to final waste management. An interest has arisen in the recent time to convert these environmental impacts to external costs, called environmental costs. These can indicate a future cost burden that needs to be paid by the actors in a product´s value chain for instance through taxes or other policy instruments. It would thereby be of interest to study these environmental costs in collaboration with the paint producer AkzoNobel.

This chapter is divided into four sections. The first section intends to give a background and problemized picture of the research problem, environmental costs for paint products. In the second and in the third section, the purpose and the delimitations of the report are stated. The chapter is finalized by an outline of the remaining parts of the report.

1.1 Background

AkzoNobel is the largest paint producer in the world according to Kougoulis et al. (2012) and one of the industry leaders in the Dow Jones Sustainability Indices (Guarino and Vetri, 2013). This measures the sustainability performances and investments regarding the three dimensions of sustainable development: social, economical and environmental (Guarino and Vetri, 2013; Pawłowski, 2008). The demand for products to be sustainable has grown in the recent years, stated by Finnveden et al. (2009) and the industry is starting to apply to these demands (KPMG International, 2012). The development and inclusion of such products in a company's product portfolio is essential for long term competitiveness according to Guarino and Vetri (2013). Sustainability is business and business is sustainability (AkzoNobel, 2013b).

Providing the society with goods and services contribute to economic growth but simultaneously to a wide variety of environmental impacts (Pennington et al., 2004). These comprise for instance of impacts on humans, resources and the natural environment (Bauman and Tillman, 2004). The impacts can lead to present and future costs that the society sooner or later has to bear (Itsubo et al., 2004; KPMG International, 2012). The interest would thereby be evident to account for these external effects, by internalizing the external environmental costs. This is to make the price of the product reflect the ecological truth (European Commission, 2005). Hence, in addition to the present product price add the economic value of the damages caused by the product in terms of environmental impacts (Sleeswijk, Bijleveld and Sevenster, 2010). This is however challenging to implement in reality and depends largely on the level of the damage cost, possibility to transfer costs to customers and type of firm (Steen, Lindblad and Palander, 2014). A study by KPMG International (2012) computed the environmental costs incurred per dollar of earnings for a wide range of industry sectors globally. For the chemical industry the environmental costs were 43 cents per dollar of earnings, which was representative for an average of all included industries in the study. These environmental costs are also expected to double every 14th year, due to for instance population growth and increased wealth. It is difficult to make 100 % accurate estimations of these costs, but it could rather acts as an indicator of a future environmental bill (KPMG International, 2012).

As future business performance will be affected by depleted ecosystems and resources according to KPMG International (2012), the pressure increases on companies to account for environmental costs. To recognize the environmental costs in the business can provide opportunities for better informed decisions regarding for instance investment risks (KPMG International, 2012). By reporting environmental costs to stakeholders, environmental impacts can be illustrated in a more comprehensive and convenient way (Steen and Lindblad, 2014). KPMG International (2012) claims that companies are more prone and motivated to act on sustainability when environmental costs can be shown in financial statements. Companies can thereby position themselves for the future and provide their customers more sustainable products (Itsubo et al., 2012). Such products are often more expensive and by accounting for and implementing environmental costs the more sustainable products could be competitive on the market (De Camillis and Goralczyk, 2013).

Translating environmental impacts to environmental costs is relatively new in practice, but starting to gain momentum (KPMG International, 2012). The idea of internalize external environmental costs is first to measure damages to the society, not paid by the main actors in the value chain. Secondly, to convert these damages to monetary values and thirdly to explore how these external environmental costs can be charged to the producers and consumers (European Commission, 2005). There is however a need for tools and methods to deal with this issue of economic assessment of environmental impacts (Pennington et al., 2004). The science encompasses this is environmental system analysis, where several methods can handle this issue (Finnveden et al., 2009). One method is Life Cycle Assessment (LCA) which identifies and quantifies the potential environmental impacts and resources used in the life cycle of a product (ISO, 2006a). These various impacts can be aggregated and converted to monetary values by using monetary weighting methods. This procedure is called monetarization. Many different weighting methods exist and their common aim is to compute the overall environmental impact of for instance a product, expressed in a single number (Huppes and van Oers, 2011a). By doing this the internal relation of the severity between the impacts are assessed, expressed in a format simplifying comparison between product alternatives (Bengtsson, 2000a; Ahlroth et al., 2011). To create a weighting method that translate environmental impacts to environmental costs in an accurate way has shown to be a challenging task (Bengtsson, 2000a; Finnveden, 1997).

In the paint industry, environmental issues are often regarded as high importance for instance since many paint ingredients are environmental harmful and toxic to humans (Kougoulis et al., 2012; Overbeek et al., 2003). One strategy aiming to lower these effects is to add colloidal silica as a raw material in the paint formulation, which could prolong the life time of the paint (AkzoNobel, 2013c). However, environmental impacts create external effects and by assessing this effects an indication of future environmental costs can be given if for example more stringent policy measures are implemented (Tekie and Lindblad, 2013; WBCSD, 2011; Itsubo et.al., 2004). It would thereby be of interest to explore and assess these external environmental costs of paint products, in a study, in order to gain knowledge of them.

1.2 Purpose

The aim of this report is to explore the environmental costs of two decorative paints by utilizing monetarization, which is a weighting methodology within Life Cycle Assessment.

From this, three goals are developed for the report.

A first goal is to gain knowledge about different monetary weighting methods, through a literature review, and determine suitable weighting methods for the case study of paint products.

The second goal is to compute the environmental costs for two outdoor decorative paints, one including modified colloidal silica and one without, in a case study. This will be performed in cradle-to-grave LCAs including monetary weighting procedures.

The third goal is to compare the different life cycle phases regarding their environmental costs, but also to put environmental costs in relation to real costs.

1.3 Delimitations

This section presents the general delimitations for the report. Case study specific delimitations can be found in Section 4.3.

- External costs can be generated by both environmental impacts and social impacts. In this report, only external costs arising from environmental impacts are studied, called environmental costs.
- There are several environmental system analysis methods available for calculation of environmental costs. The method utilized in this report is Life Cycle Assessment where weighting methods are used to monetize the environmental impacts.
- Environmental costs can be implemented in several different ways. How these costs could or should be implemented is not investigated in the report.

1.4 Outline of the report

The chapters that this report includes are stated and briefly presented below.

Chapter 2 *Method*

Chapter 3 *Literature review*

Chapter 4 *Case study*

Chapter 5 *Discussion*

Chapter 6 *Conclusions*

Chapter 7 *Further work*

The methods used in the project are presented in Chapter 2 and describe the working process approach and the data used. Chapter 3 is built up by information gathered from a literature review intending to describe the main areas in this report, Life Cycle Assessment and Paint. This chapter is an important building block to fulfill the purpose of the project and acts as a basis for the case study, presented in Chapter 4. The case study aims to evaluate the environmental costs and impacts from the products studied. Subsequent discussion, conclusions and further work for the report are presented in Chapter 5, Chapter 6 and Chapter 7 respectively.

2 Method

This chapter covers the methods and the data used to conduct the project. The chapter is divided into two parts. First, the procedure of the project is described; thereafter the collection of data is presented.

2.1 Project procedure

Regardless of the coverage of a study, an ideal research process can be described with some exceptions from Patel and Davidson (2003) as:

1. Identification of the problem area
2. Identification of purpose and questions
3. Literature review
4. Clarification of the problem
5. Case study
6. Processing and analysis
7. Reporting

In the implementation of this project, the above structure was used as guidance for the working process. Most of the above points are overlapping and could not be worked through in a sequence (Patel and Davidson, 2003); instead an iterative procedure was used as new information arose. In the study, identification of new information was a strong reason for revision of the work during the project process and iterations between the above points were conducted.

The working process for this project can in general be divided into two parts. The first part consisted of the identification & clarification of the research problem and planning the implementation of the case study. This was conducted mainly through a literature review but also through communication with the parties involved in the study, i.e. the reference group and supervisors. The basis for the project was thereby created, including point one to four in the above list. The second part consisted of the case study and its subsequent process and analysis, point five and six. In this part the environmental impacts and costs were examined and analysed in a case study, where the Life Cycle Assessment software tool GaBi6 was utilized. Both project parts did include reporting, resulting in this report. By using this procedure, both parts enabled fulfillment of the aim of the study.

2.2 Collection of data

According to Patel and Davidson (2003) there are mainly two types of data, primary and secondary. Primary data is collected from unprinted sources, for instance interviews or other face-to-face communications. Secondary data refers to data gathered from documented sources, e.g. books or articles. In addition there exist surrogate data, consisting of e.g. estimations and assumptions (Baumann & Tillman, 2004). In the study, primary-, secondary- and surrogate data was collected during the entire project period. This was processed and analysed in accordance with the project´s aim and delimitations, which steered the focus towards its core area.

The data collected can be divided between the two parts of the project, due to their different nature. In the first part, data was collected with the purpose to gain understanding of the main topics, Life Cycle Assessment and Paints. These were supposed to act as a basis for the case study, clarifying concepts and theories. In the data collection, a funnel model has been used where general literature was first reviewed where after specific literature could be studied (Nyberg, 2000). Literature used comprised of scientific articles, books, theses, various reports and internet sources. In some cases several sources have been used to verify information. The literature review was conducted mainly through electronic data collection, which enabled a broad collection of data. Search words were used for searches in databases and search engines which facilitated that sought-after information could be found. Frequently used search words were *environmental issues of paint*, *colloidal silica*, *environmental costs* and *Life Cycle Assessment*. Apart from the search words, a large amount of literature could be reviewed as only summaries, table of contents, tables and figures were studied in the documents. Thereafter an examination if the source was of interest for the report could be made. The literature review includes both English and Swedish literature, where English literature has been used as much as possible. Apart from the literature review, data has also been collected from the project´s reference group to form the content of the case study. This data composed mainly of estimations about aspects included in the case study.

During the second part of the project, the data collection was focused on the case study. This included data necessary for the implementation of the Life Cycle Assessment in the case study. The data was mainly collected within AkzoNobel and in some cases estimations were necessary where data was lacking. The data collected in this part of the project is mainly confidential, in contrast to the first part where public data was used. This had the implementation that the report is published in two versions, one confidential and one public report. The modelling of the data collected from the second part was conducted in GaBi6, where AkzoNobel database models where used. Data from the weighting methods were in some cases already included in GaBi6, while in other cases data needed to be entered. In the case study in Chapter 4 the specific methodological choices are stated regarding the modelled Life Cycle Assessment.

3 Literature review

This chapter contains the project´s literature review, which include the two main areas: Life Cycle Assessment (LCA) and Paint. The chapter acts as basis for the case study, presented in Chapter 4, aiming to clarify concepts and theories in the stated areas.

The chapter is divided into four parts consisting of: *Life Cycle Assessment (LCA) methodology, Weighting methodology within LCA, Paint products and painting,* and *Paint from an environmental perspective.*

3.1 Life Cycle Assessment (LCA) methodology

This section presents the environmental system analysis method, Life Cycle Assessment (LCA). The method assesses the environmental impacts throughout a product or a service life cycle. LCA is utilized as the method for accounting the environmental impacts and environmental costs occurring for the products included in the case study presented in Chapter 4.

3.1.1 Introducing the method

LCA is according to Bengtsson (1998) a quantitative review of the potential environmental impacts and resources used in the life cycle of a product, where the term `product´ includes both goods and services (ISO, 2006a). A product´s life cycle starts at the raw material extraction, via processing, manufacturing, distribution, use to waste management, i.e. cradle-to-grave (ISO, 2006a). Thereafter the product returns to nature or takes part in other products (Bengtsson, 1998). The overall environmental impacts that are recommended to be considered in an LCA are natural environment, human health and resources (ISO, 2006a), where man-made environment sometimes is suggested (Pennington et al., 2004). These four are usually referred to as `areas of protection´ or `safeguard subjects´, which represent what humans want to protect and preserve (Rebitzer et al., 2004; Finnveden et al., 2009).

In Baumann and Tillman (2004), three application areas for LCA are suggested: decision making, learning and communication. Performing an LCA has often the purpose to compare different products from an environmental viewpoint. This is either accomplished for an existing system or a comparison between the existing system and proposed changes of the system (Bengtsson, 1998; Rebitzer et al., 2004). As an LCA study maps out the environmental impacts occurring in the life cycle, it is possible to analyse the impacts in order to reduce them where they are most critical (Finnveden et al., 2009).

LCA is internationally standardized in accordance with the ISO 14040/14044 (ISO, 2006a; ISO, 2006b). Besides ISO, there exist a number of acknowledged guidelines (European Commission, 2010; Guinée et al., 2002), textbooks (Wenzel, Hauschild and Alting, 1997; Baumann and Tillman, 2004) and review papers (Finnveden et al., 2009; Pennington et al., 2004) on LCA. The LCA procedure is divided into four different interacting and iterative phases. These are *goal and scope definition*, *inventory analysis*, *impact assessment* and *interpretation*. A representation of their connection can be found in Figure 1 below.

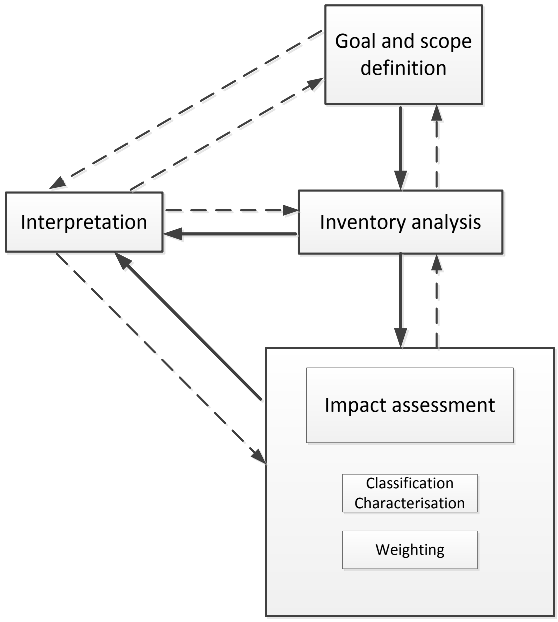

Figure 1 – Phases in Life Cycle Assessment according to Baumann and Tillman (2004).

The boxes in the above figure represent the procedural phases. The arrows represent in which order these are performed and broken arrows indicate that possible iterations are required (Baumann and Tillman, 2004). This is a common life cycle model and similar to the ISO model (ISO, 2006a). Part of the strength in LCA lies within this kind of standardized models and the harmonized language developed (Steen, 1999a). The following subsections will elaborate upon these procedural phases, agreed upon by the LCA community (Rebitzer et al., 2004; Finnveden et al., 2009).

3.1.2 Goal & scope definition

The first part of an LCA is the determination of the goal & scope for the study (ISO, 2006a). In the goal definition, application, purpose and objectives for the study are included (Bengtsson, 1998). This can be represented by answering the questions: What is studied?, Why is the study conducted?, Who are the intended audience? and Which questions are to be answered in the study? (Hildenbrand, 2013). By defining this, the context of the study is set.

The scope definition contains the modelling aspects of the study. The aspects considered in the case study presented in Chapter 4 are described below. The scope defines what to include in the study and what to exclude from the study. A model for the technical system can thereby be created, defining the activities related to each product in the study (Bengtsson, 1998). The modelling aspects must be carefully considered since the choices and the assumptions made are often decisive for the results of the LCA study (Rebitzer et al., 2004).

Functional unit

The functional unit is the basis that enables comparison between product alternatives, stated in Rebitzer et al. (2004), and is the key element for an LCA (DANTES, 2006). This should include a quantified description of the functions provided by the products included in the study (Rebitzer et al., 2004; Weidema et al., 2004; Finnveden et al., 2009). The functional unit is related to a reference flow in the life cycle, which all other flows modelled in the system are related to (Baumann and Tillman, 2004). An example of an appropriate functional unit for paint systems could according to DANTES (2006) be defined as the unit surface area protected for 10 years. Comparison of the environmental impacts for the paint systems can thereby be made based on this functional unit.

Choice of impact assessment

This modelling aspect should reveal which environmental impacts that are considered in the study and how the results are intended to be presented (Baumann and Tillman, 2004). There are many types of environmental impacts that can be considered in an LCA, where different scientific models can be used when calculating the impacts (Pennington et al., 2004). The impacts have usually their origin from the aggregated impact categories found in `areas of protection´. The results from an LCA can be presented in different formats, as inventory data, characteristic results or as weighted one-dimensional index (Baumann and Tillman, 2004).

Type of LCA

There are basically two types of LCAs, attributional and consequential, where the difference between them origins from the goal of the study stated by Rebitzer et al. (2004) and Finnveden et al. (2009). An attributional study has the goal to describe the product system and its relevant environmental flows. A consequential study has the goal to describe how environmental relevant flows will change in response to actions taken in the study (Rebitzer et al., 2004; Finnveden et al., 2009). These can be referred to as descriptive or change-orientated, respectively. Depending on which one is chosen, it entails important consequences for how the system should be modelled, which affect the results (Lindfors et al., 2012). The opinion is divided for which situations the different variants shall be used (Finnveden et al., 2009). According to European Commission (2010) attributional LCA is in most cases recommended.

System boundaries

The system boundaries separate the parts considered and not considered in the study according to Finnveden et al. (2009). Three types of boundaries are usually identified: (1) between the technical system and the environment, (2) between significant and insignificant processes and (3) between technical systems under study and other technical systems. Time and geographical limits can also be included (Finnveden et al., 2009).

Allocations

If there are ambiguities in how environmental impacts shall be partitioned between different product systems, there is a need to allocate the impacts. In the development of LCA, this has been one of the most controversial issues (Rebitzer et al., 2004). According to the ISO 14040/14044 it is recommended to avoid allocation when it is possible, either by expanding the system or through subdivision of processes. If allocation is not possible to avoid, it is recommended to allocate the impacts based on physical relationships, e.g. mass or energy content or allocation by the economic values of the products (ISO, 2006a; ISO, 2006b).

Flowchart for the LCA

The flowchart describes the life cycle product system. This includes the activities (e.g. manufacturing, transportation, waste management) and flows between them (Baumann and Tillman, 2004).

Scenarios

Scenarios can describe future modelling of the product system and are relevant in many applications according to Finnveden et al. (2009). This can be used in both consequential and attributional LCAs, aiming to assess future systems. A decision must be made for how to model the future system. One easy solution is to assume that the future is like the present, which may sometimes be a good assumption (Finnveden et al., 2009).

Data quality requirements

Depending on the data utilized, this will affect the study´s relevance, reliability and accessibility. These three factors can be further subdivided. Relevance of the data is about if the data represents what it is supposed to represent. The different aspects of relevance can for instance be time-related, geographical- & technology coverage and the completeness of the data. Reliability deals with the precision of the data and accessibility with the ability to review and reuse the data (Baumann and Tillman, 2004).

Assumptions and delimitations

Major assumptions and delimitation in the study should be stated in the goal and scope definition (Baumann and Tillman, 2004).

3.1.3 Life Cycle Inventory (LCI)

The subsequent phase is the Life Cycle Inventory (LCI). In the LCI, a flow model is constructed in accordance with the system boundaries decided upon in the goal and scope definition (Rebitzer et al., 2004). This is represented in a flowchart and includes all processes relevant for the LCA, e.g. transportation and manufacturing. In connection to the processes, the flowchart include input flows e.g. raw materials & energy, output flows e.g. emissions & waste and flows linking the processes e.g. the refined product. The linked process flows take place in the technical system, while input and output flows cross the boundaries between the technical system and the environment (Rebitzer et al., 2004; Ahlroth et al., 2011). Input flows are materials drawn from the environment to the technical system without previous human transformation. Output flows are material released from the technical system to the environment without subsequent human transformation (Finnveden et al., 2009). These input and output flows are called environmental interventions or environmental flows and are the link between economic activities and environmental impacts (Huppes and van Oers, 2011a; Dong, Laurent and Hauschild, 2013). This can schematically be presented in the below Figure 2.

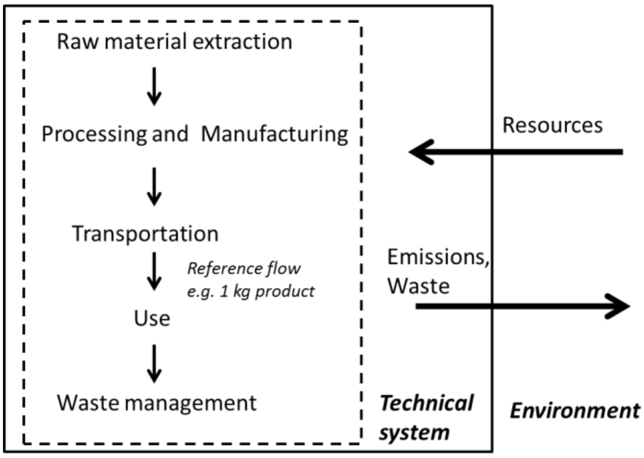

Figure 2 – Schematic illustration of the Life Cycle Inventory model, based on Baumann and Tillman (2004).

An important part in the construction of the flow model is collection of data for all the activities in a product´s life cycle, both from processes and flows (Baumann and Tillman, 2004). When data is collected, it needs to be related to the functional unit (Pennington et al., 2004). This will enable a comparison between the products in the study. The result from this life cycle phase forms the inventory of the LCA acting as the basis for the assessment of the environmental impacts (Baumann and Tillman, 2004).

The LCI phase is often the most time and labour intensive stage in an LCA (Finnveden et al., 2009; Rebitzer et al., 2004). This is often due to lack of data for the product system under study and it is often better to refer to searching for data than collection of data (Baumann and Tillman, 2004). In order to cope with this challenge, many databases have been developed in the recent years e.g. national and industry databases (Finnveden et al., 2009). When using this ready-made data, one needs to be aware of the quality and reliability of such data. Aspects of this could for instance be the age of the data, data source (primary, secondary, surrogate), location and the technology of the processes (Baumann and Tillman, 2004).

3.1.4 Life Cycle Impact Assessment (LCIA)

The aim of the Life Cycle Impact Assessment (LCIA) is to understand and evaluate the potential environmental impacts including the magnitude and significance. This is to provide information to help assessing the results from the LCI (Huppes and van Oers, 2011a; Rebitzer et al., 2004). This gives according to Baumann and Tillman (2004) more environmental relevant and easier comparable results. Environmental impacts can be derived from the environmental interventions. These are usually modelled in cause-effect chains (see Figure 4 below) until environmental impacts are reached (Huppes and van Oers, 2011a; Rebitzer et al., 2004).

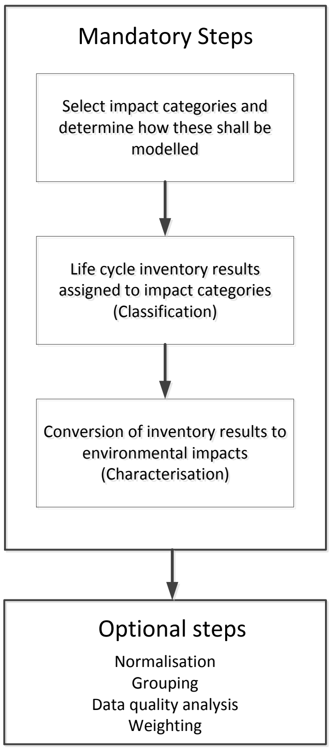

Figure 3 – Mandatory and optional steps in LCA, based on Pennington et al. (2004).

According to for example Baumann and Tillman (2004) and Pennington et al. (2004) the procedure of the LCIA starts with a number of mandatory steps, followed by the optional steps. These steps are presented in Figure 3 above. The first mandatory step is the selection of impact categories relevant for the study and a description of how these should be modelled. There are in general two types of impact categories: at midpoint level (e.g. eutrophication potential) or at endpoint level (e.g. reduced crop growth). Midpoint impact categories represent potential impacts somewhere in (but before the end of) the cause-effect chain, while endpoint impact categories represent damages at the areas of protection. In the second mandatory step, the environmental interventions collected in the LCI are classified and assigned to the respective impact categories they contribute to, called classification. Thereby, the often large number of inventory parameters can be reduced in aggregated impact categories. Characterisation is the third and last mandatory step in LCIA. The LCI data is converted to environmental impacts by using characterisation models determined upon in the first step. This is often represented by a number of different impact categories for which each of them sum their impacts in a common unit called category unit (e.g. CO_2-equivalents for the impact category global warming). In general the environmental impact for each impact category can be calculated according to equation (1) below. These three steps comprise the mandatory part of the LCA and are often referred to as objective, based on natural science (Baumann and Tillman, 2004; Pennington et al., 2004; Finnveden et al., 2009).

$$\text{Category Indicator} = \sum_s \text{Characterisation factor(s)} * \text{Inventory data} \quad (1)$$

The inventory data is generated during the LCI and the characterisation factors represent the environmental impact per unit of inventory data. The subscript *s* in the equation represents the specific inventory data, e.g. the emission or resource used (Pennington et al., 2004).

The optional part of LCIA comprise of four steps according to Baumann and Tillman (2004), which can normally be conducted independent of each other. One is called normalisation, where the impacts are related to a reference value, e.g. impacts in a region. Another step is grouping, where the impact categories are qualitatively ranked or grouped with respect to their importance. Data quality analysis is the third and is an evaluation of the reliability of the impact assessment results (Baumann and Tillman, 2004). This is accomplished by identifying major contributors, uncertainties and sensitivities. The last step is named weighting where the different environmental impacts are weighted against each other by using quantitative factors (Baumann and Tillman, 2004; Pennington et al., 2004). By using weighting, an overall indication of the environmental impacts can be calculated revealed in a single number (Huppes and van Oers, 2011a). Whether these optional steps are conducted depend on the goal and scope of the study (Rebitzer et al., 2004). In the case study presented in Chapter 4, weighting will be utilized together with the mandatory elements in LCA. A more elaborate description of the weighting procedure can be found in Section 3.2.

In order to simplify for the LCA practitioner, different ready-made LCIA methods have been created according to Baumann and Tillman (2004). In these methods, mandatory and sometimes optional steps are included. It is thereby possible to insert the results from the LCI and get the LCIA results in a convenient way. Different methods often contain different environmental impacts. They also differ with respect to how the LCIA results are presented, e.g. in characterisation data or as weighted single score index. Changes in the ready-made LCIA methods can however be possible in order to reflect for instance values in a company (Baumann and Tillman, 2004).

As the purpose of LCIA is to understand and evaluate the environmental impacts, it is important to stress that impacts depend on several factors (Steen, 1999a). This can include the quantity of the environmental intervention, properties of the intervention, the characteristics of the emitting source & the receiving environment (Baumann and Tillman, 2004; Steen, 1999a) and when the intervention occurs (Jackson and Jackson, 2011). The link between environmental interventions and the environmental impacts can be modelled through cause-effect chains, where the complexity of such chains is illustrated in Figure 4 below. Several orders of effects and impacts can be found as well as feedback effects in the system (Baumann and Tillman, 2004).

