The influence of impact categories and monetization factors on true costing: the case of disposable and cloth diapers

“This report was prepared to partially fulfill the requirements for the master’s degree in Environmental Sciences. It is not an official publication of Wageningen UR”

The influence of impact categories and monetization factors on true costing: the case of disposable and cloth diapers

19th of May 2023

MSc Environmental Sciences

Specialization: Economics and Natural Resources

Course code: ENR80436

Student: Annabelle Poventud (student number: 1145525)
Supervisor: dr. ir. Rolf Groeneveld
Examiner: dr. Hans-Peter Weikard

Table of contents

Abbreviations:

Life Cycle Assessment	LCA
True Price foundation	TP
Environmental Agency	EA
Marginal Abatement Cost	MAC
Social Cost of Carbon	SCC

1. Introduction

1.1. Problem Statement

Sustainable products suffer unfair competition by unsustainable products because of the externalities that are not accounted for. An externality arises when a party's activities in terms of production and/or consumption affect the utility or profit of another party without compensation (Perman et al., 2011). Externalities are unintended and an example of market failure because the optimal resource allocation cannot be reached if the externalities are not accounted for (Pizzol et al., 2015). Therefore, externalities, also referred to as hidden costs, are consequences for others which are not accounted for (Dasgupta & Ehrlich, 2013).

There is a growing interest from stakeholders, such as researchers, entrepreneurs, and NGOs, to estimate true costs as it can be used by stakeholders to compensate for externalities. After identifying and quantifying externalities, they can be monetized to get true costs, also referred to as external costs, public costs, or the true price gap. True cost can be used as a tool to calculate financial compensation for externalities. Through the compensation of true costs, externalities can be internalized. In some cases, true costs are paid by the party inducing the damage as a compensation to the affected party (Matthews & Lave, 2000). In this case, true costs can determine the behavior of the polluter (Perman et al., 2011). Also, true cost is used to support financial incentives by taxing unsustainable practices or subsidizing sustainable ones (Sustainable Food Trust, 2019).

More recently, true costs have been added to market prices resulting in end consumers paying the "true price" of a product (True Price, 2021). The true price of a product is the aggregation of the private costs reflected in the market price, and the public costs which are the true costs. Purchasing and consumption are degrading the environment (Chekima et al., 2016). To tackle this, consumers need to make more sustainable choices (Chekima et al., 2016). Prices are a major factor the purchasing decision, thus adding true costs to the market price could affect the purchasing behavior of the consumer. Adding true costs to a product affects the demand of a product or service. Market prices are at the equilibrium, thus at the intersection of the demand and supply curve of a product or service. When the price is higher than the equilibrium price (market price), the demand for this product decreases. Therefore, when true pricing is adopted, the demand for unsustainable products should decrease.

An example of price disparity between sustainable and unsustainable goods is the choice between disposable or cloth diapers. The diaper trade-off is an important dilemma because baby diapers have a market demand of over 20 billion euros in the European Union. This translates to 690 kilotons of diapers (Mendoza et al., 2019). Given that there were 178 441 births in the Netherlands in 2021 (Statista, 2021) and that children take on average 2.5 years to be toilet trained, the number of children wearing diapers in the Netherlands is around 446 000 per year. Disposable diapers are mostly made out of plastics and cloth diapers out of cotton. Pampers is the most popular disposable diaper brand in the Netherlands which cost 0.25 euros per diaper (Pampers, 2023), whereas an average cloth diaper, also referred to as washable or reusable diaper, is 18.75 euros per diaper. Therefore, the price of a disposable diaper is about 1.3% of the price of a cloth diaper. It seems obvious that consumers choose the disposable diaper based on financial comparison per diaper. It is however important to mention that over a diaper period of 2.5 years, the cloth diaper becomes financially more attractive because fewer diapers are needed as they are washed and reused. Around 48 cloth diapers are needed instead of 5400 disposable diapers per toilet trained child (Milieu Centraal, n.d.). Indeed, the cost of disposable diapers over the period of one toilet trained child is 1350 euros. The costs related to the usage of cloth diapers for one toilet trained child is 500 euros cheaper on average than the cost of disposable diapers (Milieu Centraal, n.d.). This results in the cloth diaper costing on average 850 euros per toilet trained child.