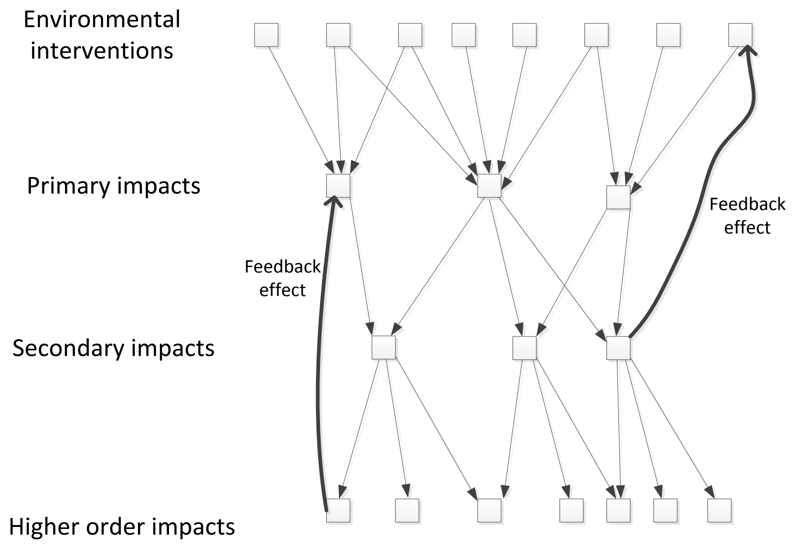

Figure 4 – Schematic view of cause-effect chains, from environmental interventions to environmental impacts, based on Baumann and Tillman (2004).

An example of such cause-effect chain presented in Jackson and Jackson (2011) and Dong, Laurent and Hauschild (2013) describe the environmental impact, global warming. The cause-effect chain starts as substances that absorb infrared radiation are released to air, e.g. carbon dioxide and methane. This change the balance between the energy that the earth absorbs and that it releases. A primary effect is thereby reached called radiative forcing. The radiative forcing is expected to change the global temperature, referred to as a secondary effect. This effect can give rise to e.g. ice melting, changed weather patterns, infectious diseases, which can lead to for instance societal and environmental damages. Positive and negative feedback effects could also be present along the cause-effect chains, which increase or decrease the effects (Jackson and Jackson, 2011; Dong, Laurent and Hauschild, 2013).

As the environmental impacts are often derived from cause-effect chains, the result can be presented in a dose-response model (European Commission, 2005) shown in Figure 5 below. The output from such a model represents the characterisation factor for each environmental intervention related to a certain impact category (Pennington et al., 2004). These factors are often simplified as linear due to the difficulty to model environmental impacts (Baumann and Tillman, 2004).

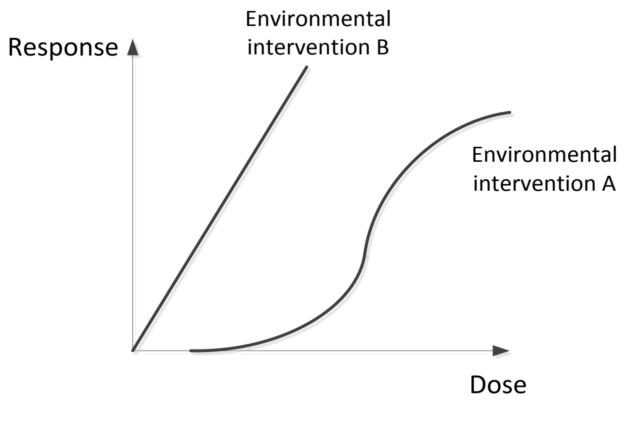

Figure 5 – Dose-response curve based on European Commission (2005).

Two different hypothetical environmental interventions are presented in the figure above. The amount of the environmental intervention (dose) is related to the environmental impact (response) in a specific relation (e.g. linear or non-linear). Knowledge about the cause-effect chains is critical in order to create models for the environmental impacts. It is however important to stress that such models can only describe the effects, not valuate them (Bengtsson, 1998).

3.1.5 Interpretation

The interpretation is where the results from previous phases are evaluated in relation to the goal and scope. This is to reach conclusions, explain limitations and provide recommendations consistent with the goal and scope (ISO, 2006a). The following shall be included in the interpretation according to Baumann and Tillman (2004) and ISO (2006a):

- Identification of significant issues based on results from the LCI and LCIA phases and methodological choices.
- Completeness, sensitivity and consistency check are to be evaluated for the results.
- Conclusions, limitations, recommendations and reporting.

When interpreting the results from an LCA one needs to understand the accuracy of them (second bullet), ensuring they meet the goal and scope of the study. This is accomplished by identifying significant issues from the results and the methodological choices (first bullet). After understanding how the LCA was performed and results were developed; conclusion, limitations, recommendation and reporting can be conducted (third bullet).

3.2 Weighting methodology within LCA

Weighting is one part of Life Cycle Impact Assessment (LCIA) and often discussed whether it should be utilized in LCIA, for instance due to its subjective nature. This section starts by giving a general overview of the principles of weighting and an introduction to monetary weighting, monetarization. After that, a review of arguments for and against weighting is presented. The section is finalized by reviewing five monetary weighting methods including selection of methods to be utilized in the case study in Chapter 4.

3.2.1 From weighting to monetarization

The ideal situation in an LCA is that the result is unequivocal, telling that one alternative is better than the others for all included environmental aspects of the study. This is however seldom the case (Bengtsson, 1998; Sleeswijk, Bijleveld and Sevenster, 2010). Normally it arises a trade-off between different environmental disturbances and it becomes more complicated to state which alternative to prefer from an overall environmental perspective (Bengtsson, 1998; Bengtsson, 2000b; Johansson, 1999). Another similar situation is when a change in a system has occurred, where some impacts are worse and some are lower from an environmental point of view. It is difficult to determine whether an improvement of the environmental performance has occurred or not (Bengtsson, 2000a). Since environmental impacts have different units they cannot be added directly, why it is necessary to convert them to a common unit. An overview of the total environmental impact could thereby be given (European Commission, 2005), which can simplify the comparison between alternatives but also between different environmental impacts (Finnveden, Håkansson and Noring, 2013). This can be done through weighting.

Weighting expresses the internal relation of the severity or importance for different types of environmental impacts or damages into one single number (Bengtsson, 2000b; Huppes and van Oers, 2011a). In common language this could be referred to as level of environmental friendliness (Steen, 1999a). In order to compare the overall environmental effects between studies and to make sure that weighted values are stated explicitly (Bengtsson, 1998; Johansson, 1999), weighting has been standardized in accordance with ISO 14040/14044 (ISO, 2006a; ISO, 2006b). Weighting is an optional part of LCA stated in Baumann and Tillman (2004) and is expressed as, the process of converting indicator results by using numerical factors based on value choices (ISO, 2006b). Bengtsson (1998) suggest a model for weighting which is called "fullt realiserad viktningsmetod" (full realized weighting method), where a schematic view can be seen in Figure 6 below.

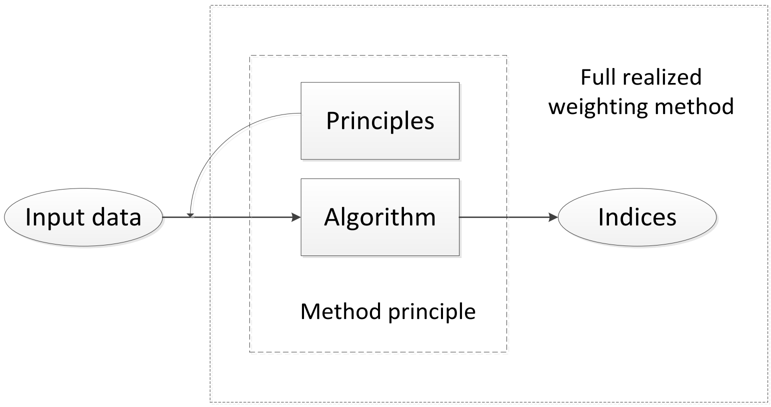

Figure 6 – Components in a weighting method, based on Bengtsson (1998).

According to Bengtsson (1998), there is a request for principles (e.g. to weigh environmental impacts using political targets) telling which kind of input data (e.g. Swedish "emission to air" data from year 2013) that should be the basis for the weighting. There is a request for an algorithm telling how these input data should be transformed to weighting indices. The principles steer what kind of input data that should be the basis for the weighting, while the algorithm describes how the data is transformed to indices. This results in a series of weighting indices, where each index states the weight given for an emission, impact or damage. The principles and the algorithms for a specific weighting method can be used over time and for different geographical regions. The indices are however connected to a specific set of input data and are therefore bound to a specific time and place. It is therefore important with regular updates of the indices according to Bengtsson (1998), both since people´s preferences and knowledge about environmental relationships change with time (Finnveden, Håkansson and Noring, 2013). A total environmental impact can be calculated based on the indices and the environmental load, as can be seen in equation (2) below (Baumann and Tillman, 2004).

$$\text{Total environmental impact} = \sum_{s} \text{Indices} * \text{Environmental loads} \qquad (2)$$

Different weighting principles are based on different viewpoints, but have one thing in common. They are not based on natural science, which separate the classification & characterisation from weighting in LCIA (Finnveden, 1997; Bengtsson, 2000b). Weighting can require political, ideological and ethical values (Finnveden, 1997; Sleeswijk, Bijleveld and Sevenster, 2010). There are however no societal consensus about these issues and will probably never be, why weighting has been questioned at all levels: if weighting should be used at all to which weighting factors to be used. Quite a few different weighting methods have evolved from different principles (Finnveden, 1997). Bengtsson (1998) summarize some principles used in weighting methods, which can be viewed in Table 1 below.

Table 1 – Common principles in weighting methods, based on Bengtsson (1998).

Different principles used in weighting methods	**Examples**
Political decisions	• Environmental distant to target (general reduction targets, general environmental quality target) • Other targets by authorities (cleaning requirement; actions to avoid or reset damage)
Techno-economic conditions	• Energy use • Cleaning cost • Land use demand • Material movement demand
Natural conditions	• Critical assimilation capacity • Concentration where no affect can be measured • Background concentration • Natural flow • Decomposition time • Resources (e.g. average concentration in Earth crust)
Health effects	• Relative contribution on human health
Panels	• Willingness to pay by individuals (answer on a direct question i.e. contingent valuation) • Other panels composing of scientists, politicians, NGOs, cross-section of the population and their procedure: Delphi, structured dialogue, survey, negotiation
Behavioral studies	• Individuals revealed willingness to pay including e.g. changes in market price "hedonic pricing", willing to sacrifice (e.g. travel time and travel costs), resources´ market prices
A combination	

Included in the principles are sources for the principles: by what weighting is based on (e.g. techno-economic conditions, natural conditions), how the values are derived (e.g. directly by questions or passively via studying decision making) and by whom they are stated (e.g. politicians, scientists, authorities, cross-section of the population) (Bengtsson, 2000a). A weighting method often comprises a combination of different principles.

For the result of the indices from the above model, a division can be made based on if the weights are expressed in monetary (i.e. in a currency) or in non-monetary units (e.g. dimensionless). For monetary weighting methods, the monetary values can be derived in different ways according to Ahlroth et al. (2011). This depends on the type of economic value that the environmental good, service or other entity that we want or protect have. In environmental economics a distinction is first made between use values and non-use values. Use values are either direct (e.g. timber value of the forest) or indirect (e.g. value of carbon fixation of a forest which gives a functional benefit), while non-use values represent peoples attached value to an amenity. Environmental goods and services do often lack market prices because they are not sold and bought on the market, why it is difficult to put monetary values on them (Ahlroth et al., 2011; Sleeswijk, Bijleveld and Sevenster, 2010). The valuation of these differs in different parts of the world and over time (European Commission, 2005). It is however of crucial importance for our economy and our existence that ecological systems are functioning to provide us with environmental goods and services according to Johansson (1999) and not a matter of philanthropy (Corporate eco forum & the Nature Conservancy, 2012). Especially the services are often by nature free to the public and the ownership is difficult to define. The question is thereby, how should the values be set and by whom? (Johansson, 1999).

There are a number of economic valuation methods, apart from the market value, for which the values of environmental assets can be derived. These include revealed- and stated preference methods. Reveal preference methods use information from related markets while in stated preference methods people are asked to make hypothetical choices in hypothetical markets (Ahlroth and Finnveden, 2011). Examples of such methods could be revealed-, stated-, imputed- or political willingness to pay (Ahlroth et al., 2011). Willingness to pay reveals someone´s attitude (in monetary terms) towards a change of a value i.e. the willingness to pay to preserve a value (Steen, 1999a; Johansson, 1999). A monetary weight could thereby be assigned to the underlying interventions threatening the value. A variant of this is willingness to accept of losing a value, which asks how much someone wants to be paid in order to cope without the value (Johansson, 1999). Another way to derive these economic values is to estimate what it would cost to attain or retain a value at a certain level e.g. through emission limits. This is called avoidance cost or restoration cost (Steen, 1999a; Ahlroth et al., 2011)

For the question of whose preferences that should be used, it depends generally on the specific study. Huppes and van Oers (2011a) makes a first distinction on whether the preferences are collective or individual. Further, the preferences can be derived from various parts and levels of the society, e.g. scientists, politicians, cross-section of the population (Bengtsson, 1998). Itsubo et al. (2012) stress the importance of representativeness that includes the whole population.

Human activities resulting in environmental interventions and subsequent impacts and damages can create economic consequences called environmental (damage) costs that the society has to bear. Examples of damages can be reduced crop growth or increased hospitalization due to illness. Simultaneously the reason for the interventions is increased economic welfare (Sterner and Coria, 2012; European Commission, 2005). Thus, there is a trade-off between interventions to accept and costs to reduce these interventions. An optimal level could be found in the schematic Figure 7 and Figure 8 below (Sterner and Coria, 2012).

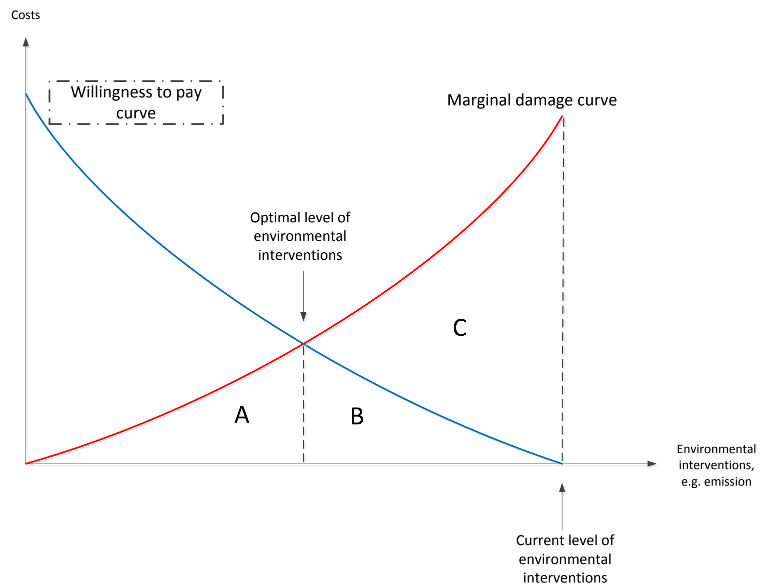

Figure 7 – Relation between willingness to pay and marginal environmental damage, based on Sterner and Coria (2012).

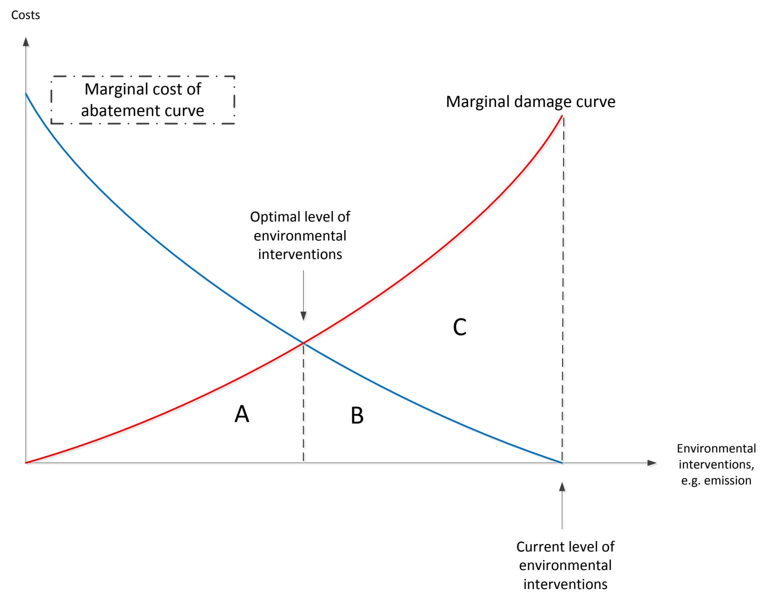

Figure 8 – Relation between marginal abatement cost and marginal environmental damage, based on Sterner and Coria (2012).

Basically, the above figures show the same thing but from different perspectives. They show the most cost effective level of the environmental interventions based on the trade-off between environmental costs arisen and costs to mitigate these interventions. The perspectives differ in how the optimal level is set. Figure 7 reveals humans' willingness to pay to avoid environmental damages, while Figure 8 estimates what it actually would cost the society to lower the interventions (i.e. abatement costs). The optimum is found in the intersection of the lines for both graphs. The red lines (up-sloping) in the figures estimate the marginal environmental damage cost that the interventions cause at a specific intervention level, and the areas under the red curves represent the total environmental costs. For example, current intervention level cause environmental costs A+B+C and an optimal intervention level cause environmental cost A. The blue line in Figure 7 represents the marginal willingness to pay and the blue line in Figure 8 represents the marginal abatement cost at a specific intervention level. The covered area between two points at the blue curve represents the total societal costs. For example, current level of interventions give no societal abatement costs while a reduction to an optimal intervention level gives abatement cost B in both figures. It is however difficult of construct such curves in reality because the environmental damage curves and the costs for reducing the interventions are uncertain. Also, spatial and temporal changes make it difficult to construct such curves (Sterner and Coria, 2012). However this would indicate the societal costs (environmental cost) to be paid for public goods and services, which would allow consumption and production at optimal levels (Johansson, 1999).

In weighting methodology it is an issue of where in the cause-effect chain the weights should be applied. As described in Subsection 3.1.4, prior to the first step in the cause-effect chain the environmental interventions are recorded. These interventions lead to primary effects and higher order effects damaging values people want to protect and preserve (Huppes and van Oers, 2011a). The question is where in the cause-effect chain the values should be placed. This depends on how good we think we are to predict environmental impacts and to make them relevant for us (Finnveden, 1997; Bengtsson, 2000a). For some environmental impacts the cause-effect chain is well known and well defined (e.g. acidification), while some impacts are difficult to model (e.g. toxicity and resources) (Baumann and Tillman, 2004). From another aspect, it is easier to draw conclusions from harm closer to the environmental entities people are valuating (Bengtsson, 2000b). Bengtsson (2000b) uses the example that it is easier to have an opinion about illness and water quality than about potential threats from impact categories e.g. global warming potential. This dilemma of where to put the values is illustrated in Figure 9 below (Bengtsson, 2000a).

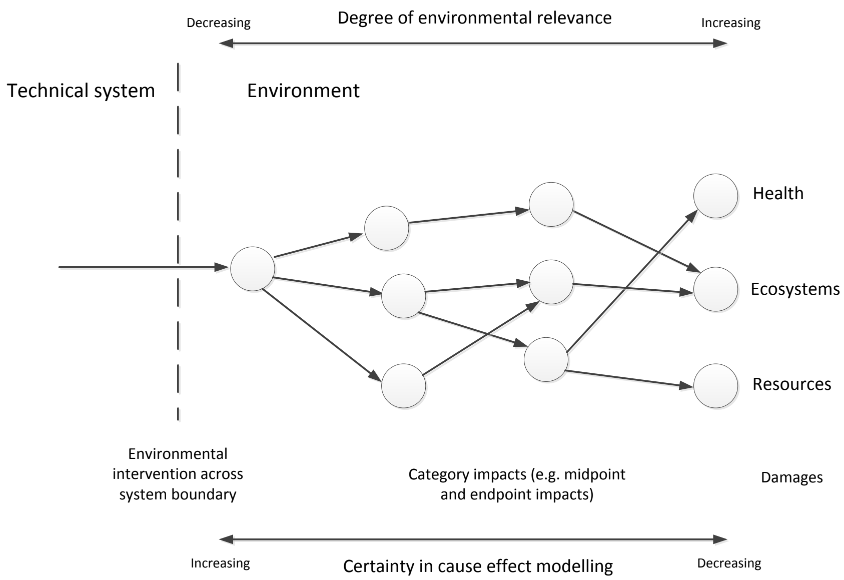

Figure 9 – Where to attach the values in the cause-effect chain? Figure based on Bengtsson (2000a).

Figure 9 shows that if weighting is performed close to the technical system (i.e. at the interventions) the environmental relevance is low and thereby it is difficult to make sound judgment in the valuation. At the same time, the scientific certainty is high early in the cause-effect chain. In contrast at higher order in the cause-effect chain, increased environmental relevance is given (i.e. easier to make sound judgment) but with larger scientific uncertainty (Bengtsson, 2000a; Pennington et al., 2004; Dong, Laurent and Hauschild, 2013). The choice of how far in the cause-effect modelling one should go depends largely on the type of environmental impacts to evaluate. Huppes and van Oers (2011b) identifies this issue of where to attach the weights and present three general procedures, which could be viewed in Figure 10 below.

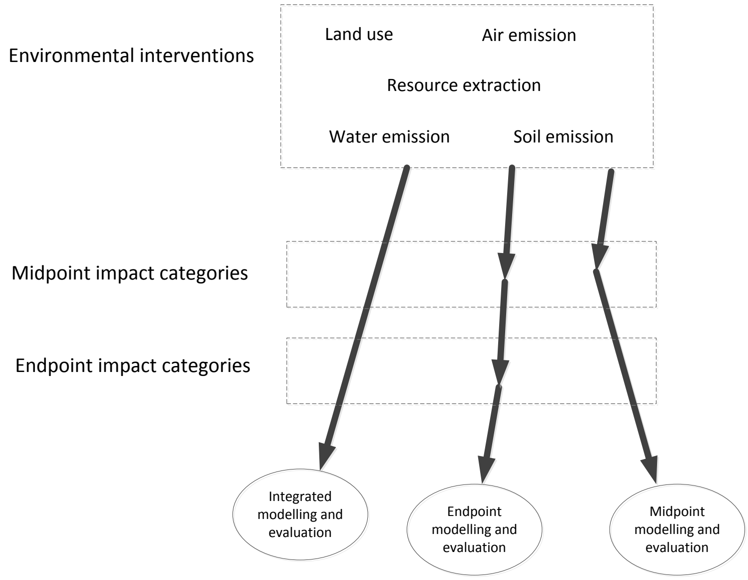

Figure 10 – Different procedures for attaching weights, based on Huppes and van Oers (2011b).

The criterion separating the three general methods is where in the cause-effect chain the weights are attached. For the *integrated modelling and evaluation,* weighting is attached to the interventions through integrated modelling of their environmental effects. The *midpoint modelling and evaluation* refers to midpoint impact categories (e.g. potential environmental impacts) where weighting is conducted at this point. In *Endpoint modelling and evaluation*, damages at endpoint level are weighted (Huppes and van Oers, 2011b).

The application of weighting methodology, within LCA, is mainly concentrated to product development and strategic decision making processes. The largest need of weighting is where a clear result is required. One example of that is early in a product development phase. A large number of alternatives need to be quickly screened through to choose those that are more environmentally friendly but also economically beneficial (Bengtsson, 1998). Weighting is also used to make strategic decisions, reducing the environmental risks in supply chains and make better informed decisions including large investments (WBCSD, 2011). It is thus mostly used as an in-company-tool (Bengtsson, 2000b). Weighting, by monetary terms, can also be utilized in other environmental system analysis methods such as Cost-Benefit Analysis and Life Cycle Costing (Finnveden et al., 2009).

When weighting is utilized it is a challenge to choose between ideological profiles and social preferences relevant for the particular study according to Bengtsson (2000b). The decision of which values to use will depend on several factors including the purpose of the study and the intended audience of the results. In situations when weighting are implemented, one tries to model both environmental effects associated with a certain product and attitudes of the selected social groups toward these effects (Bengtsson, 2000b). It is about to put the largest weighs and considerations to the most problematic environmental damages to avoid and mitigate them. This is often a good plan but not easy to realize (Bengtsson, 2000a).

3.2.2 Weighting: To be or not to be a part in LCA

Weighting is a highly discussed part within the LCA framework. It is continuously questioned whether it should be used in LCA studies or not (Baumann and Tillman, 2004; Ahlroth et al., 2011; Huppes and van Oers, 2011a; Itsubo et al., 2004). A dichotomy could be identified between those who are for and those skeptic to weighting. This subsection elaborates upon this issue, stating common argument for and against weighting.

According to Steen (1999a) many LCA-experts express dissent of the one number concept. They fear that transparency is lost when environmental impacts are described in one number. When weighting is implemented more information is added but less become communicated (Bengtsson, 2000b). This is however not a methodological issue since the underlying calculations are available for those who want to review them (Steen, 1999a). It is instead a communication problem, because people who are not involved in an LCA study and non-experts in LCA need to refer to result they lack background information about (Steen, 1999a). It is however very difficult for the decision maker (e.g. the designer) to process all the information from an LCA. Many companies instead use weighted results to indicate environmental impacts (Bengtsson, 2000b).

Apart from the importance of transparency and communicability in LCA, comprehensiveness is also required. This comprises the ability to evaluate all present environmental problems (Bengtsson, 2000b). For this issue, data gaps (i.e. inability to cover all parameters investigated in an LCA) are seen as a weakness for weighting methods (Bengtsson, 2000a; Bengtsson, 2000b). To find a method covering all the parameters in the inventory data is however unrealistic (Bengtsson, 2000b). It is also the case that LCA studies never cover all environmental impacts and thereby weighting methods could not be expected to cover all impacts in an LCA (Bengtsson, 2000a).

The basis for weighting is according to Bengtsson (2000a) subjective valuations based on humans´ perceptions of environmental problems, which is not desirable in a scientific method (Sleeswijk, Bijleveld and Sevenster, 2010). Weighting is thereby questioned to be a part of LCA, mixed with other objective steps (Bengtsson, 2000b). According to the ILCD (International Reference Life Cycle Data System) handbook, value choices in the impact assessment shall be avoided (European Commission, 2010). It is stated that this subjective information should not be the basis for decisions (Bengtsson, 2000a). On the other hand all the results in an LCA study are fed into the subsequent step called interpretation (Bengtsson, 2000b). This implies that weighting is not meant to deliver final verdict about the choice between alternatives (Bengtsson, 2000a; Bengtsson, 2000b). The weighting procedure should instead be regarded as something that can contribute with additional information to a decision making process (Bengtsson, 2000b). Weighting can thereby provide an indication of the option with the lowest environmental impacts (European Commission, 2005). It is also important to both include several weighting methods and to make sensitivity- and uncertainty analyses of them (Bengtsson, 2000b; Ahlroth et al., 2011). Since different weighting methods use different values, that may lead to different results and could assail the decision situation from different angles according to Bengtsson (2000b) and

Ahlroth et al. (2011), while sensitivity- and uncertainty analyses would grasp a wide range of possible results of the study (Bengtsson, 2000b).

The major application area nowadays for weighting is also criticized, stating that weighted results are difficult to use in product development. The two reasons stated in Bengtsson (2000a) are (1) weighted results are expressed in units unfamiliar to the product developers and other decision makers and (2) that the technical performance rarely are constant, since it differs between products. An example of the second argument is that it is difficult to determine if a technical improvement is enough to compensate for increased environmental impacts (Bengtsson, 2000a). In contrast Steen (1999a) states designers´ and decision makers´ request for practical, easily handled tools.

It has not been a tradition among the LCA-practitioners to use weighting why there is low support of it. Weighting is often approached with critical or restrictive attitude (Bengtsson, 2000b). An example documented in ISO states that: "weighting shall not be used in LCA studies intended to be used in comparative assertions intended to be disclosed to the public" (ISO, 2006b). On the other hand, in some countries and companies the weighting issue is less controversial (Bengtsson, 2000b). Despite all the controversies of weighting, it is widely used in practice (Hanson, 1999; Ahlroth and Finnveden, 2011).

The contradictions of weighting are based on several different issues where some are stated above: transparency, subjective nature and tradition in LCA. Examples of other aspects that are discussed in weighting are: geographical and temporal representativeness (e.g. geographical boundaries, temporal cut-off and discounting of the impacts and equity). Weighting methods are difficult to evaluate since no one knows which values are the correct to use, stated in Bengtsson (2000a), and since the values are difficult to find (Ahlroth and Finnveden, 2011). The evaluation of weighting sets are however of high importance with regard to scientific quality, consistency and data gaps (Ahlroth et al., 2011). According to Ahlroth et al. (2011) important criteria for the evaluation are:

- Are the methods logically consistent or are there any errors?
- Are there significant data gaps concerning impacts and/or interventions?
- Are the methods and data used updated and reflecting best available science?
- Are the results reasonable?

Whether or not weighting will be an accepted part of LCA remains to be seen. It is the case that natural science provides environmental system analysis models in order to connect technical activities with environmental impacts. This will always remain as a descriptive state of the environment. It cannot distinguish between what is serious and what is not or whether a certain kind of impact is desired or not. In these situations, practical reasoning and judgment will be needed for making these decisions. A way to accomplish this is by weighting (Bengtsson, 2000a).

3.2.3 Monetary weighting methods in LCA

Many different weighting methods have been generated in order to match different decision situations (Bengtsson, 2000a; Ahlroth et al., 2011). Several weighting methods developed 20-30 years ago, are still used. They are based on different principles and could for example be separated based on how their indices are expressed, either in a monetary or a non-monetary unit (Ahlroth et al., 2011; Ahlroth and Finnveden, 2011). The monetary term is a useful metric since it is easily comparable and a widely understood measure for the user (Steen, 1999a; Nordén, 2013). Generally any environmental impact can be converted to a monetary value, given that it reach public acceptance (e.g. controversial issue of valuating a human´s life) (ExternE - External Costs of energy, 2012a).

Monetary weighting gives according to Steen and Lindblad (2014) an added value towards non-monetary weighting and provides the opportunity to compare the external environmental costs against other internal costs and benefits (e.g. production cost and profit margin). By accounting for the environmental externalities the opportunity is given to internalize environmental externalities, which gives an indication of possible future costs if for instance more stringent policy measures are implemented (e.g. taxes on emissions and resources) (Tekie and Lindblad, 2013; WBCSD, 2011; Itsubo et.al., 2004). An externality is either a positive or a negative spillover effect, affecting others than those involved in the transaction of a good or a service. This means that one part of the cost is not reflected in the price agreed upon the market (Sterner and Coria, 2012). Before the external environmental costs can be internalized in e.g. business calculations or as a tax of a product price, they need to be estimated (European Commission, 2005). This can be performed through different monetary weighting methods. Several methods are found in the literature, for example EPS, LIME, Ecovalue, ASEK, PUMA - E P&L[2], ExternE, Ecotax and Stepwise. Five of these methods are reviewed below, namely EPS, ExternE, Stepwise, LIME and Ecovalue. Some of these methods are full LCIA methods, i.e. both the mandatory LCIA steps and weighting are included e.g. EPS and LIME, while others only contain a weighting set e.g. Ecovalue.

The chosen methods to review are selected based on several criteria such as how common the method is, data availability of the method, age of the data in the method, different values represented among the methods. The review consists mainly of:

- How is the method used?
- Which are the values in the method?

In the next Subsection 3.2.4 the methods are evaluated based on their strengths and weaknesses. This gives an opportunity to select proper weighting methods to be utilized in the case study presented in Chapter 4.

Environmental Priority Strategies (EPS)

The weighting method Environmental Priority Strategies (EPS) was one of the first monetary weighting methods and created by Bengt Steen, starting in the year 1989. The latest method update is from year 2000, named EPS2000d and is fully described in Steen (1999a) and Steen (1999b). The creation of EPS started on a request from the Swedish industry with the purpose to create a tool in the area of product design to evaluate products´ performances from an environmental perspective. This is to make engineers aware of environmental costs and design products with lower environmental impacts. Apart from companies' internal product development, EPS can also be used in other situations, such as environmental declarations, purchasing decisions and environmental accounting (Steen, 1999a).

In EPS several endpoint impact categories and safeguard subjects are included. The impact categories convey damages to humans and the environment (Steen, 1999a;

[2] Environmental profit and loss account

Steen, 1999b). The cause-effect chains start however with the environmental interventions (i.e. inventory data). Pathway specific characterisation factors are specified for each substance in the cause-effect chain from environmental intervention until endpoint effects are reached. The different endpoints are given monetary weights (expressed in the monetary unit, ELU) according to their severity (Steen, 1999a). The connection between the safeguard subjects, the endpoint impact categories and their specific weighting factors can be seen in Table 2 below (Steen, 1999a; Steen, 1999b).

Table 2 – Connection between safeguard subjects, the endpoint impacts and their specific weighting factors in the EPS method, based on Steen (1999a) and Steen (1999b).

Safeguard subjects	Impact category	Category indicator	Indicator unit	Weighting factor (ELU/indicator unit)
Human health	Life expectancy	YOLL (years of life lost)	Person∗years	85 000
	Severe morbidity	Severe morbidity	Person∗years	100 000
	Morbidity	Morbidity	Person∗years	10 000
	Severe nuisance	Severe nuisance	Person∗years	10 000
	Nuisance	Nuisance	Person∗years	100
Ecosystem production	Crop growth capacity	Crop	Kg	0.15
	Wood growth capacity	Wood	Kg	0.04
	Fish and meat production capacity	Fish and meat	Kg	1
	Soil acidification	Base cat-ion capacity of soil	Mole H+ -equivalents	0.01
	Production capacity for irrigation water	Irrigation water	Kg	0.003
	Production capacity for drinking water	Drinking water	Kg	0.03
Abiotic stock resources	Depletion of reserves	Reserves	Kg of element	0 – 59 400 000 [3]
	Depletion of oil reserves	Fossil oil	Kg	0.506
	Depletion of coal reserves	Fossil coal	Kg	0.0498
	Depletion of natural gas reserves	Natural gas	Kg	1.1
Biodiversity	Species extinction	NEX (Normalised extinction of species)	Dimensionless	$1.10*10^{11}$
Cultural & recreational values[4]	-	-	-	-

In order to calculate the environmental cost according to the EPS method, a general example is given. A substance is first released to the air (kg of a substance). The substance cause endpoint impacts through the cause-effect chain modelling (Impact category per substance emission, see equation (3)) and a monetary weight is given for how severe these impacts are (Value per Impact category, see equation (3)). The total economic value for the certain substance emission can be given by multiplying these

[3] Depend on reserve considered.

[4] No general values have been developed. Values have to be found in each specific case (Steen, 1999b)

three factors, as in equation (3) below (Steen, 1999b). In Chalmers (2000) all economical values per environmental intervention can be found, based on Steen (1999b).

$$\text{Substance emission (kg)} * \frac{\text{Impact category [Indicator unit]}}{\text{Substance emission (kg)}} * \frac{\text{Value (ELU)}}{\text{Impact category [Indicator unit]}} = \text{Value (ELU)} \quad (3)$$

By modelling the environmental damages (i.e. endpoint orientated modelling), it is easier for a layperson to understand what to valuate. The result from the above equation is expressed in ELU (environmental load unit) which has its origin from the more commonly known currency Euro (Steen, 1999a). The monetary values are derived from the impact categories by different valuation methods including willingness to pay and in some cases avoidance costs and market prices (Steen, 1999a; Huppes and van Oers, 2011a). The ILCD handbook advocates according to Lindfors et al. (2012) damage (endpoint) based modelling but simultaneously states that only a few such methods, in contrary to midpoint based models, are robust enough to be recommended.

The willingness to pay is measured for the 1998´s OECD population, applied to those affected by a change (Tekie and Lindblad, 2013). Although these values were taken for quite some time ago, basic values of the environment are stable over time. The 1998 year´s values are today also used both for non-OECD countries and for future generations (Steen, 1999a).

The impact models are spatially measuring effects on global levels and temporally effects as long as the impacts prevail. The EPS method is using discount rate of 0 %, which states that present and future effects are equally severe. This could be interpreted as present and future generations are equally worth a good environment (Steen, 1999a). In some studies where discount rate is used, the greenhouse effect tends to be negligible (Azar and Sterner, 1996).

As in all models, large uncertainties exist and should be handled in some way according to Steen (1999a). In EPS, an uncertainty principle is used where all data in the analysis should be accompanied a quantitative estimation of the uncertainty. When uncertainty in input data is estimated the uncertainty range in calculated values can be determined. It can thereby be illustrated how the variations in input data influence the conclusions. A Monte Carlo analysis could be used in order to handle uncertainties and sensitivities in the EPS method. It is also possible to alter the default EPS method to allow design for alternative impact assessments. For instance different spatial and temporal conditions could be used site-specific, which affect the pathway specific characterisation factors. These could also be altered to match a company´s environmental policy, given other priorities than in the default method (Steen, 1999a).

Externalities of Energy (ExternE)

Externalities of Energy (ExternE) is nowadays the most common method for monetary evaluation of externalities in Europe and advising decision makers about environmental, energy and transport issues (Huppes and van Oers, 2011a). In the scientific community, ExternE is widely accepted and often considered as the world reference of monetary evaluation (European Commission, 2005). It was started in year 1991 by American and European experts in a joint project of the EC/US Fuel Cycles Study that evaluated externalities of energy use. The project was completed in year 2005 publishing the methodological updated report: European Commission (2005). There have been a number of follow up projects to ExternE, refining the framework with new scientific knowledge and reducing uncertainties and data gaps

(ExternE - External Costs of energy, 2012d; Tekie and Lindblad, 2013). During the ExternE project, a web-based software tool was created from the ExternE framework, where environmental interventions could be monetized (European Commission, 2005). The software has been updated in line with the follow-up projects and can be found in EcoSenseWeb (2011), where 13 pollutants are included (Weidema, Hauschild and Jolliet, 2008).

The environmental impacts and costs in ExternE are quantified via the Impact Pathway Approach (IPA). In IPA one tracks the impacts from the source emission through the chemical transformation effect on receptors such as air, soil and water to physical impacts and thereafter expressing them in monetary terms. This is conducted in four steps (European Commission, 2005):

(1) Emission: source of the pollutant including the specific site, technology and the amount of the emission is stated.

(2) Dispersion: calculating the chemical conversion in the atmosphere, this is the quantification of the increased amount of pollutants in the affected area.

(3) Impact: Estimate the dose-response function, which shows the effects on different receptors (e.g. population and forest). The effects include the physical damage that the pollutant causes (e.g. increased number of hospitalizations).

(4) Cost: Monetary valuation estimated, e.g. the monetary cost of medical treatment and people´s willingness to pay to avoid residual suffering (i.e. welfare loss for individuals).

This result in externality costs expressed in Euro per mass unit of the intervention (European Commission, 2005).

Large uncertainties within the method are recognized in European Commission (2005), stated below.

- Data uncertainties (e.g. values of the input data)
- Model uncertainty (e.g. causal links between a pollutant and a health impact, appearance of the dose-response function)
- Uncertainty about policy and ethical choices (e.g. discount rate, value of human life)
- Uncertainties about the future (e.g. potential for a reduced amount of crops)
- Idiosyncrasies of the analyst (e.g. interpretation of incomplete and ambiguous information)

ExternE aims to cover all relevant impacts that can give external effects (European Commission, 2005). Effects included are divided into human health and environmental effects, presented in Table 3 and Table 4 below. Human health effects include mortality and morbidity, while environmental effects include building material, crops, global warming, amenity loss, land use change and ecosystem (ExternE - External Costs of energy, 2012b; ExternE - External Costs of energy, 2012c). These damages represent welfare losses for individuals, where some impacts (crops and building material) use market prices to evaluate the damage costs. Other impacts (especially mortality and morbidity) are evaluated through willingness to pay or willingness to accept based on European inhabitants´ individual preferences (ExternE - External Costs of energy, 2012e).

Some of the environmental interventions covered in ExternE can be viewed in Table 3 and Table 4 below, connected to their impact categories and effects. It is however difficult to explicitly state the monetary value for certain interventions or impact categories, since that depends on the impact pathway described in IPA but also the discount rate. The evaluated damages are stated in Euro and a discount rate of the impacts is suggested to be 0-6 % (European Commission, 2005).

Table 3 – Interventions in ExternE and their impacts, based on European Commission (2005).

Effects on		**Environmental interventions**	**Examples of effects**
Human health	Morbidity	PM_{10}, $PM_{2.5}$, SO_2, O_3	Reduction in life expectancy due to short and long time exposure
		Heavy Metals	Reduction in life expectancy due to short and long time exposure
		Accident risk	Fatality risk from traffic and workplace accidents
		Noise	Reduction in life expectancy due to long time exposure
	Mortality	PM_{10}, $PM_{2.5}$, O_3, SO_2	Respiratory hospital admissions
		PM_{10}, $PM_{2.5}$, CO	Congestive heart failure
		Heavy Metals	Cancer risk
		PM_{10}, $PM_{2.5}$	Respiratory symptoms
		Mercury	Loss of IQ of children
		O_3	Asthma attacks
		Noise	Myocardial infarction, Sleep disturbance
		Accident risk	Risk of injuries from traffic and workplace accidents

Table 4 – Interventions in ExternE and their impacts, based on European Commission (2005).

Effects on		**Environmental interventions**	**Examples of effects**
Environmental damage	Building material	SO_2	Ageing of: galvanized steel, limestone, mortar and paint
		Combustion particles	Soiling of buildings
	Crops	NO_x, SO_2, O_3	Yield change for wheat, rice and potato
		N, S deposition	Fertilising effects
	Global warming	CO_2, CH_4, N_2O	World-wide effects on mortality, morbidity, agriculture, energy demand, and economic impacts due to temperature change and sea level rise
	Amenity loss	Noise	Amenity losses due to noise exposure
	Ecosystem	SO_2, NO_x, NH_3	Acidity, euthrophication, PDF (Potentially Disappeared Fraction) of species
	Land use change	Land changed	PDF of species

Stepwise

Stepwise is a European LCIA method origin from the LCIA methods EDIP2003 and IMPACT2002+ with some adjustments (e.g. new impact categories). The aim of the method is to reduce uncertainties and incompleteness accompanied with monetizing environmental impacts, which has been a problem for LCIA weighting methods (Weidema, 2009).

The Stepwise method covers the three safeguard subjects: human, ecosystem and resource. These provide a complete framework of all imaginable values for protection, where a parallel can be drawn to people, planet and profit popularly used by WBCSD (World business council for sustainable development). The safeguard subjects are generated from a number of midpoint impact categories which can be seen in Table 6 below. Several hundred interventions are covered in Stepwise which are classified in midpoint impact categories. The pathway modelling of the interventions are based on EDIP2003 and IMPACT2002+. The result of the midpoint modelling is transferred into a monetary value in four steps (Weidema, 2009).

In the first step weights are put on the midpoint impact categories based on their contribution to the safeguard subjects they belong to. The weights are expressed in Biodiversity Adjusted Hectare Years (BAHY), Quality Adjusted Life Years (QALY) and in the currency Euro with the average value from year 2003 ($EURO_{2003}$), per characterised unit at midpoint (e.g. kg $PM_{2.5}$-eq. for Respratory inorganics). $EURO_{2003}$ is the unit of the safeguard subject *resources* measuring the resource productivity. BAHY is the metric of the safeguard subject *ecosystem* measuring the state of the ecosystem. QALY is the metric of the safeguard subject *humans* measuring the human-wellbeing. 1 BAHY is defined as 1 ha∗yr with full protection of an ecosystem, while 1 QALY is defined as 1 human life-year lived at full well-being. All species in an ecosystem are given equal weight and likewise for all humans (Weidema, 2009).

The second step contains the monetary evaluation of the safeguard subjects. The transformation procedure to monetary terms ($EURO_{2003}$) from BAHY and QALY are shown in equations (4) and (5) below (Weidema, 2009).

For impacts on human well-being:

$$\text{Transformed monetary value [EURO2003]} = \text{QALY [yrs} * \text{ha]} * \frac{\text{EURO2003}}{\text{QALY[yrs} * \text{ha]}} \quad (4)$$

For impacts on ecosystems:

$$\text{Transformed monetary value [EURO2003]} = \text{BAHY [yrs} * \text{ha]} * \frac{\text{EURO2003}}{\text{BAHY[yrs} * \text{ha]}} \quad (5)$$

No transformation is needed for impacts on resource productivity ($EURO_{2003}$) since that is already presented in monetary terms, as economic production value forgone for future generations (Weidema, 2009).

The values in Stepwise represent the society´s willingness to pay to avoid environmental damage and the transformation factors are based on budget constraints. The budget constraints are set to the average annual income which is the maximum an average person can pay for an additional life year. The transformation factors are set to 74 000 $EURO_{2003}$/QALY with an interval of 62 000-84 000 $EURO_{2003}$/QALY and 1400 $EURO_{2003}$/BAHY with an interval of 350-3500 $EURO_{2003}$/BAHY. This enables determination of externalities in a monetary unit (Weidema, 2009).

The third step contains the aggregation of all monetary values for each safeguard subject, which is expressed in $EURO_{2003}$ per characterised unit at midpoint (Weidema, 2009).

In the fourth and final step the total monetary value for each midpoint category and intervention can be calculated by using the equation (6) below (Weidema, 2009).

$$\frac{\text{Monetary value [EURO2003]}}{\text{Environmental intervention}} = \frac{\text{Monetary value [EURO2003]}}{\text{Characterised unit at midpoint}} * \frac{\text{Characterised unit at midpoint}}{\text{Environmental intervention}} \quad (6)$$

A summarizing figure of these four steps can be found in Table 5 below, by using the intervention CO_2 and midpoint category global warming as the example. In this example the characterised unit at midpoint is kg CO_2-eq.

Table 5 – Calculation procedure in Stepwise, based on Weidema (2009).

Impact category		Global warming
Unit of characterised value at midpoint		Kg CO_2-eq
Impact on ecosystems	BAHY/ Kg CO_2-eq (step 1)	$5.8 * 10^{-5}$
	$EURO_{2003}$/BAHY (step 2)	1 400
	$EURO_{2003}$/ Kg CO_2-eq (step 2)	0.082
Impacts on humans	QALY/ Kg CO_2-eq (step 1)	$2.1 * 10^{-8}$
	$EURO_{2003}$/QALY (step 2)	74 000
	$EURO_{2003}$/ Kg CO_2-eq (step 2)	0.0016
Impacts on resources	$EURO_{2003}$/ Kg CO_2-eq (step 1)	$-3.7 * 10^{-4}$
All impacts aggregated	$EURO_{2003}$/ Kg CO_2-eq (step 3)	0.083
Characterisation unit at midpoint/ environmental intervention	Kg CO_2-eq/ Kg CO_2 (step 4)	1
Monetary value	$EURO_{2003}$/ Kg CO_2 (step 4)	0.083

The calculation procedure is presented in the above Table 5. In step 1, all weights (BAHY, QALY and EURO) per characterised unit (kg CO_2-eq) at midpoint are identified. In step 2 these are translated to $EURO_{2003}$ per characterised unit (kg CO_2-eq) at midpoint. In step 3 these values are summed for all impacts (ecosystems, humans and resources). In Step 4 the economic value per specific intervention is calculated. A complete list of all monetary weights for the included midpoint impacts can be seen in Table 6 below.

Table 6 – Complete list of all monetary weights included in Stepwise, from Weidema (2009).

Midpoint impact category	Characterised unit	$EURO_{2003}$/characterised unit at midpoint
Acidification	m^2 UES	0.0077
Aquatic Ecotoxicity	kg-eq. TEG water	$7.1 * 10^{-6}$
Terrestrial Ecotoxicity	kg-eq. TEG soil	0.0011
Aquatic Eutrophication	kg NO_3-eq.	0.1
Terrestrial Eutrophication,	m^2 UES	0.013
Global warming	kg CO_2-eq.	0.083
Human toxicity	kg C_2H_3Cl-eq.	0.27
Injuries (road or work)	fatal injuries-eq.	$4.2 * 10^{6}$
Ionizing radiation	Bq C-14-eq.	$2 * 10^{-5}$
Mineral extraction	MJ extra	$4 * 10^{-3}$
Nature occupation	m^2 arable land	0.12
Ozone layer depletion	kg CFC-11-eq.	100
Photochemical ozone	m^2 UES∗ppm∗h	$3.7 * 10^{-4}$
Respiratory inorganics	kg $PM_{2.5\text{-eq.}}$	68
Respiratory organics	Pers∗ppm∗h	0.26

According to Weidema, Hauschild and Jolliet (2008) the main reason for choosing these specific impact categories is the completeness in coverage, in terms of substances included and the extent of the cause-effect chain covered. These are expected to cover all potentially important environmental impact categories, except from noise and invasive species dispersal. However, uncertainties in the Stepwise method are recognized both related to the characterisation factors, i.e. in the transition from inventory to impact result, and the weighting factors. The uncertainties vary between different impact categories where some can have an error margin of several orders of magnitude (Weidema, Hauschild and Jolliet, 2008).

LIME

LIME was created with the purpose to develop a Japanese version of a damage oriented impact assessment method, and thereby its name LIME (Life-cycle Impact assessment Method based on Endpoint modelling). Further it was developed to create a database allowing the industry to perform reliable LCAs and the first version of LIME was launched in year 2003 (Itsubo et al., 2004). Three years later in 2006 a revised and improved version was developed. More impact categories were included with improved representativeness and credibility of the weighting factors. LIME is widely used by Japanese companies to evaluate their products´ environmental performances and environmental costs (Tekie and Lindblad, 2013).

The impact assessment in LIME consists of a stepwise procedure. The environmental interventions are translated to impact categories at midpoint level. The midpoints impacts are further transformed to damages on endpoint level. Weighting can be performed for each of the safeguard subjects for which the endpoints are related to. In Figure 11 below, a schematic view of the method is presented (Huppes and van Oers, 2011a; Itsubo et al., 2004; Itsubo et al., 2012).

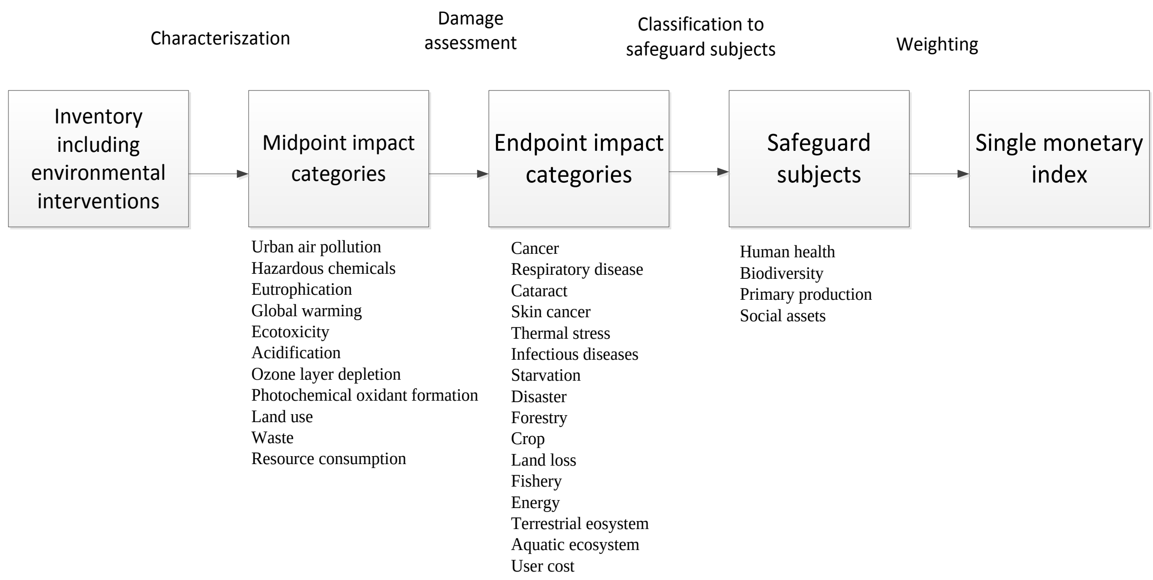

Figure 11 – Schematic representation of the modelling in LIME, from environmental interventions to weighting. Based on Huppes and van Oers (2011a), Itsubo et al. (2004) and Itsubo et al. (2012).

In Figure 11, all midpoint and endpoint impact categories and the four safeguard subjects for LIME can be seen. Table 7 below presents some of the inventory data that are covered in LIME.

Table 7 – Inventory data covered by the LIME method. Based on Huppes and van Oers (2011a), Itsubo et al. (2004) and Itsubo et al. (2012).

Midpoint impact category	**Inventory (example[5])**
Urban air pollution	SO_X, NO_X
Hazardous chemicals	Mercury, Benzene
Eutrophication	Total N, Total P
Global warming	HCFCs, CO_2
Ecotoxicity	Mercury, Benzene
Acidification	SO_X, NO_X
Ozone layer depletion	HCFCs
Photochemical oxidant formation	Benzene, NMVOC
Land use	Land
Waste	Waste
Resource consumption	Copper, Oil

The valuation method for which the weighting factors in LIME (see Table 8) were derived consisted of four steps: sample selection, creation of questionnaire, interview survey and analysis (Itsubo et al., 2012).

Sample selection: Random sample selection of about 1000 respondents from all parts of Japan.

Creation of questionnaire: Questionnaire to be used for the interview survey, where the respondents have to choose between different policy alternatives of current and hypothetical situations revealing their willingness to pay to avoid damages to the safeguard subjects.

[5] Full list can be found in Itsubo et al. (2004)

Interview survey: Face-to-face interviews where the respondents were answering the questions in the survey.

Calculations: The results collected were statistically analysed to derive weighting factors. This was done in accordance with the random parameter logit model (RLP), where the detailed procedure can be found in Itsubo et al. (2012). The cause-effect modelling was made on Japanese level with some exceptions for global level (climate change, ozone layer depletion and resource depletion). The interventions are based on present time, while in the cause-effect modelling different time horizons are used. Discounting of future effects are not made (i.e. 0 % discount rate) (Huppes and van Oers, 2011a). The derived weighting factors for the four safeguard subjects can be seen in Table 8 below.

Table 8 – Weighting factors across the safeguard subjects used in LIME, from Itsubo et al. (2012).

Safeguard subject	Monetary weighting factors per unit of the safeguard subject	Safeguard subject unit
Human health	$1.47 * 10^7$	1 DALY[6] [year]
Social assets	$1.00 * 10^4$	10000 (Japanese Yen)[7]
Primary production	$4.63 * 10^4$	1 ton[8]
Biodiversity	$1.42 * 10^{13}$	1 EINES[9]

Ecovalue

The Ecovalue method contains a monetary weighting set to be used in LCIA. The first version of the method was published in year 2011, named Ecovalue08 (Ahlroth and Finnveden, 2011). The method has recently been updated and is now called Ecovalue12 (Finnveden, Håkansson and Noring, 2013). The monetary weighting set is attached to impact categories and represents the loss of benefits due to a lower environmental quality. This intends to reflect the economic damage value caused by different environmental flows (Ahlroth and Finnveden, 2011). The weighting set contains the following impact categories including the weights presented in Table 9 below (Finnveden, Håkansson and Noring, 2013).

[6] Disability Adjusted Life Year (DALY) is the sum of the years of life lost due to premature mortality and the years lived with disability. (Itsubo et al., 2004)

[7] Estimated loss of economic value through e.g. fishery, agriculture and forest (Itsubo et al., 2004; Itsubo et al., 2012).

[8] The net primary production (i.e. plant growth inhibition) (Itsubo et al., 2004; Itsubo et al., 2012).

[9] Expected Increase in Number of Extinct Species (EINES) is summing the number of species existing in Japan multiplied by the incremental risk of extinction of the species (Itsubo et al., 2004).

Table 9 – Weighting set in the Ecovalue method, from Finnveden, Håkansson and Noring. (2013).

Impact category	**Weights (mean value)**	**Weights (interval values)**
Abiotic resources	SEK 0.12 /MJ	SEK 0.004-0.24 /MJ
Global warming	SEK 2.85 /kg CO_2-eq	SEK 0.1-5.6 /kg CO_2-eq
Photochemical oxidation	SEK 27 /kg C_2H_2-eq	SEK 14-40 /kg C_2H_2-eq
Terrestrial acidification	SEK 30 /kg SO_2-eq	SEK 30 /kg SO_2-eq
Eutrophication, marine	SEK 90/kg N	SEK 90/kg N
Eutrophication, fresh water	SEK 670/kg P	SEK 670/kg P
Human toxicity	SEK 2.81 /kg 1,4 DB-eq	SEK 0.02-4.89 /kg 1,4 DB-eq
Marine water toxicity	SEK 12 /kg 1,4 DB-eq	SEK 12 /kg 1,4 DB-eq
Particulate matter formation	SEK 273 /kg PM_{10}-eq	SEK 273 /kg PM_{10}-eq

For some of the impact categories, interval values are suggested (e.g. abiotic resources and human toxicity) and some are fix (e.g. marine water toxicity and particulate matter formation). Interval values are included due to identified uncertainties in the valuation studies or due to that different interventions are included (Ahlroth and Finnveden, 2011). For instance 1 MJ of copper is not equally valuated as 1 MJ of oil. The full list of abiotic resources covered in Ecovalue can be found in Table 10 below (Finnveden, Håkansson and Noring, 2013; Ahlroth and Finnveden, 2011).

Table 10 – Abiotic resources covered in the Ecovalue method, from Finnveden, Håkansson and Noring. (2013), and Ahlroth and Finnveden (2011).

Substance	**Value [SEK/MJ]**
Copper	0.08
Lead	0.15
Nickel	0.12
Zinc	0.18
Gold	0.01
Iron ore	0.24
Gas	0.017
Hard coal	0.004
Oil	0.034

The Ecovalue method comprises of a two steps procedure (Tekie and Lindblad, 2013). Before the first step is conducted, all environmental interventions are classified to their respective impact category and calculations of environmental impacts are conducted in line with the characterisation procedure in LCIA. In the original Ecovalue08 version, this was conducted mainly through the CML (Institute of Environmental Sciences) baseline characterisation method. (Ahlroth and Finnveden, 2011). In the updated version, Ecovalue12, the characterisation methods ReCiPe (Goedkoop et al, 2009) and Cummulative Exergy Demand (Bösch et al, 2006) are used according to Finnveden, Håkansson and Noring (2013).

The first step in Ecovalue is the estimation of the damage values for the different impact categories (Tekie and Lindblad, 2013). Different procedures are used for determining this. It is either based on actual prices (i.e. market prices) or hypothetical prices where stated preference methods have been used. It is stressed that revealed preference methods have not been used since they do not include non-use values. For instance global warming and depletion of abiotic resources are based on global market prices; eutrophication and acidification are valued based on the Swedish inhabitants´ values, while formation of photochemical oxidation is based on both methods

(Ahlroth and Finnveden, 2011). In the second step the impact categories are weighted against each other based on the valuation in step 1 (Tekie and Lindblad, 2013).

Ahlroth and Finnveden (2011) points out that there are uncertainties in the values that one should be aware of when analyzing the results. Further tests of the weighting set in case studies are however important in order to develop a sound weighting set and to learn from the importance of different impact categories in order to come close to the "true" values (Ahlroth, 2009; Finnveden, Håkansson and Noring, 2013). A way of doing that is to compare results with other weighting methods (Ahlroth and Finnveden, 2011).

3.2.4 Monetary weighting methods to be used in the case study

From the above monetary weighting methods, arguments and aspects for the selection for which methods to use in the case study are presented. To choose proper weighting methods is a challenging task. There are many aspects of a weighting method and different methods could be suitable in different application areas and decision situations (Ahlroth et al., 2011). To identify proper methods to be used for paints, it might be a good idea to identify the environmental issues of these. Such issues are presented in Section 3.4. Also one needs to examine desirable principles (i.e. what are the values based on, whose values are used and how the values are derived) for the weighting method. As weighting is controversial in LCA, it is important to include weighting methods representing different principles. This is to assail the result from different angles and to overcome data gaps (Bengtsson, 2000b; Ahlroth et al., 2011).

When considering the weighting methods reviewed, no one is especially constructed to monetary evaluate environmental impacts from the life cycle of paint products. Instead some methods cover broad industrial product categories and related impacts, e.g. EPS and LIME, while others are more specified in a certain sector and specific impacts, e.g. ExternE. All of the reviewed methods are however created with the purpose to account for environmental costs caused by environmental impacts or damages.

Three of the methods reviewed (EPS, LIME and ExternE) put their weighs at the endpoint level, while the other two (Stepwise and Ecovalue) put weights at midpoint level. All of them have scientific bases as cause-effect chains are determining pathway specific characterisation factors. All of the methods, except Ecovalue use acknowledged safeguard subjects, identifying important values to protect and preserve. The Ecovalue method instead uses several midpoint categories and it is thereby difficult to judge which overall damages that should be avoided.

Apart from the data coverage, the data quality of the methods is also important. This could for instance include the age of the data. There are both advantages and drawbacks with old and new data. The reviewed methods´ data is in three cases (Stepwise, ExternE and LIME) from around year 2005, while one method (EPS) has older values from around year 2000 and one method (Ecovalue) with new values from year 2013. An advantage of old data might be that it has been reviewed and tested for example in case studies, where the opposite could be true for new data. On the other hand, data in weighting methods needs updating why new data could be preferable from this aspect.

The values in the methods are derived in different ways, e.g. through market prices, stated- and revealed preference methods. Some of the methods use values agreed upon by broad ranges of societies while others are using society specific values. It is however impossible to judge which are the correct values to use and how broadly accepted values should be. The project´s reference group requested for methods having Scandinavian and European values.

Thus, there are several aspects to include when choosing weighting methods and it is a trade-off in which methods to use. The selection of methods to be used in the case study is however made with respect to the strengths and weaknesses of the methods based on the above aspects and criteria. It is also important that they should be fairly simple to handle and to use. The selection of weighting methods is accomplished in cooperation with the project´s reference group. Thus, not only the weighting methods are subjective, also the selection of them is inherently subjective.

The monetary weighting methods that are utilized in the case study presented in Chapter 4 are:

- EPS
- Ecovalue
- Stepwise

3.3 Paint products and painting

A lot of different paints exist which are different from each other. Often, these have only two things in common, they are liquids and they are drying after attachment. This section aims to give a general overview of paint products including the following four subsections: *Characteristics of paint products, Content of paint products, Nano particles in paint products, Painting procedure.*

3.3.1 Characteristics of paint products

Paints could broadly be classified into two categories, Decorative paints and Industrial coatings. Decorative paints are applied on-site aiming to decorate, protect and prolong the life time of buildings and similar objects, while industrial coatings are applied on manufacturing goods in factories (University of York, 2013). In this report, outdoor decorative finishing paints applied on wood facades are studied and will be referred to as `paint´ if nothing else is stated. This is the type of paint product that is in focus in the case study in Chapter 4.

Paint is a liquid, intended to be used indoor or outdoor. Outdoors, it is used to protect the facade from dust and solar radiation and to decorate the facade creating a satisfying environment. When applying the paint on the facade it creates a continuous layer (a film) on the substrate, which enables longer durability time and makes it easier to clean the substrate (Chemiewinkel, Enterprise Ireland and WIMM, 2000; Kougoulis et al., 2012).

The properties of the paint are defined by the components in the paint (Hjort, 2012). It is rather difficult to state which properties that are the most desired, since that varies due to the paint´s application, function and the surrounding environment. Some usually desired attributes according to University of York (2013) are: ease of application, forming a continuous protective film, high opacity, color stability, durability, flexibility, easily cleaned and resistant to corrosion, water, heat, abrasion and scratch.

In the European standardization (CEN) a classification of outdoor paints are made. The paints are divided according to their appearances, application areas and functions. The appearance is divided into: layer thickness, coverage ability and gloss.

Application area is divided dependent on which type of construction the paint is applied on: stable constructions (e.g. wood windows and wood doors), semi-stable constructions (e.g. wood facades) and not stable constructions (e.g. fences). The function is described from the application area and from the impact on the product (mild, middle or hard climate) (Svenskt Trä, 2012b).

3.3.2 Content of paint products

The ingredients of paint are highly determining the function of the paint (CEPE, 2012). The content of paint distinguishes normally between the four components: binders, solvents, pigments and additives (Sveff, 2009, University of York, 2013). There are paints with only some of the components included, which indicate that paints are very different from each other.

A large number of paint variations can be produced because there are about 10 000 different binders, 9000 different additives, 4500 different pigments and fillers (Kougoulis et al., 2012). Many of these are not compatible with each other but still millions of paint combinations can be produced and many are decorative paints (Kougoulis et al., 2012). The various paint components are built up by different substances. The four main components and common substances are presented below.

Binder:

The binder is a polymer and acts as a fundamental ingredient in paints (University of York, 2013). It may be dissolved as a solution or carried as a dispersion of microscopic particles or droplets in a liquid (University of York, 2013; Svenskt Trä, 2012a). The binder binds the pigment together and enables adhesion to the substrate (NZIC, 2002). This makes the paint create a coat or a film on the substrate. Without binders, no protective surface is given. Instead the substrate is exposed to wear (Målare i Göteborg, 2011c).

Most of the paint types are named after the binders included in the paint (e.g. alkyd paint, acrylic paint, lin-seed oil paint and latex paint) (NZIC, 2002). Binders are usually divided into two groups based on how they are drying, physical or oxidative (Svenskt Trä, 2012a). In physical drying, the polymers are coalescence and subsequent integrated into a hard polymer matrix. In oxidative drying, the polymers are cross-linked by an oxidation reaction with oxygen (University of York, 2013; Overbeek et al., 2003). The different procedures can be seen in Figure 12 below. An example where physical drying occurs is for acrylic paints while alkyd paints are drying oxidative (Svenskt Trä, 2012a).

Common ingredients in binders are for instance alkyds, cellulose, acrylics and linseed-oil (Kougoulis et al., 2012; Axelsson et al., 1999; NZIC, 2002).

Solvent:

Also solvents (sometimes called thinners) are included in most paints. They dissolve the paint´s components and carry them in a liquid solution. The solution is thereby suitable for attachment on a substrate and thereafter the solvents evaporate (Sveff, 2009; NZIC, 2002). The drying procedure is thus due to both that the solvents are evaporating and that the binders are either oxidized or physical dried (Sveff, 2009; Målare i Göteborg, 2011c). Figure 12 below illustrates this procedure, which creates a film.

There are two types of solvents; water based and organic based (NZIC, 2002). Water based solvents are the most common and acrylic paint is one type of water based solvent. Organic solvents include volatile organic compounds (VOCs), which are harmful for both humans and the environment (University of York, 2013). Water based solvents are however the only non-harmful solvent. The paint industry is therefore striving for paints that can have water as the solvent without decreasing the quality of the paint (NZIC, 2002; Hjort, 2012). Transition challenges could be that organic based solvents are often cheaper in paints, have better appearance and application characteristics (Chemiewinkel, Enterprise Ireland and WIMM, 2000; University of York, 2013; Overbeek et al., 2003). Also end-consumers´ habits are mentioned as a barrier (Chemiewinkel, Enterprise Ireland and WIMM, 2000).

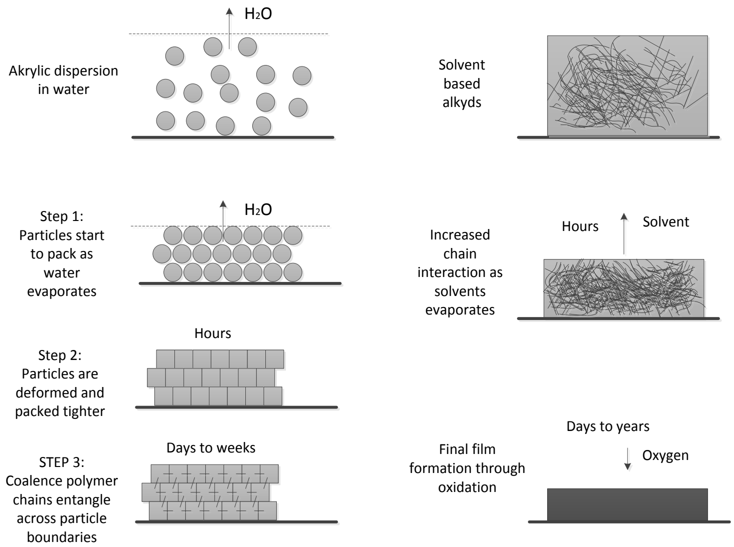

Figure 12 – Film formation of water based polymer dispersion and solvent based alkyd, based on Overbeek et al. (2003).

Figure 12 above illustrates the drying procedure and film formation when paint is attached to a substrate. The left part to the figure illustrates the physical drying procedure for water based acrylics and the right part illustrates oxidative drying for solvent based alkyds.

Common material used in solvents are for example hydrocarbons, alcohols, esters, ethers, ketones and water (CEPE, 2013b).

Pigment:

The next component is pigment. This is a powder giving the paint its color, gloss, hiding and protecting the substrate against ultraviolet light and weathering (NZIC, 2002). There are two different types of pigments, organic and inorganic (NZIC, 2002). Depending of the amount of pigment used in the paint it is divided to covering, translucent and transparent paint (Svenskt Trä, 2012a).

A covering paint (also called finishing paint) consists of as much pigment that light does not penetrate the paint coating to the wood surface. A translucent paint consists of a lower amount of pigment than a covering paint. Thereby it can partly be decomposed by sunlight reaching the wood. A transparent paint does almost miss protection against sunlight and is usually called varnish (Målare i Göteborg, 2011c; Svenskt Trä, 2012a).

Usual ingredients in pigments are white titanium dioxide (represents 70 % of all pigments used), zinc oxide, iron oxide, calcium carbonate and talc (CEPE, 2013b; Axelsson et al., 1999; University of York, 2013).

Additives:

Additives are often included in paints. Their aim is to improve the properties of the liquid paint or the dry film. This could for instance include dryers to reduce drying time, silicones to improve weather resistance, and additives for preservation of the paint, improving the texture and fungi inhibitory etc. (Sveff, 2009; Svenskt Trä, 2012a; Svenskt Trä, 2012b; University of York, 2013). Additives are normally added to a small amount (0.2-10 w%) but have significant effects on the product (European Commission, n.d). They are often divided due to their function, where some categories are: extenders, driers, coalescents, antifoams, defoamers, dispersing agents, biocides and catalysts. Often all kind of additives are aggregated (Dcarbon, 2008; Zuin et al., 2014).

The amount of the above four components differs between paints. NCMS (2011) distinguish between the components depending on if they are liquids or solids and state the general composition for water based and solvent based paints, see Figure 13 below. For solvent based paints, half the content is considered as organic based solvents and half are solid parts, namely binders, pigments and additives. For water based paints, little more than half the content is considered as water, while less than half are solids. There is also a small part organic based solvents in water based paints.

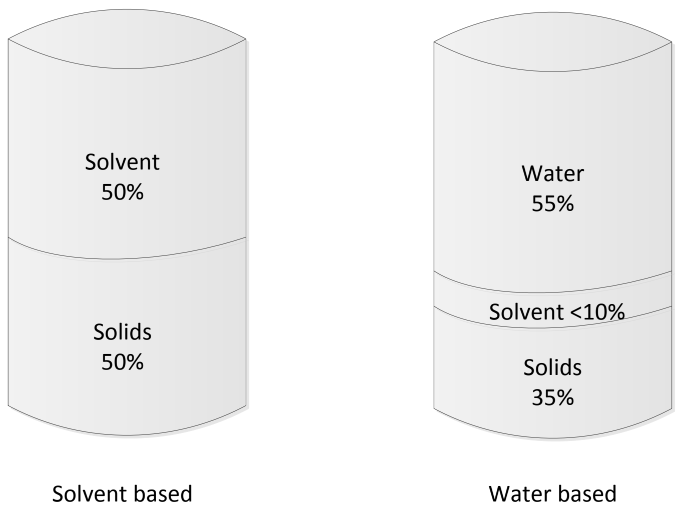

Figure 13 – General composition of decorative paints, based on NCMS (2011).

From this general picture of the paint composition, more specific compositions are sketched for two common paint types illustrated in Figure 14 and Figure 15 below (Axelsson et al., 1999). The figures below show the average ingredients for some of the most common paints, acrylic - water based and alkyd - solvent based. Binders, pigments and solvents are added to similar amounts, about 20 - 40 % each and additives are included to around 10 - 15 %. However, large variations of the composition for both paint types exist (Chemiewinkel, Enterprise Ireland and WIMM, 2000).

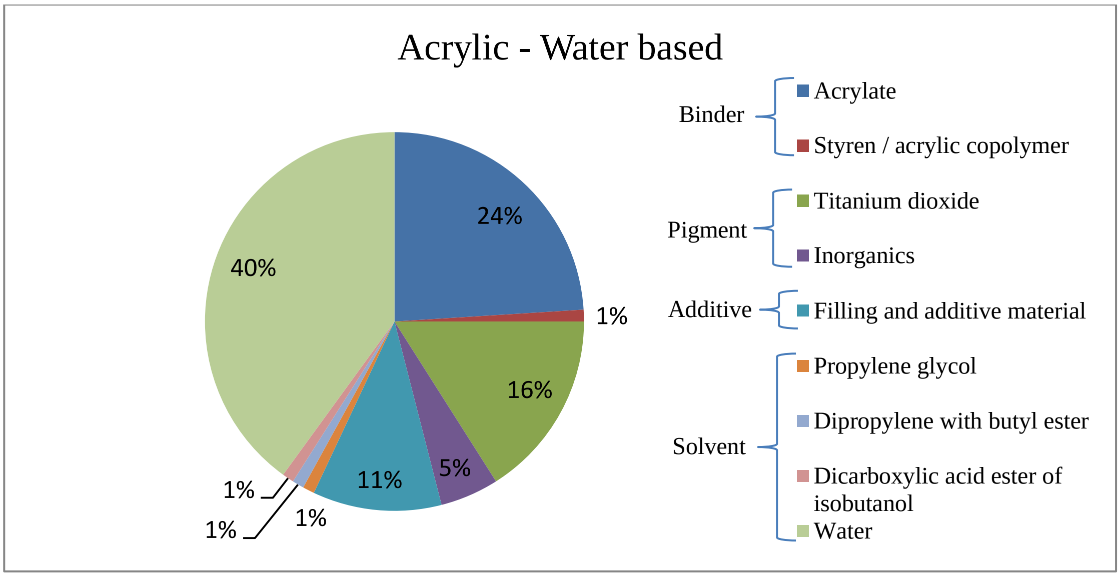

Figure 14 – Common paint composition for acrylic water based paint, based on Axelsson et al. (1999).

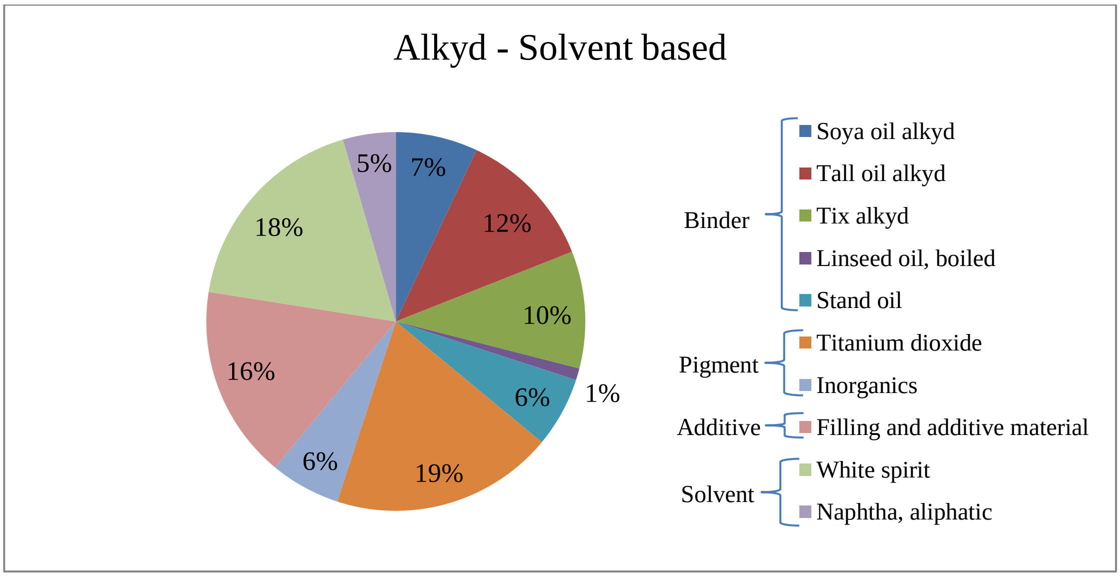

Figure 15 – Common paint composition for alkyd solvent based paint, based on Axelsson et al. (1999).

In order to protect and prolong the life time of the paint there is a need for maintenance of the paint. The maintenance comprises of washing the facade and there are generally two different product categories, separated due to their application area. One is used to remove mould and algae, so called maintenance wash. The other is used to remove fat, dirt and matte down the substrate which improves the adhesion for the next layer of paint, so called paint wash (Bok, Lindqvist and Hjort, 2009; Målare i Göteborg, 2011a). Some products can however fulfill both functions (Hjort, 2012).

Maintenance wash products can contain cationic surfactants, which neutralize the mould´s and algae´s negatively charged surface area and thereby eliminating them. Another type of maintenance wash products contains boric acid and borates, which affects the enzyme system of the microorganism (Bok, Lindqvist and Hjort, 2009). Other products that can fulfill the same function as maintenance wash products are methylated spirit, ammonia and degreasers (Målare i Göteborg, 2011a).

Paint wash products contain chemicals with high pH, such as potassium hydroxide and sodium metasilicate. A high pH contributes to dissolving the paint´s film and provides a good substrate for repainting. Simultaneously it removes fat and dirt from the substrate. An environmental friendly alternative is to use ordinary dish-soap, which was shown to perform better then several paint wash products on the market (Bok, Lindqvist and Hjort, 2009).

3.3.3 Nanoparticles in paint products

Nanotechnology is the study and manipulation of materials on nanoscale and is expected to become an important technology in the twenty-first century. One nanometer is one billionth of a meter with a width of approximately ten atoms. At this scale, materials have other properties than bulk materials (Hischier, 2014). Products with potentially better functionalities can thereby be produced and the paint industry is one sector that is starting to use these materials (Kougoulis et al., 2012). Recently, the uses of nanomaterial as an additive in paints have been widespread (Zuin et al., 2014). According to Kougoulis et al. (2012), over two hundred surface coatings and paints are registered including nanomaterials. It is also stated that many paints may contain nanomaterials without the paint producer´s knowledge of it. This could be the case for instance for titanium dioxide, silica, carbon black, alumina (Kougoulis et al., 2012).

There are mainly three types of nanomaterials: natural nanoparticles produced in biological systems, incidental nanoparticles that have been produced synthetic but without being specifically engineered to serve any purpose and manufactured nanoparticles intended to improve the product properties (Kougoulis et al., 2012). The last group consists of so called engineered nanomaterials (ENM). Common ENM substances are colloidal silica, carbon black but also titanium dioxide, silver, zinc oxide and alumina exist in nano particle size, where some of these already are included in paints (Wick, Krug and Nowackm, 2011).

In the case study in Chapter 4, modified colloidal silica will be of special interest. It is composed of amorphous silica particles (SiO_2) in an aqueous solution. The appearance of the colloidal silica depends on particle size, particle size distribution and concentration (Bergna and Williams, 2005; Zuin et al., 2014). The term colloidal is referring to how the particles are dispersed in the aqueous medium, which can occur in three different ways. There is no sharp boundary between these three phases but in general particles smaller than 1 nanometer, are said to be homogeneously solved in a solution. Particle sizes larger than 1 micrometer form a heterogeneous mixture with the aqueous medium falling down to the bottom due to sedimentation. The particle size in between 1 nanometer and 1 micrometer are in a colloid system with the aqueous phase (Slomkowski et al., 2011; Klint, 2011). The principle of colloidal dispersion is presented in Figure 16 below, where the circles illustrate the particles hold in an aqueous phase.

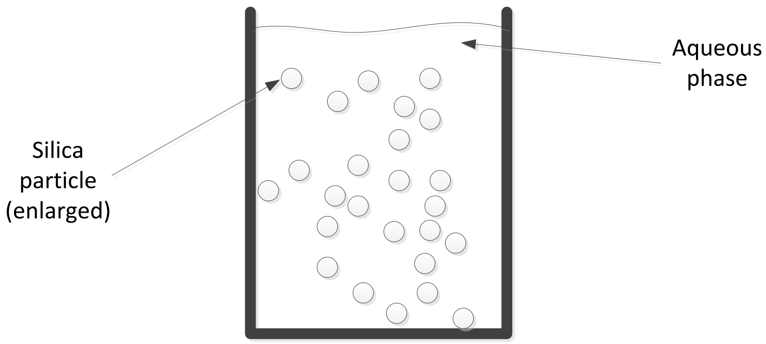

Figure 16 – Schematic sketch of colloidal dispersion of nanomaterial in an aqueous solution, based on Slomkowski et al. (2011).

The inclusion of colloidal silica in the paint formulation may lead to improved and new properties and functionalities, e.g. dirt resistance, humidity tolerance, water resistance (EMPA, 2014; Kougoulis et al., 2012), scratch resistance, tackiness (AkzoNobel, 2013c), self-cleaning, colour effects, antifouling, UV- and IR-blocking and biocide properties (Greenwood, 2012) to name a few, leading to a prolonged life time of the paint. The key for these performance improvements is according to Wick, Krug and Nowackm (2011) the large surface area of the particle compared to its volume, which simplify the possibility to modify the surface chemistry of the particles (Klint, 2011). However, environmental and health concerns regarding nanomaterials have risen (Kougoulis et al., 2012), which is described in Subsection 3.4.2.

3.3.4 Painting procedure

As the case study includes outdoor paints, the painting procedure for such paints is described. The recommendation for which type of paint that is suitable differs between different situations. Different paints give different properties e.g. durability, gloss, coverage and preservation. In addition to the paint, there are other important products required in order to fulfill the overall functions of a paint; to protect & cover the facade and create a nice appearance. This includes preparation- and maintenance products.

Except from the content of the paint, the function of it can vary a lot depending on how the paint is attached to the substrate, normally conducted either by private persons or professional painters (Chemiewinkel, Enterprise Ireland and WIMM, 2000). For instance, too little paint give worse protection and a patchy surface, while too much paint give wrinkles and prolonged drying time (Hjort, 2012). The weather is also a parameter to keep in mind, where dry weather is preferable. This keeps the moister content of the wood low, which is needed for proper application and durability of the paint (Chemiewinkel, Enterprise Ireland and WIMM, 2000).

When painting on a new facade there are mainly three things needed for the formation of a paint system: Impregnation, priming, and finish painting. The impregnator oil penetrates the wood and prohibit dust uptake and decrease crack risk and rot & blue-strain fungus (Nordsjö, n.d). The next step is priming which is conducted for two reasons mainly, (1) it eliminates mould fungus in the wood and (2) it stabilize the surface of the wood enabling proper adhesion for the finishing paint (Målare i Göteborg, 2011b). About a day after the primer has dried, the finish painting must be conducted according to Hjort and Bok (2010), to avoid spoiling the primer. The last step is the finish painting, which is conducted by two layers of paint are attached to the facade (Hjort, 2012). There are several methods for paint application in general, including brush, roller, flowcoating, spraying, electrostatic spraying, airless spraying, vacuum impregnation and immersion. The most common for outdoor painting on wood facades is to use a brush or a roller (University of York, 2013; Chemiewinkel, Enterprise Ireland and WIMM, 2000).

After the paint is applied to the facade there is a need for maintenance work, i.e. washing. This can be conducted in connection to repainting or more regularly. As described in the Subsection 3.3.2, two different types of detergents can be used, *paint wash* (improve adhesion) and *maintenance wash* (for mould and algae reduction).

In the case study presented in Chapter 4, a maintenance wash, a paint wash, an impregnator, a primer and two finishing paints are included, forming two different paint systems. Focus is on the finishing paints.

3.4 Paint from an environmental perspective

This section will sketch paints from an environmental point of view. It starts with *The life cycle of paint*. In the subsequent subsection, an identification of the main environmental issues for paints is accomplished, presented in *Environmental issues of paint*. This section is finalizing by *Eco-labeling of paint* and *AkzoNobel Eco-premium solutions*.

3.4.1 The life cycle of paint

The life cycle of a paints could be displayed in five steps (see below), contributing to environmental impacts (NCMS, 2011; Kougoulis et al., 2012; Chemiewinkel, Enterprise Ireland and WIMM, 2000). These are:

Step 1: Raw materials extraction and processing

The raw materials used to manufacture paints are extracted from the earth, which are processed to the four paint components (binder, solvent, pigment and additive) shown in Figure 17 below. These are further packaged and transported to paint manufacturing (NCMS, 2011).

Step 2: Paint manufacture

The paint manufacturing is a process industry where the manufacturing according to CEPE (2012) is mainly a matter of mixing the carefully evaluated components and is a fairly simple process (NZIC, 2002). According to Chugoku Marine Paint (2012) the following steps are included in the manufacturing: pre-mixing, milling, blending, filtering, filling and packaging. These are physical processes where no chemical reactions are involved, which can be carried out in room temperature (European Commission, n.d). The paint is thereafter transported to wholesalers or retail stores, which sell the paint to the users (Chemiewinkel, Enterprise Ireland and WIMM, 2000). Paint is in general produced in batch processes for up to 10 000 liters (Resene, 2013). The procedure differs between different paints but Figure 17 below aims to illustrate the general procedure (NCMS, 2011; Chugoku Marine Paint, 2012).

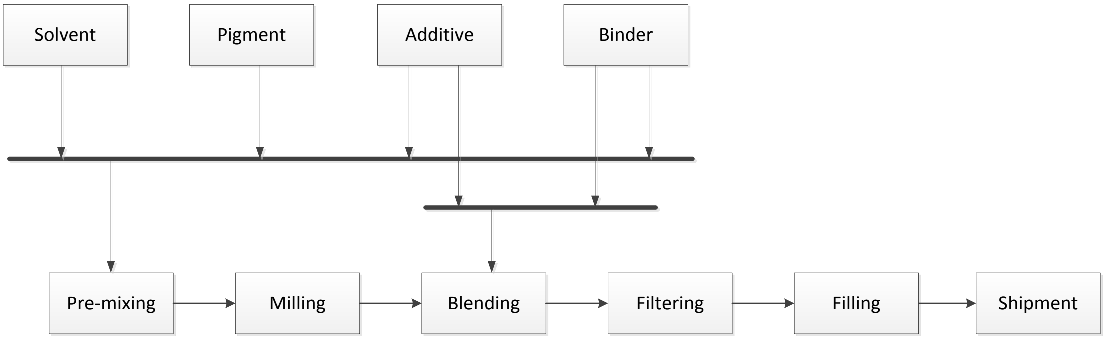

Figure 17 – General procedure for paint manufacturing, based on NCMS (2011) and Chugoku Marine Paint (2012).

In the first step, pre-mixing of all components is included. These are further processed in milling and blending where more additives and binders are included. The product is further refined and packaged until shipment.

Step 3: Paint use (application of the paint)

This step of the user phase contains all environmental impacts that are connected to the application of the paint, for instance evaporation of solvents (NCMS, 2011).

Step 4: Paint use (during service life of the paint)

This step contains impacts caused during the service life of the paint. This includes processes and materials utilized for maintenance and repainting (NCMS, 2011).

Step 5: Paint End-of-life

In the final step, impacts due to paint disposal are included. This consists mainly of impacts from the waste management, e.g. landfill and incineration (NCMS, 2011).

Along this life cycle there are also distributions of the raw materials and the products (Dcarbon, 2008).

By accounting for the environmental impacts occurring in these phases an environmental profile of the paint can be generated. LCA is an appropriate method for assessing the environmental impacts of paints according to Chemiewinkel, Enterprise Ireland and WIMM (2000).

3.4.2 Environmental issues of paint

The aim of this subsection is not to map out every environmental issue connected to the life cycle of outdoor decorative paints. It is instead to identify major concerns that are being highlighted in the literature as important to manage.

According to Sveff (2011) paint should be viewed as a product that gives environmental benefits because it prolongs the life length of the substrate, which is often accountable for much larger environmental impacts. The environmental impact of paint should however not be negligible as impacts occur in all stages of the life cycle according to Carlstedt Sylwan (2002), from raw material extraction to the end-of-life phase. In the recent years paint producers have implemented voluntary and applied to regulatory measures in order to lower the environmental impacts, stated by Sveff (2013) and CEPE (2012), where positive trends have been shown (CEPE, 2012). Also increasing customer demands have accelerated the innovations for paints including smaller environmental footprints (Chemiewinkel, Enterprise Ireland and WIMM, 2000). Since there are many decorative paints with different compositions and production techniques, it is difficult to find a standard paint formulation for the baseline of the environmental performance (Kougoulis et al., 2012; Chemiewinkel, Enterprise Ireland and WIMM, 2000). This subsection will therefore identify the major environmental issues for paint products and aspects affecting these, but also reduction potentials of these and how to compare environmental impacts of paints.

Sveff (2013), Kougoulis et al. (2012), Dobson (1996) and Chemiewinkel, Enterprise Ireland and WIMM (2000) identify several important indicators for the environmental performance of paints including emission, waste, energy usage, solvent type in the paint, the fate of the paint and the importance of environmentally certified products. Some aspects that affecting these impacts are the paint´s life length and amount required. When summarizing these commonly stated environmental issues and aspects affecting these, they can be identified as:

- Life length of paint (time between repainting) and amount used (amount of paint necessary to reach a predefined painting quality)
- Paint ingredients and their fate in the end-of-life phase.
- Raw material- and paint production and waste produced.

Environmental impacts from paint distribution is considered to be low according to Kougoulis et al. (2012) and Axelsson et al. (1999), but there is however concern about the risk of accidents during transportation where often hazardous materials are transported (CEPE, 2013a). The characteristics of a good paint from an environmental perspective is thus: small amount required with long life time, non-harmful substances included with proper waste management, low impacts from the various production processes and waste generated and a safe distribution chain.

Life length of a paint and amount used

This is not an environmental issue in itself, but is important to consider as it affects other environmental issues. A critical phase in a paint´s life cycle is the use phase. A large responsibility is put on the user of the paint, who will determine how much paint that is needed and how often repainting is conducted (Kougoulis et al., 2012). Kougoulis et al. (2012) claims that the parameter that can contribute the most to a reduced environmental impact is the paint´s life time. Increased life time can be reached by better informed end-consumers regarding for instance paint application and maintenance. Also increased quality of the paint could lead to extended life time (Kougoulis et al., 2012; The Nordic Swan, 2013).

Regarding the amount paint used in the use phase, there are large potentials for decreasing the environmental impacts. There have been investigations, studying the amount of paint wastage (paint not used in the can) estimating up to 25 % of all paint goes unused (Kougoulis et al., 2012; Lee, Vaughan and Willis, 2011). This cause approximately 12 % of the environmental burden from paints. A reduction of the amount unused paint would thereby significantly reduce the environmental footprint. Possible methods to achieve the reduction include:

- More appropriate quantities of paint in each can (i.e. more can sizes to choose between or some kind of dosing system) (Kougoulis et al., 2012).
- Better information to the end-consumers about amount of paint required (Kougoulis et al., 2012).
- Take-back schemes that could limit the wasted paint (resell or recycle it) (Dcarbon, 2008).

- Information and Training Campaigns (education for professional painters in order to use the products more efficiently and thereby reduce the consumption and impacts) (Chemiewinkel, Enterprise Ireland and WIMM, 2000)

Paint ingredients and its fate

The composition of paint is determined by several factors, such as technical possibilities, governmental regulations and customer demands. This type of environmental issue is often regarded as high importance since many traditional paint ingredients are environmental harmful and toxic to humans (Kougoulis et al., 2012; Overbeek et al., 2003).

Solvents

One of the main environmental issues in the paint industry has been the composition of the solvents. Since solvents evaporate directly to air after attachment of the paint, non-harmful substances are desired. It has been an ongoing transition from organic based solvents to water based solvents since they have lower VOC content (Overbeek et al., 2003). This is a family of substances, reacting with oxygen in the presence of sunlight forming ground level ozone, but they do also contribute to global warming (ICI Paints, 2009). The decorative paint industry (indoor and outdoor paints included) contributed 15 years ago with about 5 % of the total man-made VOC emissions in the European Union (Chemiewinkel, Enterprise Ireland and WIMM, 2000). Since then more stringent regulations have been in forced in the paint industry which has put large pressure on the actors to develop products with low VOC content (CEPE, 2012). One major challenge for decorative paints regarding VOC emissions in the use phase is that they do not occur in a controlled environment and cannot be captured, as occurring for industrial coatings (Dobson, 1996). Several members from the European paint branch organization, CEPE, believe that today´s VOC level represent the practical limit for technical feasibility without compromising quality and usability of paints. If lower limits are in forced without compliance from the industry, counter-effects could be the case including reduced life time of paints (CEPE, 2012). A strategy suggested by University of York (2013) to lower the VOC content is to increase the solid content in the paint implying that less solvent is used. The problem is however that the viscosity increases which hampers proper application of the paint (University of York, 2013). Positive results from tests show however that the best performing paints are water based, which are often the most environmentally friendly alternative (Hjort, 2012; European Commission, n.d). In connection to this, a large majority of the paints sold on the market are water based with a trend towards even more water based products (Sveff, 2013; European Commission, n.d, Chemiewinkel, Enterprise Ireland and WIMM, 2000).

Nanomaterial

Today there is a concern about the inclusion of nanomaterials in products, independent of the industrial sector (Wick, Krug and Nowackm, 2011). The main concerns of nanomaterials are connected to its small size and the unclear picture of the health and environmental risks of nanomaterials (Kougoulis et al., 2012). One major challenge to determine the impacts is that harmless bulk materials can pose toxic effects on nanoscale. The severity varies between different substances since they vary in size, purity, crystalline form, porosity, surface area and how they are modified (Wick, Krug and Nowackm, 2011).

Facade paints are under constant influence of microorganisms, traffic exhaust and weathering. By applying nanotechnology, these effects can be counteracted (Wick, Krug and Nowackm, 2011). Properties that nanomaterials provide for paints are according to Som et al. (2013) and Wick, Krug and Nowackm (2011) for instance antimicrobial, UV protection, scratch resistance and self-cleaning and many companies expect those paints to have longer life time than conventional paints (SAFENANO, 2014). Nanomaterial can also bring sustainability advantages to the paint industry including replacing hazardous substances having an even greater impact on the environment and health, reduced generation of hazardous waste and reduced energy consumption (Wick, Krug and Nowackm, 2011; EMPA, 2014; Som et al., 2013).

For the group of nanomaterials called ENM (engineered nanomaterial) there are large potentials when considering the above stated opportunities but also risks connected to the use (Wick, Krug and Nowackm, 2011; Zuin et al., 2014). So far, this has been studied to a limited extent according to Zuin et al. (2014) but there is ongoing research in the area (Kougoulis et al., 2012). For example the European Union funded research project NanoHouse aims to evaluate the environmental and health risks of ENMs applied on facade buildings (SAFENANO, 2014). This is to understand whether the potential benefits can outbalance the risks (EMPA, 2014). When evaluate products including ENMs, one should also consider products not including ENMs in the assessment (Kougoulis et al., 2012; Hischier, 2014). LCA is expected to be an appropriate method to evaluate the environmental impacts of nanomaterials, but unfortunately these substances have not been adopted for such use according to Hischier (2014). No public databases exist for inventory data for any type of ENM. Moreover, characterisation factors are lacking, which would be necessary on order to determine the impacts from different ENMs. As a consequence very few LCA studies are published in the area of nanotechnology. Most of them are incomplete and incomprehensive, due to lack of inventory data and characterisation factors for the nanomaterials (Hischier, 2014).

According to Wick, Krug and Nowackm (2011) there are large uncertainties about the effects of ENMs, much due to standardized methods and instruments to measure the exposure are lacking. This has implied lack of regulations, which have made it difficult for the paint industry to know which standpoint to take in this issue. The research project NanoHouse reached the conclusion that several factors determine the risks of ENMs (Wick, Krug and Nowackm, 2011). Since risk includes both probability and consequence (Burgman, 2007), one needs to study the probability that ENMs are released to the environment and the subsequent effects of a release (Wick, Krug and Nowackm, 2011). The releases of ENMs in paints depend on the type of bond between the ENM, the coating matrix and the substrate. ENMs may be loosely embedded or strongly bound to the matrix material. Silica, which is one type of ENM, reacts chemically with the binder forming a strong network with the polymers (silicate polymers). A release of silica from the paint film is expected to be very low and such paints can be viewed as ENM-free according to Wick, Krug and Nowackm (2011). A low rate of nanoparticles is however expected to be released to the environment. The releases are in most cases as large paint particles, which reduce the nano-scale effects (SAFENANO, 2014). A study by Zuin et al. (2014) concluded that the release of silica differs for different paint formulations.

Considering the consequences of a release, acute effects from ENMs are not expected but long term effects can be considered due to bioaccumulation in cells according to

Wick, Krug and Nowackm (2011). Regarding the impacts from ENM-silica, they differ for example depending on which form of silica that is studied. It can be present in amorphous and crystalline form, where amorphous silica is used in industrial applications. Only unrealistic high doses of amorphous silica can pose effects to the environment and human health. Otherwise, it does not pose any significant effects (Wick, Krug and Nowackm, 2011). An inclusion of silica in paint products instead of other substances can give environmental benefits as more environmental regulations are expected to boost (AkzoNobel, 2013c). The inclusion of silica in paint need however to be further examined since there is a toxicologically concern about such materials (Zuin et al., 2014).

In many cases, products including ENMs that are better from an environmental point of view are considered as bad because they contain ENMs. This reflects the use of nanomaterials in general and it is only the societal acceptance, involving the guarantee of their safety for humans and the environment that can make nanomaterials become popular in the paint industry (EMPA, 2014; Golanski et al., 2013).

Potent chemicals

Another environmental issue regarding the content of paints is to remove substances that are extremely harmful when they are exposed to the environment and humans. Examples of such substances are: Alkylphenolethoxylates (APEOs), Perfluorinated alkyl sulfonates (PFAS), Formaldehyde, Halogenated organic solvents, Phthalates, Heavy metals and Isothiazolinone compounds (Kougoulis et al., 2012; The Nordic Swan, 2013; European Commission, 2009b; Chemiewinkel, Enterprise Ireland and WIMM, 2000).

Environmental fate

Since paints exist close to humans and the environment, it is important to consider the fate of the paint ingredients (Kougoulis et al., 2012). Some of the components are released during the use phase (e.g. VOC and VAC) (The Nordic Swan, 2013). Others are released or captured in the waste management of the paint e.g. through thermal incineration or through landfill, where over 90 % of all water based paints in Sweden are incinerated and 10 % are landfilled (Sundqvist and Palm, 2010). It is important to use waste management treatment methods that reduce the environmental impacts from paints (Zuin et al., 2014). Kougoulis et al. (2012) is estimating that about 40 % of the environmental impacts from paints arise in the end-of-life phase. In order to avoid treatment of hazardous and harmful substances and to lower these impacts, a transition to natural biodegradable and bio-based material is suggested for the paint industry (Dcarbon, 2008; CEPE, 2013a). This could include all components in the paint (CEPE, 2013a). A special interest is put on water based paints which often end up in the sewage system or surface water resulting in water pollution. The reason for this is that end-consumers tend to clean their brushes under tap (Chemiewinkel, Enterprise Ireland and WIMM, 2000).

Raw material- and paint production and waste produced

This issue includes the environmental impacts during the various steps of the production of paints e.g. raw material production and paint manufacturing. This issue has been shown accountable for a large contribution of the environmental impacts according to Dcarbon (2008) and Kougoulis et al. (2012). Below follows two examples of this aspect outlined in the literature.

Titanium dioxide

One aspect in the raw material production that is recurrent is the production of titanium dioxide, as it is the most commonly used substance in pigments (Chemiewinkel, Enterprise Ireland and WIMM, 2000; Royal Society of Chemistry, 2005; Dobson, 1996). Chemiewinkel, Enterprise Ireland and WIMM (2000) identifies production of titanium dioxide as one of the major sources of environmental effects for paints. Titanium dioxide can be produced by any of the two chemical processes, the sulfate- or the chloride process. It is produced by a reaction of different feedstock of titanium ore with either chlorine gas or sulphuric acid depending on the process used. Large quantities of solid & acid waste and dust from titanium production have led to increased regulatory attention regarding the production (European Commission, 2014b; Royal Society of Chemistry, 2005). The environmental burden from the production depends largely on which type of process and feedstock that are used and how the waste from the production is treated (Reck and Richards, 1997). The importance of low environmental impacts from the titanium dioxide production is also acknowledged in several eco-labels for paints, e.g. The Nordic Swan and EU Ecolabel (The Nordic Swan, 2013; European Commission, 2009b). Titanium dioxide is however expensive, which force the producers to reduce its content in paints (Chemiewinkel, Enterprise Ireland and WIMM, 2000).

Production environment

For the manufacturing of paint, the emissions can be kept low as they occur in a controlled environment. The major factors affecting the emissions in the paint production are: which solvent types are used and the mixing temperature. Under proper conditions about 1 % to 3 % of the solvents are emitted and for uncontrolled conditions the number can be 15 % (European Commission, n.d).

Environmental comparison between products

Several environmental concerns are presented above, which can be assessed by using the environmental system analysis method, LCA. Different types of paints do have different environmental impacts, which are not additively. Valuation of their relative importance could be required. Weighting methods within LCA can be used for this purpose by aggregating the impacts to a single environmental metric. This procedure is presented in Sections 3.1 and 3.2 and will be applied in the case study in Chapter 4.

There is also a need for comparison between products included in a study. Comparability of products in the paint industry is however disputable and often a difficult task according to Chemiewinkel, Enterprise Ireland and WIMM (2000). In order to compare the environmental profiles of paints, a functional unit must be defined. There are two critical aspects that must be captured for paints, coverage and durability. The coverage determines the amount of paint used per square meter facade, at the optimal thickness of the dry paint film. Durability determines the period of time the paint film will last. In most paint LCAs the coverage is defined as the technical required film thickness and is considered equal for all paints. There are however different opinions among the experts in this issue. Depending on the film thickness chosen, the environmental impacts would differ since film thickness and amount paint used per square meter facade correlate strongly. Considering the durability of the paint, it depends on the technical lifespan of the paint. The durability is often measured through accelerated aging tests, where the paint is exposed to extreme weather conditions (APM Testing, 2013). Repainting is however often conducted due to aesthetic reasons, making the technical and actual life time differ. By increasing the life time of paints, the environmental impacts decrease since less is consumed (Chemiewinkel, Enterprise Ireland and WIMM, 2000). Chemiewinkel, Enterprise Ireland and WIMM (2000) concludes that the most appropriate functional unit for paint would be "the amount of paint necessary to paint a certain area in the technically demanded dry paint film thickness over a certain period of time".

3.4.3 Eco-labeling of paint

It is from the application area of communication within LCA environmentally labeling and declaration of products have arisen. This is used to communicate the environmental performance of products to stakeholders (Baumann and Tillman, 2004; Del Borghi, 2012). The International Organization for Standardization (ISO) has classified the labels into three types (Del Borghi, 2012):

Type I: A voluntary, third party program awarding a license that authorizes the use of environmental labels (eco-labels) on products indicating the environmental performance of a product within a particular product category based on life cycle considerations.

Type II: Informative environmental self-declaration claims.

Type III: Voluntary programs providing quantified environmental data of a product for pre-set categories of parameters (environmental declaration). This is set and verified by qualified third parties and based on LCA.

(Global Ecolabelling Network, 2014c)

The type I category include labels that probably most end-consumers come in contact with. Examples of labels are The Nordic Swan (Scandinavia), EU Ecolabel (EU), Green Seal (US), Blue Angel (Germany) (Amacher, Koskela and Ollikainen, 2004). These labels cover broad categories of products e.g. paints, cloths and cleaning products (Global Ecolabelling Network, 2014b). A firm that wants to use a specific eco-label for a certain product must for that product comply with the environmental criteria. Third party organizations control that the product fulfill the criteria, which differs for different product groups. The firm needs to update their license to more stringent requirements with a few years in between. The costs for the firm are both in terms of the license fee and the possible required eco-improvements of the product (Amacher, Koskela and Ollikainen, 2004; Global Ecolabelling Network, 2004). According to Kougoulis et al. (2012) these are examples of issues that could be considered in eco-labeling of paint products:

Volatile organic compound (VOC), heavy metal compounds, titanium dioxide production, reactive solvents, alkylphenol ethoxylates (APEO), aromatic hydrocarbons, formaldehyde, nanomaterials, packaging, disposal, fitness for use and end user information.

There is an ongoing growth of interest in eco-labeling globally as a market based method to promote sustainability (Global Ecolabelling Network, 2014a). This is done by communicating the environmental performance of products and services to stakeholders, via the eco-label (Del Borghi, 2012). Eco-labels are well established by end-consumers and often increase their willingness to pay. From the firm´s perspective, this can be seen as an important strategic variable, differentiates their product from competitors´ products by an eco-label (Amacher, Koskela and Ollikainen, 2004). This has shown to be a good method for reducing the environmental impacts of paints (Chemiewinkel, Enterprise Ireland and WIMM, 2000).

The following part will elaborate upon the environmental criteria for two labels (The Nordic Swan and EU Ecolabel) that are used by the specific paints in the case study.

The Nordic Swan and EU Ecolabel

The Nordic Swan and the EU Ecolabel are two similar type one environmental labels, both voluntary and positive labels without industry or beneficial interest (The Nordic Swan, 2014c; The Nordic Swan, 2014a). The Nordic Swan is the official environmental label for the nordic countries and EU Ecolabel for the European Union. Products and services can have their respective environmental labels by fulfilling certain environmental criteria based on LCA according to The Nordic Swan (2014a) and The Nordic Swan (2014b) conducted by independent experts (European Commission, 2014a). The criteria have been developed and agreed upon by scientists, NGOs and stakeholders in order to be credible and reliable (European Commission, 2014a). The criteria for The Nordic Swan and the EU Ecolabel are set for different product categories based on their application. The criterion category for the outdoor paints in the EU Ecolabel is *Outdoor paint and varnishes* and for The Nordic Swan it is *Chemical building products* (European Commission, 2009a; The Nordic Swan, 2013).

For both the Nordic Swan and the EU Ecolabel the criteria are set from a life cycle perspective including environmental impacts origin from the manufacturing phase, the use phase and the end-of-life phase. Table 11 below intends to map out the specific criteria included for these steps for the two eco-labels. The aim is not to point out each and every substance limitation but to give a general overview of the criteria. This will clarify what can be expected regarding the environmental impacts and quality requirements for the two decorative paints in the case study, since they hold the eco-labels.

Table 11 – Criteria and expectations for the Nordic Swan and EU Ecolabel, based on The Nordic Swan (2013), European Commission, (2009a) and European Commission (2009b).

<table>
<tr><th rowspan="2">Life cycle phase</th><th colspan="2">Criteria</th><th colspan="2">Expectation</th></tr>
<tr><th>EU Ecolabel</th><th>The Nordic Swan</th><th>EU Ecolabel</th><th>The Nordic Swan</th></tr>
<tr><td>Manufacturing</td><td colspan="2">Reduction of by-product hazardous waste (production of titanium dioxide)</td><td>Emissions and discharges of waste from the production of titanium dioxide may not exceed:
266 mg SO_2 emission/m^2 of dry film
19 g sulphate wastes/ m^2 of dry film
3.9-12.5 g chloride wastes per m^2 of dry film (depend on the titanium ore used)</td><td>If product contains more than 3 w% titanium dioxide, emissions from the production of titanium dioxide must not exceed:
7.0 kg SO_2/ton TiO_2
500 kg sulphate waste/ton TiO_2
103-329 kg chloride waste/ton TiO_2 (depends on the titanium ore used)</td></tr>
<tr><td rowspan="2">Use</td><td colspan="2">Limitation of air pollutants and solvents</td><td>Volatile Organic Compounds (VOCs): Differ between paints but about: 15-100 g/l (inc. water).
Volatile Aromatic Hydrocarbons (VAC)s: < 0.01 % of the end product.</td><td>VOCs: Differ between paints but about: 15-100 g/l (inc. water)
VACs: < 0.01 % of the end product</td></tr>
<tr><td colspan="2">Performance requirements</td><td>Fulfilling ISO related requirements regarding the following tests:
Spreading rate, Resistance to water, Adhesion, Abrasion, Weathering, Water vapour permeability, Liquid water permeability, Fungal resistance, Crack bridging, Alkali resistance.</td><td>Fulfillment of all requirements in the standard DS/ EN ISO/IEC 17025 regarding laboratory tests: Weathering test, Water Vapour Permeability, Liquid Water Permeability, Resistance to fungal growth.</td></tr>
<tr><td rowspan="8">End-of-life</td><td colspan="2" rowspan="8">Limitations of the use of substances dangerous for the environment and health</td><td colspan="2">Examples of restricted substances:</td></tr>
<tr><td>Formaldehyde: max 0.001 w% in end product
Isothiazolinone compounds max 0.2 w%: in end product</td><td>Formaldehyde: max 0.001 w% in end product
Isothiazolinone compounds max 0.2 w% in end product
Iodopropynyl butylcarbamate (IPBC) max 0.450 w% in the end product</td></tr>
<tr><td colspan="2">Example of banned substances:</td></tr>
<tr><td>Alkylphenolethoxylates (APEOs), Perfluorinated alkyl sulfonates (PFAS), Halogenated Organic Solvents, Phthalates.</td><td>Alkylphenol ethoxylates (APEO)
Halogenated organic compounds
Phthalates, Isocyanates, Bisphenol-A compounds, Fragrance.</td></tr>
<tr><td colspan="2">Heavy metals banned:</td></tr>
<tr><td>cadmium, lead, chromium VI, mercury, arsenic, barium , selenium, antimony.</td><td>heavy metals, cadmium, lead, chromium VI, mercury, arsenic, barium, selenium, antimony.</td></tr>
<tr><td colspan="2">Nanoparticles:</td></tr>
<tr><td>Not considered</td><td>May not be added, unless they will not entail health- or environmental problems. Particle sizes smaller than 100 nm are considered.</td></tr>
</table>

There are also requirements on e.g. the end-consumer information, packaging and return system. The requirements do also include more substances and specifications for different types of paints included in each label. The above stated criteria are of specific importance in the paint industry nowadays. The major issues are covered in the Subsections 3.4.2.

3.4.4 AkzoNobel Eco-premium solutions

The demand for products to be sustainable from an environmental point of view has grown. The industry does now need to apply to these demands (Palme, 2011). The sustainability achievements of companies depend on the sustainability performance of their product portfolio (Widheden, Petersson and Ringström, 2011). To be able to improve the products´ sustainability performances, one needs to assess them. An example of how AkzoNobel works with this matter is through the in-house developed method Eco-premium solutions (EPS[10]). In this method, product performances are evaluated in terms of health, safety and environmental aspects. EPS acts as a compass to develop and market the more sustainable solutions required for the future (AkzoNobel, 2013a).

When evaluating if a product is EPS, one needs to compare it with another product according to AkzoNobel (2013a). This should be the mainstream solution on the market, i.e. the most commonly available product fulfilling the same function or customer benefit. The assessment of the intended EPS product and the mainstream product can be made qualitatively or quantitatively. There are several criteria that should be fulfilled by the EPS product compared to the mainstream solution, which is stated below. This is conducted from a life cycle perspective across the value chain to avoid suboptimizations (AkzoNobel, 2013a).

(1) Providing the same or better functionality or benefit for the customer application.

(2) The EPS product should perform at least 10 % better in at least one of the criterion below, and not performing more than 10 % worse for any of the criterion.

- Energy efficiency
- Use of natural resources/raw materials
- Emissions and waste
- Toxicity (human and eco-toxicity)
- Risks (for accidents during production, transportation etc.)
- Land use
- Health/wellbeing

Within the EPS, there is a subset named EPS with downstream benefits. Products with downstream benefits perform significantly better in at least one of the sustainability criterion against the mainstream, to one or more actors after AkzoNobel in the value chain. This benefit can be expressed by e.g. customers (less energy consumption), the end user (less toxic substances included, lower repainting frequency), waste management (biodegradable products) (AkzoNobel, 2013a).

Both EPS and EPS with downstream benefits are measured in percent of external sale revenue. The target lines for these are 30 % until 2015 for EPS and 20 % until 2020 for EPS with downstream benefits. It should be stressed that EPS is not marketed outside AkzoNobel, as an environmental label. It should rather acts as a catalyst for the company to promote more sustainable solutions (AkzoNobel, 2013a).

[10] Shall not be confused with the weighting method Environmental Priority Strategies (EPS).

4 Case study

This chapter contains the case study for which two paints, a primer, an impregnator, a paint wash and a maintenance wash are evaluated in Life Cycle Assessments (LCAs), where monetary weighting is included. The chapter starts by introducing the LCA case study followed by the goal and scope of the LCA, finalizing with the results.

4.1 Description of the case study

The paints examined in the LCA are two water based acrylic outdoor decorative paints. These are covering paints applied on wood facades produced in Sweden. The paints are produced with or without a specific AkzoNobel produced modified colloidal silica.

The inclusion of modified colloidal silica represents the major difference between the paints. This implies that the amounts of some other substances are lower in the paint with modified colloidal silica. By including silica in the product formulation, higher downstream performances are expected. This can for instance be self-cleaning properties, increased resistance to mould growth and dirt resistance which can imply longer time between repainting and/or washing the facade. An inclusion of colloidal silica might also lower the need for other more hazardous chemicals, which can give environmental benefits.

Apart from the paints, there are need for preparation- and maintenance work forming a paint system. Preparation work include impregnate and priming the facade. Maintenance work includes washing the facade where the products paint wash and maintenance wash are used. For the two paints in the case study, the same preparation- and maintenance products are utilized. In addition, environmental assessments of all products´ packaging materials are performed. The focus in the study is however on the two paints.

The paints are compared on the basis as they fulfill the same functionality, e.g. coverage and protection of the facade. This is described in Section 4.3, *functional unit*. In the use phase, there might be differences in for instance how often the facade needs to be repainted and/or washed. Repainting and washing are varied in different scenarios, presented in Section 4.3, *scenarios and assumptions*.

The case study is conducted by using the environmental assessment method LCA in order to identify and evaluate the environmental impacts from the products. The LCA software tool GaBi6 is utilized in the modelling to simplify and structure the analysis. In the Life Cycle Impact Assessment (LCIA), weighting by the monetary methods, Environmental Priority Strategies (EPS), Ecovalue and Stepwise are used. The results are expressed in economic units and represent the external environmental costs.

From now on the paint system with paint **including** colloidal silica and including maintenance- and preparation products will be referred to as **Paint system A**. The paint system **excluding** colloidal silica and including maintenance- and preparation products will be referred to as **Paint system B.**

Table 12 – Included products in the Paint systems.

Products	Paint system A	Paint system B
Paint A (with colloidal silica)	X	
Paint B (without colloidal silica)		X
Primer	X	X
Impregnator	X	X
Maintenance wash	X	X
Paint wash	X	X

The above Table 12 shows the products included in paint systems A and B. Included products are market with an X.

4.2 Goal for the case study

The goal in an LCA aims to set the context of the study, where the application, purpose, objectives and questions to be answered can be stated.

The goal of this LCA is to assess the environmental impacts and environmental costs for the two paint systems with a focus on the two outdoor decorative paints produced with or without modified colloidal silica. By using LCA, different environmental impacts are computed and converted to environmental costs. This is to provide a quantitative and clear picture of the life cycle, stating which paint option that is the most environmentally preferable. This is presented to involved project stakeholders in AkzoNobel with the goal to learn about the environmental costs of the products, in order to be able to reduce these costs. The result of the study is intended for internal AkzoNobel use.

4.3 Scope of the case study

In LCA, the scope is defined to set the modelling aspects of the study. The scope defines what to include and what to exclude from the study, which have implications for the result of the study. All modelling aspects of the study are presented in this section, including a short description of each aspect. More elaborate descriptions for some of the aspects are presented in Subsection 3.1.2.

Functional unit

The functional unit is critical for an LCA study. It is the basis for comparison and the unit for which all flows relevant for the study are related to. Important parameters to cover in the functional unit for paints are coverage and durability. There exist uncertainties for these parameters. Coverage is difficult to determine because it depends largely on the thickness of the coat. Different people painting will lead to different coat thicknesses. Durability on the other hand is difficult to estimate since the technical and the actual life time of the paint often differ due to end-consumer behaviours (Chemiewinkel, Enterprise Ireland and WIMM, 2000).

The functional unit in this study is defined as 1 $m^2 * yr$ paint covering and protecting the facade to a satisfying end-user demand, in the use phase (reference flow). This is the basis for comparison of all environmental interventions and impacts as well as monetary weighted values.

Neither examination of coat thickness nor end-user demands is measured in this study. Instead recommendations from the product supplier regarding quantity of products used per m^2 facade and their durability are utilized. However, durability is varied in the different scenarios further described in *scenarios and assumptions* below.

Choice of impact assessment

This modelling aspect present the environmental impacts considered in the study and how the results intend to be presented. The impact assessment in this study will include tracking inventory data (e.g. raw material used, emission and waste produced) arising in the life cycle of the products. For the paint wash, maintenance wash and impregnator, cradle-to-gate analyses are performed, while cradle-to-grave analyses are conducted for the paints and the primer. Also the products´ packaging materials are modelled cradle-to-grave. The inventory data is transferred to environmental impacts and further aggregated by weighting to environmental costs. This is conducted by the three monetary weighting methods: EPS, Stepwise and Ecovalue decided upon in Subsection 3.2.4.

Type of LCA

In LCA methodology, one usually refers to two types of LCAs, attributional and consequential. Depending of which one is used, it implies consequences for how the LCA is modelled, e.g. collection of data, allocations.

In this study, an attributional LCA is performed. This choice is based on that the purpose of the assessment is to get comparative quantitative information about the environmental impacts and environmental costs throughout the entire life cycle for the two paints.

Flowchart for the LCA

Figure 18 below illustrates the flowchart of the two paint systems, A and B. The difference between the product systems is which raw materials that are used for the paints. The circled right box in the top of the figure represents the raw materials required for paint A, while the left box represents raw materials required for paint B.

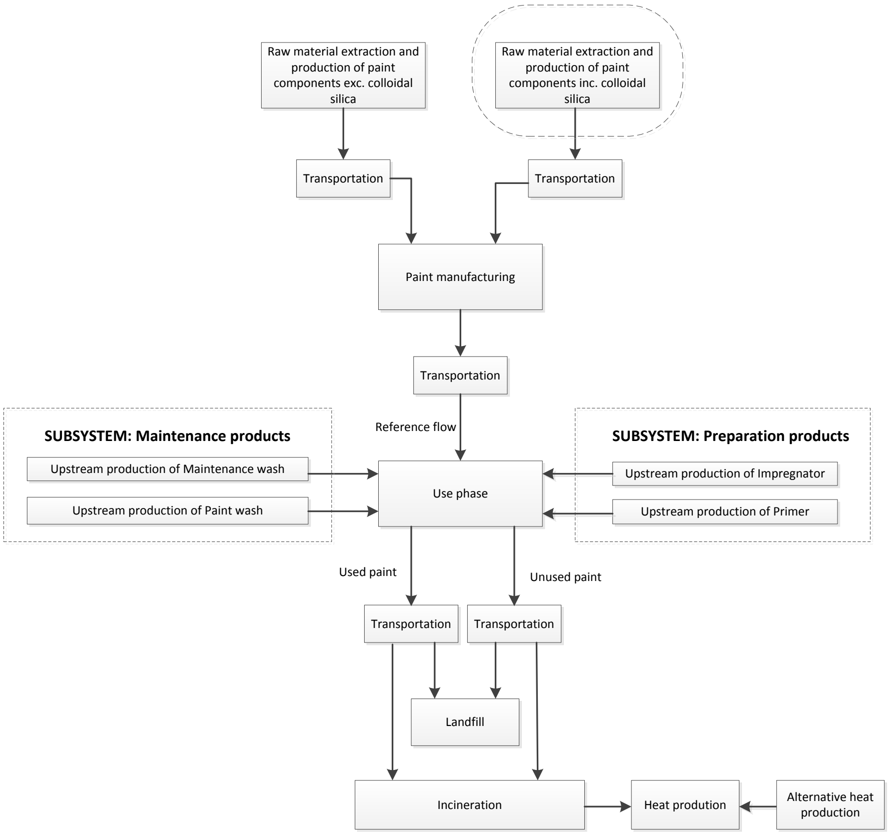

Figure 18 – Flowchart of the life cycle of the two paints, including preparation- and maintenance products. Packaging are included for all products but is not shown in the figure.

The main system in the above Figure 18 contains the paints and the life cycle starts at the top of the figure with raw material extraction and production. The raw materials are refined to paint and further transported to the use phase and finally to landfill or incineration in the end-of-life. For the incineration process, a system expansion is performed, where alternative heat production is included. Moreover, there are two subsystems in the flowchart comprising of maintenance products and preparation products. Not showed in this flowchart are the packaging products which are included for all products and modelled cradle-to-grave where recycling and incineration are performed in the end-of-life phase.

Scenarios and assumptions

When modelling a system for a future situation, scenarios can be used to illustrate this. Scenarios are created aiming to enable comparison between the paints and to capture different possible product outcomes, e.g. varying life time of the paints. A number of assumptions regarding the scenarios are presented below. The assumptions are based on communication with the reference group if no literature reference is given. This is followed by the scenarios for the case study.

Assumptions for the paints

- The functionality of the paints is high enough[11] during their lifetimes (enable comparison between the paints). Repainting is performed when the paints do not fulfill the demands from the end-consumer. This is based on the paint supplier´s recommendations regarding the products´ durability.

- The default durability of Paint A is 16 years and for Paint B it is 12 years. These are measured through accelerated aging tests, where the applied paint is exposed to extreme weather conditions.

- Default paint coverage on a new facade per coat is 5 m^2/l. Two coatings are assumed for both paints.
- Default repainting coverage per coat is 7 m^2/l. Two coatings are assumed for both paints.
- Assume that 10 % of the paint produced is never applied on the facade. This is based on estimations by Kougoulis et al. (2012) and Lee, Vaughan and Willis (2011).
- Assume the same color is used during the building´s life time (to avoid overcoating).
- Neglect spot repainting of the facade and assume repainting at cyclic occasions.
- The waste management of used and unused paints include incineration and landfill, occurring either when the building reaches its end-of-life or when the unused paint no longer can be used. This is based on Sundqvist and Palm (2010) stating that about 90 % of all water based paints in Sweden are incinerated and about 10 % are landfilled.
- It is assumed that there are no raw material losses in the paint production.

Assumptions for the maintenance products (paint wash & maintenance wash)

- Paint wash is used only when repainting.
- Coverage of paint wash is 60 m^2/kg.
- Maintenance wash is used regularly (not in connection to repainting).
- Coverage of maintenance wash is 60 m^2/kg.

Assumptions for the preparation products (impregnator & primer)

- Preparation products are only used when painting a new facade.
- Default coverage of impregnate per coat is 5 m^2/l. One coat is assumed.
- Default coverage of priming per coat is 6 m^2/l. One coat is assumed.

Assumptions for the packaging products

- The packaging materials for all the included products are polypropylene (PP). Around 40 % is recycled, 60 % is incinerated and landfill is neglected since it is less than 5 % in Sweden, based on PlasticsEurope (2014).
- The weights of these packaging products are calculated based on the properties e.g. density, geometry and with Pyr AB (2013) as a guidance. The amount of polypropylene required for paint, primer and impregnator is 35 grams per liter and for maintenance wash and paint wash it is 90 grams per

[11] The function shall meet the demands from the end-consumer, which are for instance: color, gloss, coverage, protection, low amount mould growth.

liter. The difference depends mainly on that the latter products are sold in smaller packages.

Assumptions for transportations

- Environmental assessment of transports is presented separately for the products and their packaging.
- All transportations occur with either truck (40 ton total weight, Euro 3, 70 % filling rate) or ship (>8000 deadweight tonnage, 50-60 % filling rate), where data was gathered from NTM (Nätverket för transporter och data).
- Transports for both products and packaging are accounted for. This has been divided into three parts: (1) transportations of raw material to the manufacturing, (2) transportations of products & packaging to the use phase and (3) transportations of paint, primer and packaging to waste management.
- Transportations for the products and packaging have been collected from AkzoNobel internally.

Other assumptions

- Assume that the building for which the products are applied on stands 100 years.
- Assume no environmental or technical improvements are implemented in any of the products in the future (e.g. less environmental impact, longer duration time).
- Assume repainting and washing 33 % less often with the paint where modified colloidal silica is included (Paint A), due to higher end-consumer benefits in terms of expected longer life time.
- Same amount of paint, preparation- and maintenance products per m^2 facade are used for Paint system A and Paint system B.

The following scenarios are performed for each paint system, (see Table 13). The scenarios intend to capture different outcomes from the durability parameter, by changing painting- and washing frequency. Scenario 1 and Scenario 2 can be viewed as extreme points while Scenario 3 lay between. Scenario 3 will acts as a baseline scenario.

Table 13 – Scenarios for Paint system A and Paint system B.

Scenario	Time between repainting [yrs]		Time between washing [yrs]			
	Paint A (with colloidal silica)	Paint B (without colloidal silica)	Paint wash		Maintenance wash	
	Paint system A	Paint system B	Paint system A	Paint system B	Paint system A	Paint system B
Scenario 1	8	6	8	6	8	6
Scenario 2	20	15	20	15	20	15
Scenario 3 (baseline)	16	12	16	12	16	12

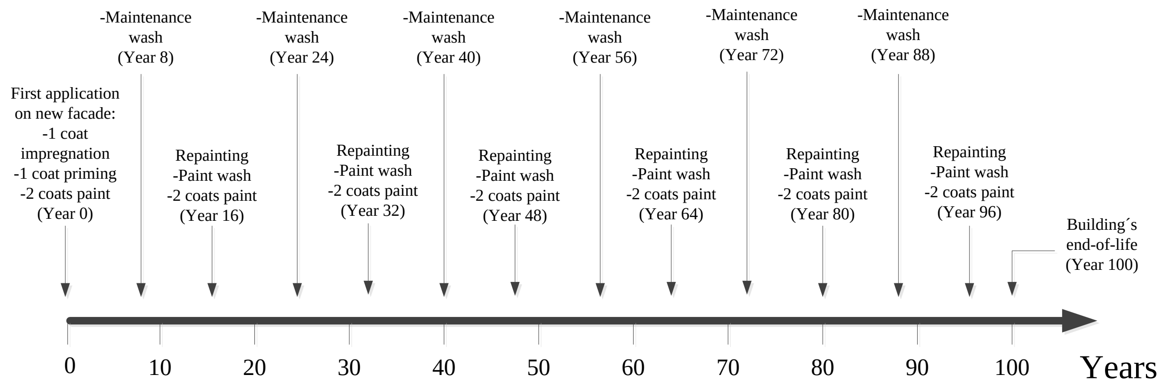

Figure 19 – Time line for the building and included products in paint system A for Scenario 3.

Figure 19 above illustrates the time line for the above scenario 3 with paint system A. This is to give a more illustrative picture of the user phase. All products included in the paint system (paint, preparation- and maintenance products) are included in Figure 19. At year 0, the first application on the new facade is performed, by using 1 coat impregnator, 1 coat primer and 2 coats paint. Maintenance wash occurs every 16th year and likewise for repainting & paint wash, but at different occasions as illustrated in Figure 19. This is assumed to be a repetitive procedure until the building reach its end-of-life. A basis for how many times each product will be applied on the facade during the life time of the building is created. This in combination with the required quantity of each product used per m^2 facade, stated in the assumptions above, enable calculation of the required amount of the products per functional unit. An example could be that 10 coats are required, with a paint quantity of 0.1 kg paint/m^2 facade. The paint used during the building´s life time spread on 1 year and 1 m^2 facade (i.e. the functional unit), are shown in the equation (7) below.

$$0.1\ \frac{\text{kg paint}}{\text{m}^2} * 10\ \text{times} * \frac{1}{100\ \text{years}} = 0.01\ \frac{\text{kg paint}}{\text{m}^2 * \text{yr}} \quad (7)$$

In the example, 0.01 kg paint is used per functional unit. The required quantities for the products in the different paint systems and scenarios are presented in Appendix A.

System boundaries

As the name indicates, the system boundaries define what to include in the technical system. This can for examples consist of temporal and geographical boundaries. Also significant and insignificant processes can be distinguished as well as boundaries between different technical systems can be identified.

Both the use phase and the end-of-life phase for the products in the paint systems occur in Sweden. The upstream raw material extraction and production are however occurring on a global level with subsequent transportation to Sweden for product manufacturing. The environmental impacts are not country specific why impacts are accounted for even if they occur outside the Swedish borders. For example, greenhouse gas emissions contribute to global warming which give effects on the global level, while for instance noise give impacts on local level (Dong, Laurent and Hauschild, 2013).

When considering the temporal boundaries for the impacts, there are no consensus in the research how long one should follow the impacts (Baumann and Tillman, 2004). The time horizon could differ from instantaneous to eternity effects depending on the impact category chosen (Goedkoop et al., 2008).

Boundaries could also be drawn between different nearby technical systems. In the case study, the building´s facade represents such a system. The facade is however not included in the system boundaries and it is assumed that the facade sustain as long as the building. This implies no need for painting on a new facade more than once, i.e. at year 0. The same goes for impregnation and priming. This is illustrated in the above Figure 19.

Allocations and avoided allocations

In an LCA study there could be a need to allocate the environmental impacts between product systems, but should be avoided if possible. One example where allocation could be necessary is if several products are produced from one process, where only one of the products is included in the studied product system. This is called multi output (Baumann and Tillman, 2004).

One allocation challenge occurring is at the end-of-life phase for the paints (also primer) and packaging (polypropylene). As these are incinerated, surplus heat are produced to the district heating system and less heat needs to be produced from alternative fuels. The energy generated per kg polypropylene and paint are about 20-40 MJ (Kubàt and Klason, 2002) and about 10 MJ (Sundqvist and Palm) respectively. There are several ways to allocate the environmental impacts from incineration between the technical systems. When performing an attributional LCA, it is recommended in Baumann and Tillman (2004) to use partitioning as allocation method. The impacts are thereby split between the technical systems by using a certain partitioning factor. Neither partitioning factor nor which part of the life cycle system that should be credited for the impacts are carved in stone. These choices can influence the result largely since a partitioning factor between 0 % and 100 % can be chosen. No appropriate factor was however found in the literature, why allocation was avoided through a system expansion. For the incineration of paint and polypropylene, system expansions are conducted where alternative heat production is modelled assuming alternative average Swedish heat production from waste in year 2010. When performing this type of system expansion, one subtracts the environmental impacts arisen for the production of the alternative heat from the impacts caused by the incineration. This can be displayed in equation (8) below, per MJ of incinerated materials.

$$\text{Allocated impacts } = \text{Impacts from incineration } - \text{Impacts from alternative heat production} \quad (8)$$

Another allocation issue that arises for the packaging is in the recycling of polypropylene. One allocation method called open loop recycling can be used when allocating environmental impacts from recycling (Baumann and Tillman, 2004). There are different variants of open loop recycling where it is possible to allocate the impacts between the different loops in the recycling. As only one allocation loop is accounted for in the recycling of polypropylene, it is assumed that this recycling loop takes place somewhere in between: after the first recycling loop and before the last recycling loop. In the first loop, the polypropylene is produced from virgin material and in the last loop the polypropylene is disposed. The encircled part of the recycling scheme is what has been accounted for, shown in Figure 20 below. The points indicate that there might be several recycling loops for polypropylene.

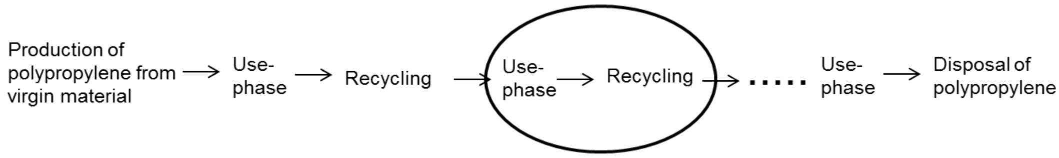

Figure 20 – Environmental impacts of polypropylene that is accounted for when it is recycled.

Data quality requirements

The data used in the LCAs are gathered from various sources. Data about the products are gathered within AkzoNobel internally. As the LCA is modelled in the LCA software GaBi6, some processes already modelled in AkzoNobel databases are utilized. These are modelled either by staff in AkzoNobel or by external sources such as PE International and Plastics Europe. Some processes are instead modelled by the LCA practitioner of the study. Data for the weighting part of the impact assessment is mainly collected from the literature, but also data from AkzoNobel databases are used. Other data and assumptions are either gathered from the literature, the reference group and the project supervisors or as own estimates.

There are several aspects of data as mentioned in Subsection 3.1.2, stated as relevance, reliability and accessibility. These three parameters have been kept in mind when data has been gathered. Since some of the data collected is confidential, the accessibility parameter has been affected. Regarding the first two parameters, data has been collected in such a way that it should represent the system as properly as possible. In some cases the data used in the modelling did not match the sought after data. Some examples of this are presented below:

- Specific emission factors for the two paints incinerated or landfilled in Sweden were not be found, instead generic emission factors for water based paints where used. The incineration and landfill are modelled for 15 years old Switzerland plants.
- A few raw materials could not be found why other similar raw materials were used. These contribute to a very small portion (about 0.1 %) of the product compositions.
- The emission factors for the incineration of plastic are obtained from Baumann and Tillman (2004), which used data from the 1990s that might be considered as too old.
- Average data (e.g. emissions and energy use) for the manufacturing of the included products (paint, primer, impregnator, paint wash and maintenance wash) are used. This since no product specific production information could be fetched.

Delimitations

In this part, the case specific delimitations are presented. This will clarify examples of related fields not studied and other aspects not included in the study.

- The paint wash, maintenance wash and impregnator are only modelled cradle-to-gate. This is both since data is not found regarding the waste management of these products and because the dissipative use is expected to be difficult to model.

- In the use phase, both the primer and the paints are expected to be used dissipative. Dissipative use is only modelled for the solvents, evaporating after attachment. Other dissipative use, e.g. paint peeled of the building is not modelled. Thus, all paint- and primer components except from the solvents are expected to be incinerated or landfilled in the end-of-life phase.
- In the weighting method Stepwise, the impact category Respiratory organics and Injuries (road and work) are not included, due to lack of characterisation data.
- In the weighting method, Ecovalue, the midpoint category abiotic resources is calculated in accordance with the CML method for oil, hard coal, iron and natural gas. This instead of using the Cummulative energy demand method.
- No sensitivity or uncertainty analyses are conducted. However variation analysis is included as different scenarios are used to check the robustness of the results.
- In the scenarios, only the durability parameter (repainting- and washing frequency) is varied. The coverage (kg products/m^2 facade) is not varied, mainly because there is no expected difference between the paints in that sense.
- Environmental impacts arising from the substrate (i.e. the wood facade) is not accounted for.
- Environmental impacts caused by capital goods (e.g. labours and manufacturing of production units) are not taken into account.

Reporting

An LCA can be published in different formats (e.g. a summary and a full report) and to different degrees of confidentiality (e.g. free to public or confidential). This report is published in two versions. One version is a full report published public, where confidential information regarding secret AkzoNobel know-how is extradited. The other version is a confidential full report intended to be used for AkzoNobel internally.

The project is presented twice. The first occasion is for involved project stakeholders who are the reference group, supervisors and the examiner. The other presentation is for examination at Chalmers University of Technology.

4.4 Results of the case study

The results from the case study are divided into four parts. In the first part the environmental costs are computed for the two paint systems, A and B, with a breakdown for the different contributors in the three scenarios. The second part focuses on the paints, where their life cycle contributions are presented for the baseline scenario 3. In the third part, the environmental costs for 1 kg of paint A and 1 kg of paint B are compared with real costs. In the final part, the underlying environmental information of the environmental costs is presented for the paints in the baseline scenario 3. Some additional results are presented in the confidential report.

All environmental costs are recalculated to the currency Euro in order to simplify comparisons, where the exchange rate 9 SEK = 1 Euro is used. When nothing else is stated, the environmental costs are presented per functional unit.

Comparison of the paint systems

An overview of how the environmental costs are distributed in the paint system is provided in this part. The costs are computed for the different products: paint, impregnator, primer, maintenance wash and paint wash. The environmental costs arising from all transportations and all the products' packaging are presented separately. Figure 21 - Figure 23 below present the total environmental costs for the three monetary weighting methods: EPS, Stepwise and Ecovalue, in the three scenarios for both paint systems.

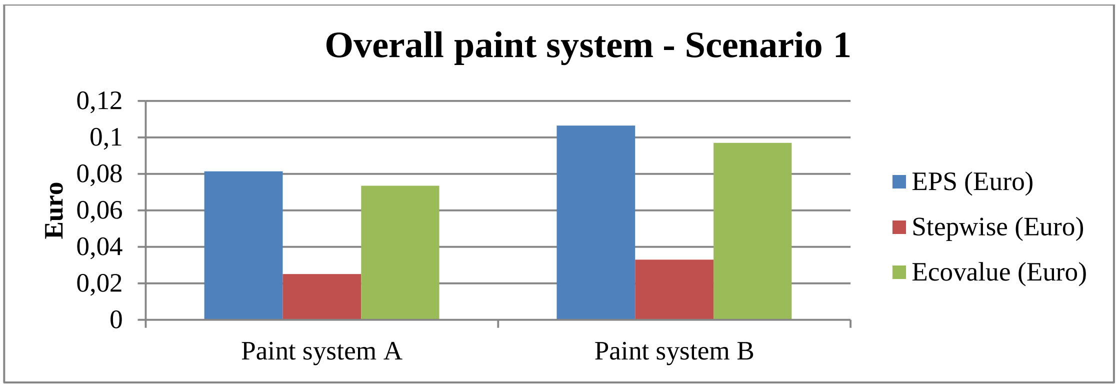

Figure 21 – Comparison of environmental costs between the paint systems for scenario 1.

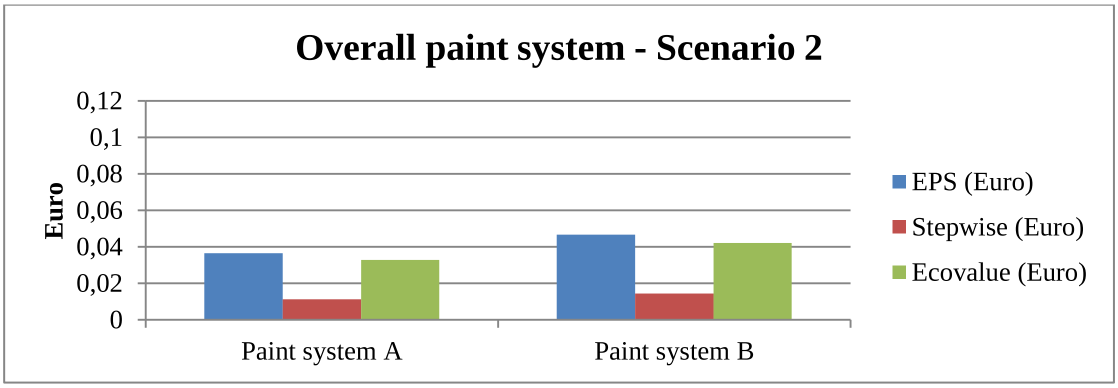

Figure 22 – Comparison of environmental costs between the paint systems for scenario 2.

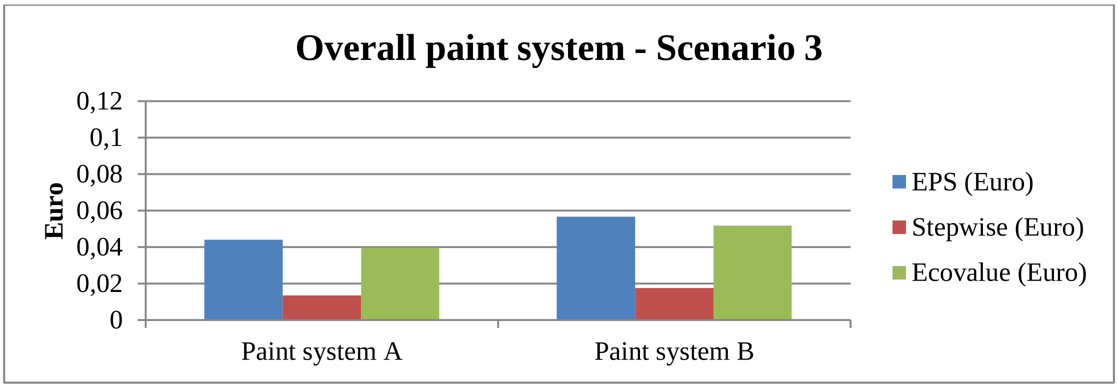

Figure 23 – Comparison of environmental costs between the paint systems for scenario 3.

In the above Figure 21 - Figure 23 there are especially three things that can be seen: (1) cost differences between the weighting methods, (2) cost differences between the scenarios and (3) cost differences between the paint systems. Firstly, the values for EPS and Ecovalue are in the same order of magnitude, with approximately 10 % higher values for EPS. Stepwise on the other hand gives much lower values, about only a third of the other methods´ values. Secondly, more frequent repainting- and washing frequencies give larger values, why scenario 1 obtains the highest environmental costs followed by scenario 3 and scenario 2. As the scenarios aim to cover the possible range of the environmental costs, it can be seen that scenario 1 gives about two times higher environmental costs than scenario 2. This means that the overall environmental impact differs in a magnitude of about two within reasonable repainting- and washing frequencies. The third aspect that can be seen in above figures is that paint system A all through gives around 30 % lower values than paint system B. A possible reason for this can be that the 33 % lower repaint- and washing frequencies play a major part and simultaneously that there is no or only small difference between the impacts from the paints. Other explanations are possible, which are further investigated in the second and third parts of the result presentation. In the two tables below, breakdowns of the above results are displayed.

Table 14 – Contributors to the environmental cost for paint system A.

Weighting method	Scenario	**Total [Euro]**	Paint A [%]	Primer [%]	Impregnator [%]	Maintenance wash [%]	Paint wash [%]	Distributions [%]	Packaging [%]
EPS	1	0.081	91.29	3.51	1.50	0.21	0.48	1.39	1.62
	2	0.036	85.26	7.84	3.34	0.16	0.38	1.41	1.61
	3	0.044	87.12	6.51	2.77	0.17	0.41	1.40	1.62
Stepwise	1	0.025	88.11	3.39	1.45	0.14	0.51	2.37	4.03
	2	0.011	82.28	7.56	3.23	0.11	0.39	2.41	4.02
	3	0.014	84.10	6.26	2.68	0.11	0.43	2.39	4.03
Ecovalue	1	0.074	87.60	3.17	1.39	0.12	0.47	2.38	4.87
	2	0.033	82.01	7.10	3.11	0.10	0.37	2.43	4.88
	3	0.040	83.76	5.88	2.57	0.11	0.40	2.41	4.87

Table 15 – Contributors to the environmental cost for paint system B.

Weighting method	Scenario	**Total [Euro]**	Paint B [%]	Primer [%]	Impregnator [%]	Maintenance wash [%]	Paint wash [%]	Distributions [%]	Packaging [%]
EPS	1	0.107	92.48	2.68	1.14	0.21	0.50	1.37	1.62
	2	0.047	87.56	6.17	2.63	0.18	0.42	1.41	1.63
	3	0.057	89.15	5.05	2.15	0.19	0.44	1.39	1.63
Stepwise	1	0.033	89.27	2.58	1.10	0.14	0.52	2.35	4.04
	2	0.014	84.53	5.93	2.54	0.12	0.43	2.40	4.05
	3	0.017	86.04	4.87	2.08	0.12	0.46	2.38	4.05
Ecovalue	1	0.097	88.23	2.40	1.05	0.13	0.48	2.87	4.84
	2	0.042	84.24	5.54	2.43	0.11	0.40	2.40	4.88
	3	0.052	85.62	4.56	2.00	0.11	0.43	2.38	4.90

A lot of data is presented in the above tables which advantageously might have been presented in other formats, e.g. figures. However it is then difficult to grasp all aspects e.g. the scenarios and the weighting methods in a convenient way. A simplification of the above tables is presented in Table 16 below, where the ranges of the above results are presented for all weighting methods and scenarios. The aim of the above tables is to show each product´s or process´ contribution to the environmental costs and how that differs between the scenarios and weighting methods. First of all it could be identified all through that the paints contribute with the major parts of the environmental costs in the paint systems, about 82-92 %. The large difference between the paint and the other products can be explained for example by larger quantities is used for the paints (see Appendix A). It can also be explained by that the paints are modelled cradle-to-grave, while e.g. maintenance wash and paint wash are modelled cradle-to-gate. For such reasons the maintenance products do only contribute with small environmental costs, less than about 0.7 % combined in all methods and scenarios. Thus, variation in washing frequency gives negligible effects for the overall environmental costs of the paint systems. The primer and the impregnator give on the other hand a higher contribution of about 2-8 % and 1-3 % respectively, depending on method and scenario considered. These two products are expected to be applied only once, on a new facade. This implies larger relative contributions from these products when the painting frequency is lower. Regarding the weighting methods, similar contributions can be seen for the various products.

In the above paragraph, the breakdown results for all products are presented. Remaining are the packaging and the distributions, which contribute with about 2-5 % and 1-3 % respectively. No significant difference can be seen between the scenarios. Regarding the weighting methods, larger contributions are given in Stepwise and Ecovalue for both packaging and distributions.

In the below Table 16, the simplified environmental cost contribution is presented with range values, for all scenarios and weighting methods considered. The major environmental cost is hold by the paint followed by the primer. Negligible contributions are given for the maintenance products.

Table 16 – Simplification of Table 14 and Table 15 presenting average values.

	Paint	Primer	Impregnator	Maintenance wash	Paint wash	Distributions	Packaging
Contribution [%]	82 - 92	2.4 – 7.9	1.0 – 3.3	0.1 – 0.2	0.4 – 0.5	1.4 – 2.9	1.6 – 4.9

Hotspots for the paints

As the focus of the study is on the paints and since the main environmental costs can be derived from the paints, these costs are further assessed based on the baseline scenario 3. In Figure 24 below, environmental costs are calculated for the life cycle phases of the paints by using the three weighting methods.

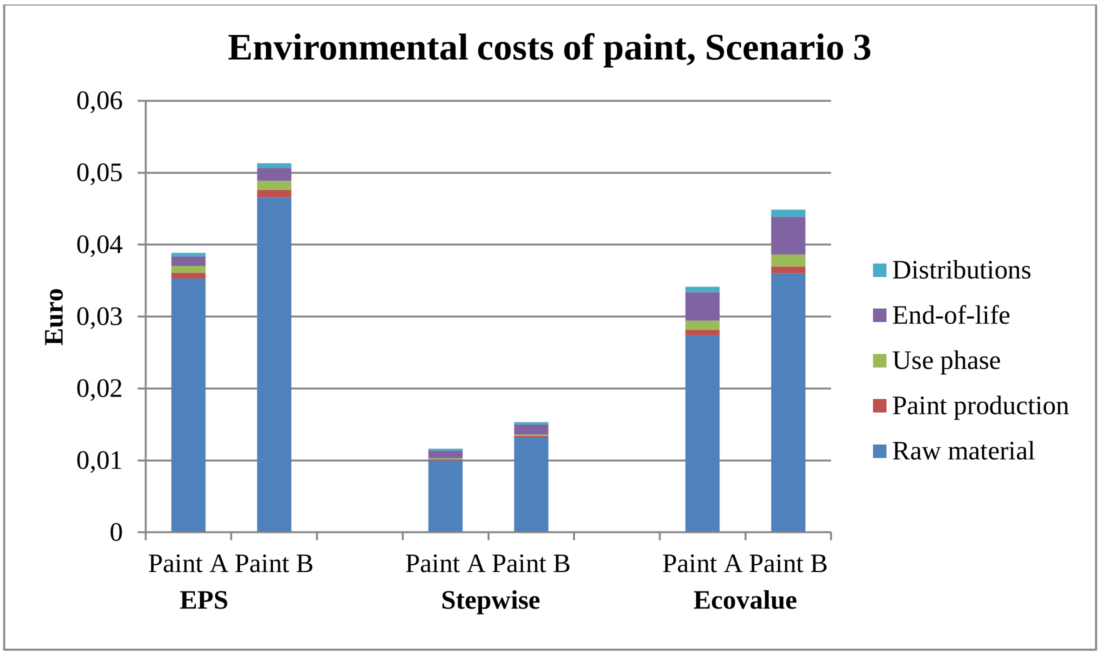

Figure 24 – Identification of the hotspots for the two paints in the baseline scenario 3.

As can be seen in the above figure, the majority of the environmental costs from both paints arise from the extraction and production of the raw materials required in the paints. This applies for all weighting methods, where some differences in the phases can be seen. In Table 17 below, the share of each contributor from the above Figure 24 is presented with the range values for the weighting methods. All phases except from the raw material phase and end-of-life phase give low shares, about 1-4 %.

Table 17 – Share of each contributor for the environmental costs of paint.

	Raw material	Paint production	Use phase	End-of-life	Distributions
Contribution [%]	80 - 91	1.4 - 2.2	1.2 - 3.8	3.5 - 12	1.3 - 2.3

Alike the result from the previous part (*Comparison of the paint systems*), EPS gives the highest values followed by Ecovalue and Stepwise. In addition, the environmental costs are about 31 % higher for paint B than for paint A. Whether this is only due to that a smaller quantity of paint A is used per functional unit or/and due to the difference between the paints, is examined in following result section, *Comparison of environmental costs with real costs*.

Comparison of environmental costs with real costs

Because the scenarios are created subjective and the results seem to depend largely on expected repainting frequency, the environmental costs for 1 kg of the different paints are compared. This would indicate the same durability of the paints during the use phase and it is presented in Figure 25 below.

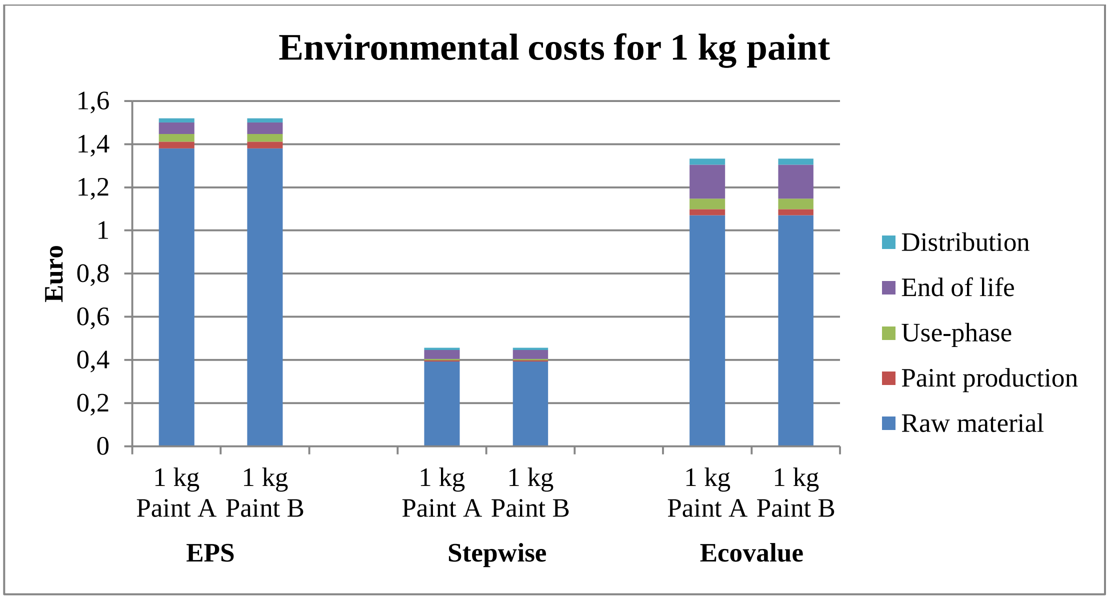

Figure 25 – Comparison of the environmental costs for 1 kg of the two paints.

In paint A, where modified colloidal silica is included some raw materials are lowered compared to paint B. Differences between the paints´ raw material contribution were on beforehand expected to be possible to measure due to differences between the products´ formulations. When studying the above figure, no such difference can be seen. There are however a small difference between the paints with about 0.1 - 0.3 % higher environmental costs for the paint A, where modified colloidal silica is included. This difference origin from the raw material phase but can be seen as negligible, stating that the environmental costs are practically the same for 1 kg of both paints. The reason for why similar values are given for the paints is that the raw materials excluded in paint A give almost the same environmental cost as for the added colloidal silica.

The result in the above figure could in one since be viewed as incomplete. This is since there are no methods available to assess the environmental impacts from the end-of-life phase of colloidal silica in a comprehensive and consistent way (Hischier, 2014). This implies that no method is available to quantify the environmental costs. This is further discussed in Chapter 5.

So far, the environmental costs have only been presented without putting it into a context. In order to better grasp the magnitude of the environmental costs, a reference point is expected to be necessary. There are several references possible, where a comparison with real costs would be interesting. The reference point chosen is the product price to the end-consumer of the paints, where paint A has about 6 % higher price than paint B. The comparison between the environmental costs and the end-consumer´s product price is displayed in Figure 26 below. Error bars are added for the environmental costs where the higher and the lower parts correspond to the weighting methods given the highest and lowest scores. The mid value is given for the weighting method in between these numbers.

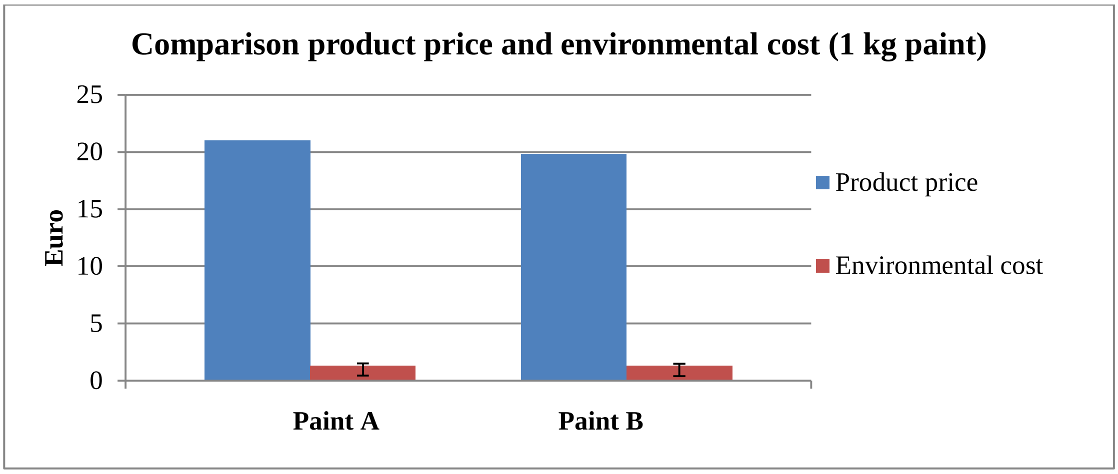

Figure 26 – Comparison between the paints´ environmental costs and end-consumer´s product price.

As indicated in Figure 26, the environmental costs constitute a minor part of the total product price. For paint A and B it varies between about 2 % (for Stepwise) and about 8 % (for EPS) of the paints´ product prices. This could illustrate the environmental costs that could be added in the future, for example through various policy measures. This is further discussed in Chapter 5.

By modifying this result slightly, it can be compared with the result from the study by KPMG International (2012), which compared environmental costs per earnings. For the chemical sector, the calculated environmental costs per earnings were computed to be 43 %. Since no exact return on sale rate is found for the specific paints, it is assumed that the return on sale rate is similar to an AkzoNobel Decorative paint which was 10 % in year 2013 (AkzoNobel, 2014). By subtracting 90 % from the above product price the quantity of the earning is given. The environmental cost per earning is thereby possible to calculate which varies between 22 % (for Stepwise) and 73 % (for EPS) for paint A. Similar numbers for paint B are given, 24 % (for Stepwise) to 77 % (for EPS) environmental cost per earning. These numbers are in the same range as the 43 % from KPMG International (2012) in the chemical industry.

Underlying environmental impacts from environmental costs

When conducting an LCA where weighting is included, it is highly important to show the underlying results. In this case it means the environmental impacts contributing to environmental costs. As the baseline scenario 3 should represent an average situation, this scenario is utilized in order to understand how the environmental costs are computed. To be able to compare the results with the environmental issues presented in the literature review (see Subsections 3.4.2 and 3.4.3), only paint and its transportations are included. This is presented for the three monetary weighting methods used in the study, EPS, Ecovalue and Stepwise. EPS differs from Ecovalue and Stepwise for instance from the aspect of where the weights are attached in the cause-effect chain. This gives implications on how the environmental impacts from these methods can be derived. In EPS, the results are presented from which environmental interventions (raw material, emission) that are occurring in the system. It might have been better to present the results for each safeguard subject affected. However, due to the modelling of the EPS method in the GaBi6 software, these results are not possible to present. For Ecovalue and Stepwise, the potential environmental impacts are presented, expressed in different impact categories e.g. global warming and acidification. The underlying environmental impacts contributing to the total environmental costs are presented in Figure 27 - Figure 29 below in percentages of total environmental cost, where further breakdowns of each weighting method´s environmental information is presented in Table 18 - Table 20.

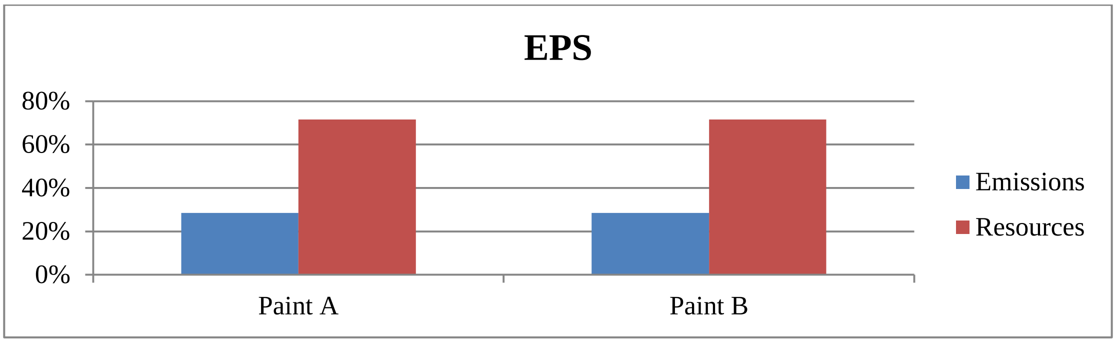

Figure 27 – Underlying environmental contribution to environmental costs for the EPS method.

In the above Figure 27, the underlying environmental costs for the EPS method can be seen. It shows that resources constitute the majority of the environmental costs. Further breakdown of the environmental interventions are found in Table 18 below.

Table 18 – Breakdown of the environmental costs for the EPS method for both paints.

Main categories	**Sub categories**	**2^{nd} Sub category**	**Paint A [%]**	**Paint B [%]**
Resources	Energy resources (non renewable)	Divided on several resources	31.60	31.32
	Material resources	Non renewable elements	29.60	29.82
		Non renewable resources	10.43	10.42
	Land use	Occupational, arable	~ 0	~ 0
Emissions	To air	Inorganic	21.22	21.24
		Organic (VOC group)	6.36	6.39
		Particles	0.67	0.62
		Heavy metals	0.12	0.19
	To fresh water	Divided on several emissions	~ 0	~ 0
	To sea water	Divided on several emissions	~ 0	~ 0
Total			~ 100	~ 100

The above table shows the valuated results, where the highest contributions are given for various material- and energy resources, but also for inorganic emissions to air. Similar percentages can be seen for both paints indicating that they give raise to similar types of environmental interventions.

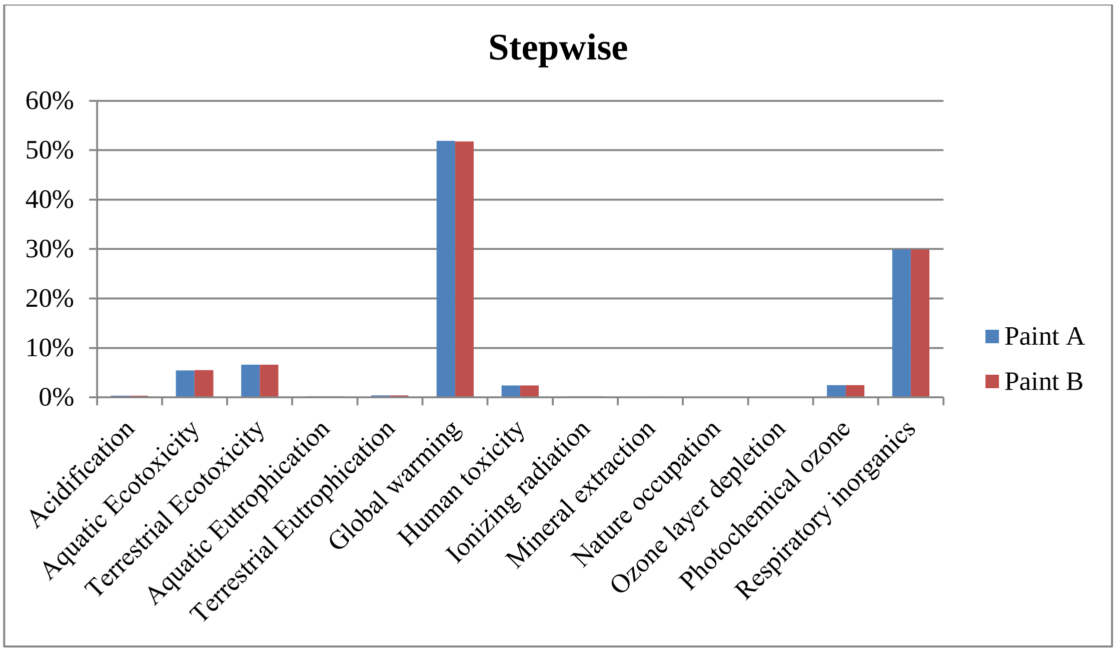

Figure 28 – Underlying environmental contribution to environmental costs for the Stepwise method.

In the above Figure 28, the underlying environmental costs are presented for the Stepwise method. The dominant environmental contributors are global warming and respiratory inorganics. All the contributors are further presented in Table 19 below.

Table 19 – Breakdown of the environmental costs for the Stepwise method for both paints.

Impact category	**Contribution (Paint A) [%]**	**Contribution (Paint B) [%]**
Acidification	0.33	0.34
Aquatic Ecotoxicity	5.44	5.49
Terrestrial Ecotoxicity	6.59	6.64
Aquatic Eutrophication	0.23	0.23
Terrestrial Eutrophication	0.38	0.38
Global warming	51.88	51.75
Human toxicity	2.42	2.42
Ionizing radiation	0.22	0.23
Mineral extraction	0.13	0.14
Nature occupation	0.01	0.01
Ozone layer depletion	0.01	0.01
Photochemical ozone	2.46	2.47
Respiratory inorganics	29.90	29.89

Except from global warming and respiratory inorganics, which are accountable for around 80 % of the environmental costs, non-negligible contributions are given for aquatic- and terrestrial ecotoxicity as well as human toxicity and photochemical ozone (see Figure 28 and Table 19 above).

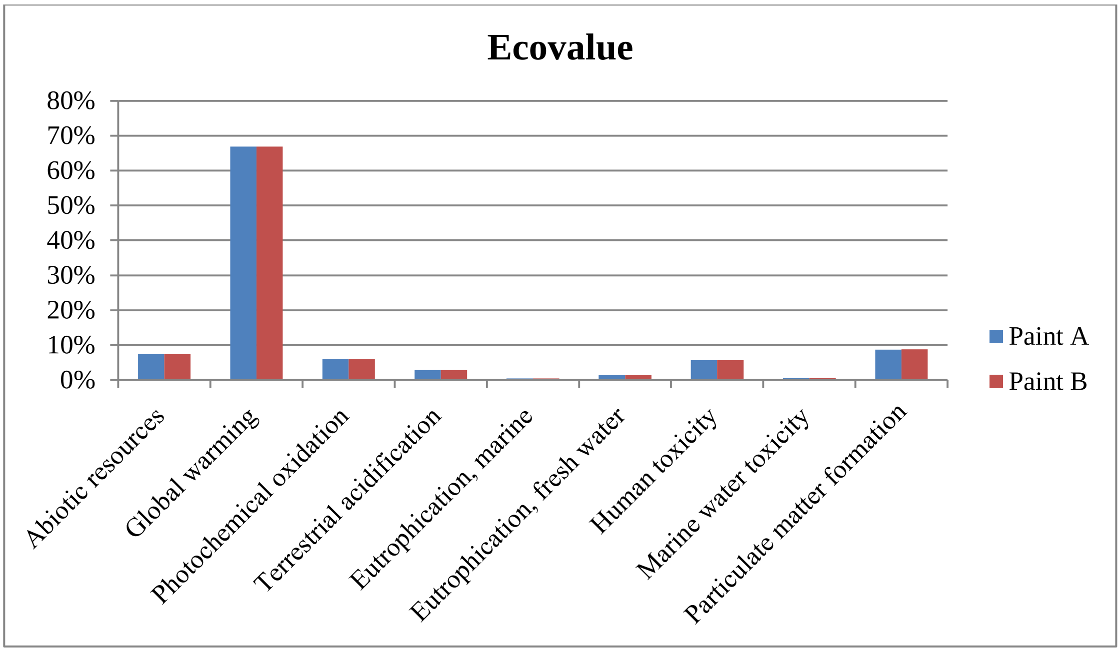

Figure 29 – Underlying environmental contribution to environmental costs for the Ecovalue method.

The last weighting method is Ecovalue and similar to previous methods, Figure 29 above presents the underlying environmental costs. The numerical values are presented in Table 20 below.

Table 20 – Breakdown of the environmental costs for the Ecovalue method for both paints.

Impact category	**Contribution (Paint A) [%]**	**Contribution (Paint B) [%]**
Abiotic resources	7.45	7.45
Global warming	66.93	66.84
Photochemical oxidation	5.96	5.97
Terrestrial acidification	2.81	2.82
Eutrophication, marine	0.44	0.44
Eutrophication, fresh water	1.41	1.41
Human toxicity	5.68	5.71
Marine water toxicity	0.57	0.57
Particulate matter formation	8.75	8.79

The two largest contributors for Ecovalue are the same as for the Stepwise method, global warming and particulate matter formation. The latter stated is similar to the impact category in Stepwise, respiratory inorganics. Other categories given non-negligible contributions are human toxicity, abiotic resources, photochemical oxidation and terrestrial acidification.

5 Discussion

In the following chapter, the results are discussed in the light of the report´s aim and literature review. The chapter is divided into five parts, which begins with a discussion of the methodological choices. Thereafter each part from the result section is discussed with a focus on the last part, *Underlying environmental impacts from environmental costs*.

Methodological choices

In Life Cycle Assessments the methodological choices are often decisive for which results that are reached (Rebitzer et al., 2004). This study is not an exception. The result is based on several choices regarding the functional unit chosen, impact assessment considered, data used, allocations, system boundaries, scenarios and many more. The aspects that are expected to be most influential for the results are addressed below.

One aspect is the choice of functional unit, where coverage and durability are two important parameters to include for paints as stated in Chemiewinkel, Enterprise Ireland and WIMM (2000). These are however difficult to determine properly, where both the amount of paint used and the life time of the paint could differ between different situations and depend on end-consumer behaviors. This implies challenges to evaluate the environmental performance of paints and comparison between them. Therefore it might be a good idea to vary these two parameters and see how that affects the result. In this study only the durability parameter (years between repainting and washing) is varied since that was expected to vary more than the coverage parameter (product quantities per m^2 facade area), according to the project´s reference group. In connection to this, another decisive aspect is the expected longer life time for the paint where modified colloidal silica is included. In order to get a more accurate comparison between the two paints an evaluation of the expected longer life time would be necessary to make in order to identify the actual benefits of colloidal silica in paint. Extending the life time means lower repainting frequency implying lower need for painting giving lower environmental impacts and costs.

The choice of which type of impact assessment that is used gives implications on for instance how the results are presented and which environmental impacts to cover (Baumann and Tillman, 2004). The starting point for the study was to make an LCA monetization study, which implies use of monetary weighting methods presenting the environmental impacts in one single number. Weighting is however not fully accepted in the LCA community and certain care should be taken before conclusions from such studies can be drawn as results always are uncertain when weighting factors have been used (Baumann and Tillman, 2004). Therefore, weighting should only acts as a complement to the assessment of the environmental impacts. It is also important to use several weighting methods where the underlying environmental impacts are presented (Bengtsson, 2000b), which is performed in this study. Regarding the coverage of the environmental impacts, the weighting methods chosen claim to capture a wide variety of environmental impacts. The coverage of the weighting methods is further discussed in *Underlying environmental impacts from environmental costs*, in this chapter.

The most time consuming phase in an LCA is often the data collection (Finnveden et al., 2009). In relation to this it is important to discuss the aspects of the data, especially its relevance, reliability and accessibility (Baumann and Tillman, 2004).

Most of the data used is internal AkzoNobel data with different precision for instance a specific raw material considered where the data is up-to-date or non up-to-date emission factors of water based paint for average European conditions. In this study, data with high reliability and high relevance was sought after and since most of the data used is confidential AkzoNobel data, the accessibility parameter is unfortunately affected.

Comparison of the paint systems

In the result section of this aspect, three things are identified, (1) differences between the weighting methods, (2) differences between the scenarios and (3) differences between the paint systems. These three are to be discussed by starting with the first one stated. It can be seen that Stepwise in general gives lower environmental costs than both EPS and Ecovalue. There are several reasons for such a difference, for instance environmental impacts covered in the methods and the quantity of the valuation factors. The main explanation for the difference in this case is that Stepwise has lower valuation factors than the other methods. This can for example be depicted when looking at the major contributor for both Stepwise and Ecovalue, global warming (see Figure 28 and Figure 29). In Stepwise each carbon dioxide equivalent is valuated to 0.083 Euro (see Table 6), while the weighting factor for Ecovalue is higher, about 0.32 Euro[12] (see Table 9). As EPS differs from the two other methods, the valuation per intervention or endpoint impact can only be stated, thus not per midpoint impact category. The valuation of carbon dioxide in EPS is 0.108 Euro per kg, found in Steen (1999b). Differences between the weighting methods are further discussed in *Underlying environmental impacts from environmental costs*.

Secondly, it is identified in the scenarios that higher repainting- and washing frequencies give larger environmental costs. This is perhaps not a surprising result since larger quantities of paint- and washing products are required during the building´s life time implying larger quantities per functional unit. It is however only changes in repainting frequency that exhibit the major variation in the environmental costs between the scenarios. The proportion of the contributors to the environmental costs for the paint systems are dominated by the paints. As suggested in the result, this can for instance be due to quantity differences between the products (see Appendix A) but also due to the coverage in how much of the life cycle that is modelled. A clearer picture of how the environmental costs are distributed in the paint systems might have been given if all products were modelled cradle-to-grave. This would for example imply a better understanding of how the environmental costs differ between different scenarios. On the other hand, as more products are studied, there is less time to immerse in each product.

Regarding the third aspect about the difference between the paint systems, the main reason for the difference is the expected increase in the life time of the paint including modified colloidal silica. There are other reasons such as impact differences per mass of paint, density variations between the paints, smaller need for washing products, transportations and packaging in paint system A. The main reason is however the increased life time which is further discussed. By including modified colloidal silica, it is expected to increase the life time of the specific paint in the case study by 33 %. This number could however vary which implies that the difference between the paint systems would vary similarly. As this number is highly decisive for the results, it is

[12] Recalculated from SEK to Euro with exchange rate 9 SEK = 1 Euro

important to better identify an accurate range of that life time benefit. The environmental cost benefits given for paint A also relate to lower real costs for the end-consumers as they do not need to repaint their houses as frequent, implying lower material and working costs. An evaluation of the real costs is presented in the confidential result part of the report.

Hotspots for the paints

When studying the most contributing phases of the life cycle, extraction and production of raw materials is shown to be responsible for the major part of the environmental costs. In this study, that is not surprising due to several reasons. The data for this life cycle phase is more accurate and comprehensive than the other phases. It is also important to mention that the dissipative use of paint (except for the volatile components) is not modelled in the use phase why a different result in that phase might be possible. For example Kougoulis et al. (2012) estimates that 40 % of the environmental impacts from paint are expected to arise in the end-of-life phase. Other allocation methods in the end-of-life could also alter the result.

On the other hand, the dominance of the raw material phase in relation to the smaller contributions from the other phases can be substantiated by the literature review, where some examples are followed. According to CEPE (2012) the paint production is expected to give low contribution as it is only a matter of mixing the carefully evaluated components. The use phase gives only small contribution, mainly since the studied paints are water based implying low VOC content evaporating after attachment (Overbeek et al., 2003). The environmental impacts and costs in the end-of-life phase are expected to vary depending on how the waste management is undertaken. For example when conducting the system expansion for incineration, the fuel used in the nearby energy producing system determine how much of the impacts in the product system that can be subtracted (Mattson, 2013). Regarding the distributions, these are considered to be a major contributor to environmental impacts in society, but often small contributions are given in LCA studies (Baumann and Tillman, 2004).

Comparison of environmental costs with real costs

To state environmental costs by its own and not put them into a context make only little or no sense. A reference point is thereby expected to be necessary enabling comparisons. Such reference point chosen in the result presentation is the product price, which would indicate a possible future cost burden for the paint producers implemented from for instance policy instruments (Tekie and Lindblad, 2013). This comparison could also acts as a guidance to choose between product alternatives. It is however important to recognize that such imaginary environmental costs are societal costs, not owned by a specific object. In contrast, product related costs are real costs owned by someone. It is thereby highly unsure if these environmental costs would be internalized in the product price in the future. It is however widely recognized by companies that an ongoing environmental degradation impairs their businesses (KPMG International, 2012). By accounting for environmental costs will not decrease the environmental impacts by themselves, but it could rather acts as a catalyst for firms to actively reduce their environmental impacts and thereby reduce the environmental costs.

In the case study, environmental costs are compared between the paints but also to the product prices and estimated earnings. When comparing the environmental costs between 1 kg of paint A and B only small cost differences are found, about 0.1 - 0.3 % lower values for the paint without modified colloidal silica. When comparing these environmental costs to the product price it showed to constitute about 2 - 8 % of the product price. When studying the environmental costs in relation to the earnings, this proportion is much more uncertain since earnings can vary a lot. By using a 10 % return on sales rate, environmental costs per earnings are varying between 22 % and 77 % depending on weighting method considered. A similar number can be found in the KPMG International (2012) study, where the average environmental cost for the chemical industry is 43 % of the earnings. How or if these costs will be imposed on any actor in the value chain of paints is uncertain. An example provided by Chemiewinkel, Enterprise Ireland and WIMM (2000) shows an implementation of a tax system where the paint producer needs to pay a tax determined by the environmental impacts from the paint. As the implementation of the environmental costs are uncertain, there are also uncertainties in how to distribute the costs fairly between the actors.

To come up with precise numbers of environmental costs that are widely accepted and can be put in relation to real costs is not an easy task. For example, how should the environmental costs be calculated? As real costs are usually derived from market prices, it might be appropriate to derive environmental costs from market prices as well. A main difficulty, according to Ahlroth et al. (2011), is however that environmental goods and services often lack market prices, why it is necessary to find the right price in another way that the society can agree upon. There are more aspects of how to find the right price, e.g. geographical differences implying different prices in different regions, variations in how much people are willing to protect and preserve the environment and the possibility to make people agree upon fair environmental costs in terms of e.g. taxes. It might however be interesting to study how much the environmental impacts would cost the society, in order to better understand environmental costs.

Underlying environmental impacts from environmental costs

Valuation of environmental impacts is a controversial issue, outlined in Section 3.2. There are several different monetary weighting methods to use for converting environmental information to economic information. They are taking their starting point in different values with respect to for instance whose values they are, by whom they are stated and how they are derived (Bengtsson, 2000a; Bengtsson, 1998). Different values could be required in different decision situations and there will probably never be any method covering all situations (Ahlroth et al., 2011). As the weighted results often differ depending on the weighting method chosen, it is important to use several methods in the study (Bengtsson, 2000b). To get close to the true ecological cost, it is also important to find comprehensive weighting methods covering many environmental impacts. Thereby, it is expected that the probability is reduced for environmental costs not accounted for. Thus, it is assumed that a discussion is necessary of how applicable the weighting methods used in the case study are for the specific paints studied.

The monetary weighting methods in the study: EPS, Stepwise and Ecovalue all varies with respect to which environmental impacts that are considered and the values used. This is why the result differs between the methods. For example EPS and Stepwise cover all three safeguard subjects agreed upon by the LCA community. Valuation of waste could however not be found for any method except from EPS where littering is included (this is however not accounted for in the report). When comparing Stepwise and Ecovalue it can be identified that Ecovalue do not cover as many environmental impacts as Stepwise. On the other hand, the weighting factors in Stepwise are in general lower than they are in Ecovalue, why monetary values are lower for Stepwise than for Ecovalue (see for instance Figure 24 and Figure 25). As stated in the literature review, it is difficult to evaluate valuation methods since the values involved are difficult to identify and evaluate (Ahlroth and Finnveden, 2011).

Regarding the matter of if high environmental costs from impact categories can be related to the environmental issues, stated in Subsection 3.4.2 and Subsection 3.4.3, is to be evaluated. Environmental issues presented in these sections are for instance solvents, nanomaterial, hazardous chemicals and titanium dioxide production. According to the last part of the result presentation, *Underlying environmental impacts from environmental costs*, major contributors in the different methods are: global warming, respiratory effects/particulate matter formation, human- and ecotoxicity, abiotic/material and energy resources. Different names are used in the different weighting methods where respiratory effects and particulate matter formation are similar, measuring impacts from particles smaller than 10 micrometers. The toxicity categories include toxic substances exposed to humans and the environment. The major contributors from the resource category are energy and material.

When comparing the issues stated in the literature review by these in the case study, a few things can be said. First of all, the reason for the dominance of the impact category global warming in the case study cannot fully be explain by the literature review. Two reasons for this are however suggested. One is that the raw material phase is rather energy intensive (e.g. titanium dioxide production) why major impacts from carbon dioxide is present in for example the electricity production. Another possible reason is that the global warming category is considered as more severe and important than other categories in the weighting methods and is therefore valuated higher. Regarding the toxicity categories, their contributions are not surprising, since paint often include hazardous substances. Other impact categories such as acidification and eutrophication not outlined as environmental issues in the literature review give small contribution to the environmental costs in the case study. One category that gives smaller contribution than expected is photochemical oxidation/ozone or organic emissions to air (VOC), which has been one of the core environmental issues in the paint industry. One explanation for its small contribution could be the low VOC content in the paints, as they are water based. When comparing the contributions from the different weighting methods, it can be seen that environmental costs from EPS mainly are derived to resources. This could be explained by a wide coverage of resources in the EPS method, while resources give lower cost contributions for both Stepwise and Ecovalue.

Since the environmental impacts are valuated, it is difficult to draw definite conclusions about which environmental issues that are most prominent for the paints in the case study. Even though the environmental impacts from the paints could be presented (in their units), it might be difficult to compare different impact categories since they have different units. There are some environmental issues not fully captured in the assessment due to expected difficulties in the modelling, e.g. dissipative use of potent chemicals and nanomaterial. When considering nanomaterials, it is important to stress that none of these weighting methods valuate the cause-effect chain of colloidal silica or other such particles of nanosize. The environmental science of such materials is in its yearly stages and future evaluations are necessary according to Hischier (2014), to determine the environmental effects and the subsequent environmental costs. The NANOHOUSE study showed however that ENM (engineered nanomaterial) of colloidal silica pose no major environmental and health effects (Wick, Krug and Nowackm, 2011). Trust can however not be put only based on one single study, why more studies are required to determine both the probability of ENM releases and the environmental impacts of ENMs and valuation of these impacts.

To find an appropriate applicable monetary weighting method is a challenging task since many aspects need to be considered, e.g. application area, impacts to cover, values to apply to. It is difficult to tell which method that is the best in a certain situation but the three methods used in the case study showed however to be applicable, since they cover the main environmental issues for paints and their different values created a range of the result. As stated in the literature review and especially in Section 3.2, the concept of translating environmental impacts to economic values are fairly new (KPMG International, 2012), why more work is required in the area to create more comprehensive and consistent monetary weighting methods.

6 Conclusions

In the following list the most important conclusions that can be drawn from the report are stated.

- Environmental benefits are possible when including modified colloidal silica in paint as a prolonged life time of the paint is expected. The expected increase in the paint´s life time showed to be a decisive factor for the environmental benefits in the case study, why this needs further evaluation.

- All three monetary weighting methods (EPS, Stepwise and Ecovalue) used in the case study are applicable when evaluating the environmental impacts from the paints in the case study.

- The production and extraction of raw materials are responsible for the major part of the environmental costs for the life cycle phases of the studied paints.

- About 2-8 % of the end-consumer´s product price of the paints in the case study is environmental costs. This proportion can indicate costs that might be paid in the future by the actors in the value chain of paint or by the society as a whole.

- To find an appropriate functional unit for paint is challenging as important parameters are the paint´s coverage and durability, which are difficult to determine.

- The scientific knowledge about the environmental impacts from nanoparticles the context of Life Cycle Assessment is low. Characterisation- and valuation factors translating inventory data to environmental impacts and further on to environmental costs are lacking.

7 Further work

To put this report in a larger perspective, it is important to suggest further work from the report both for AkzoNobel internally and for other studies. Firstly, the use of this report for AkzoNobel is not exactly defined. Since the concept of monetize environmental impacts in companies is fairly new, AkzoNobel is working explorative with this issue of how or if external costs can be utilized both internally in for instance product development and externally as a communication tool. There have been studies within AkzoNobel evaluating external costs, including both environmental- and societal costs. It is however expected that some more work has to be done before a framework of how to deal with external costs in AkzoNobel is created.

Regarding further work in other studies the below list presents important study areas.

- Identify how companies can work with environmental costs but also societal costs in their businesses, for example in order to gain competitive advantages.
- Investigate how environmental costs can be implemented in different industry sectors (e.g. the chemical industry) with regards to for instance necessity, fairness and implementation procedure.
- Increase the knowledge about how to handle colloidal silica and other such materials in LCA.
- Tests where the durability of paint with and without colloidal silica is investigated.
- Further tests of different monetary weighting methods in for instance case studies.

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Appendix A – Product quantities in the paint systems

In Table 21 – Table 23 below the quantity of the included products (paint, primer, impregnator, paint wash and maintenance wash) and packaging (polypropylene) required (in grams) per functional unit are displayed for the different scenarios.

Table 21 – Product quantities per functional unit.

Scenario	Paint A [g]	Paint B [g]	Impregnator [g]		Primer [g]		Paint wash [g]		Maintenance wash [g]	
Belongs to paint system	A	B	A	B	A	B	A	B	A	B
Scenario 1	49.6	65.6	2.0	2.0	2.15	2.15	1.92	2.61	1.92	2.61
Scenario 2	20.7	27.1	2.0	2.0	2.15	2.15	0.67	0.94	0.67	0.94
Scenario 3	25.5	33.5	2.0	2.0	2.15	2.15	0.88	1.22	0.88	1.22

Table 22 – Packaging quantities per functional unit.

Scenario	Paint A [g]	Paint B [g]	Impregnator [g]		Primer [g]		Paint wash [g]		Maintenance wash [g]	
Belongs to paint system	A	B	A	B	A	B	A	B	A	B
Scenario 1	1.43	1.90	0.07	0.07	0.06	0.06	0.17	0.24	0.17	0.24
Scenario 2	0.60	0.79	0.07	0.07	0.06	0.06	0.06	0.09	0.06	0.09
Scenario 3	0.74	0.97	0.07	0.07	0.06	0.06	0.08	0.11	0.08	0.11

Table 23 – Packaging quantities per functional unit aggregated.

	Paint system A[13] [g]	Paint system B[14] [g]
Scenario 1	1.91	2.51
Scenario 2	0.85	1.09
Scenario 3	1.03	1.33

[13] Packaging required for: paint A, impregnator, primer, paint wash, maintenance wash
[14] Packaging required for: paint B, impregnator, primer, paint wash, maintenance wash