Despite the cloth diaper being less costly over the period of one toilet trained child than its disposable alternative, consumers are more tempted to buy disposable diapers which may have some financial explanation. Since the financial benefits become apparent only after several years, temporal discounting may cause consumers to perceive a lower present value because future benefits are postponed (Green et al., 1997). Another aspect affecting the consumer's behavior towards the purchase of disposable diapers is consumer myopia where the consumer's decision making is focused on short term returns instead of long-term returns. In this case, the consumer does not perceive add-on costs to the market price (Wenzel, 2014). An example of an add-on cost are true costs.

In the Future Diaper Report by the True Price Foundation (TP) the true costs of the disposable and cloth diapers over one toilet trained child were compared and it was argued that the disposable diapers have a higher true cost than the cloth diapers (Future Diaper Project, 2022). The true cost of the disposable diaper is 350 euros per toilet trained child whereas the true cost of a cloth diaper is 250 euros per toilet trained child when considering only environmental impact categories (Appendix 1).

The results by the TP are however bond to two main assumptions: (1) the impact categories are based on one Life Cycle Assessment (LCA); and (2) the monetization factors are based on their own databases presenting only one monetization factor per impact category. Indeed, in the Future Diaper Project, the TP used their own LCA presenting 6 environmental impact categories: contribution to climate change, land occupation, fossil fuel depletion, scarce blue water use and water pollution. However, many environmental impacts can be included in a LCA. This suggests that the TP had to make a selection of impact categories. Furthermore, the TP monetized the LCA using their database with one standard monetization factor per impact category which are said to be universal and based on global averages (Future Diaper Project, 2022). But, many monetization factors exist per impact category as they are calculated in different manners, adapted to geographical contexts and dependent on the discount rate used (Amadei et al., 2021; Arendt et al., 2020). Using the TP monetization factors in true cost calculations across studies, products and sectors could be seen as a benefit transfer because the data would be used in other settings than it is collected for (Rosenberger & Loomis, 2003).

Therefore, the selection of impact categories and monetization factors of the TP might affect the true cost results.

It is however unknown how influential the selection of impact categories and monetization factors are on the true cost results between the disposable and cloth diapers. It could even be the case that the selection of impact categories and monetization factors are decisive in the trade-off between the disposable and cloth diaper.

1.2. Objective and research questions

This research aims to analyze whether the ranking in true costing of the disposable and cloth diaper is dependent on the impact categories and monetization factors selected.

The following research questions are addressed:

1. Does the selection of environmental impact categories affect the ranking of the true costs of the disposable and cloth diapers?
2. Does the selection of monetization factors affect the ranking of the true costs of the disposable and cloth diapers?

1.3. Methodology

The true cost of the disposable and cloth diapers were assessed based on the functional unit "one toilet trained child". For the first research question, variability in impact categories is achieved by calculating the true costs of the disposable and cloth diapers based on a LCA by the Environmental Agency. Indeed, the Environmental Agency's LCA selected different impact categories than the TP's LCA. For both LCA studies the true costs of the disposable and cloth diapers are ranked, then the two rankings are compared. For the second research question, one of the two previously studied LCAs was selected. Then, monetization factors were found to establish a range of monetization factors per impact category. The differences in monetization factors were established based on differences of calculation methods, geographical context and discount rates and were found in gray literature and academic literature via Scopus, Google Scholar, and Connected Papers. Once the ranges were established a sensitivity analysis was conducted to analyze whether variations in monetization factors would affect the ranking in true cost of the disposable and cloth diapers per toilet trained child. Further information on the methods used in this paper are presented in Section 3.

2. Background

2.1. True cost purpose, method, and limitations

Monetization of environmental impacts have been mostly used for Cost Benefit Analysis but is now emerging in the valuation of LCAs (Arendt et al., 2020). A Cost Benefit Analysis is a tool to measure the costs and benefits of a project which is related to the aggregated utility of every stakeholder involved over time in this project (Perman et al., 2011). The environmental impacts, mostly non-market goods, are in this case monetized in order to add these impacts to other costs and benefits of market-goods related to the project (Ahlroth, 2009; Arendt et al., 2020) In contrast, in a LCA, monetary valuation is used as a weighing method in order to compare environmental impacts with each other (Pizzol et al., 2015). A LCA is a tool that estimates the total environmental impact of a product through its life cycle (Perman et al., 2011). The results are then discussed within the goal and scope of the LCA, often this tool is used to compare the environmental impact of different products (Ahlroth & Finnveld, 2011; Amadei et al., 2021).

The valuation of LCAs can be challenging because of: (1) the use of potential impacts; (2) the impacts are dispersed over space and time; (3) the definition of the systems boundaries; and (4) the use of both midpoint and endpoint indicators. Firstly, a LCA is not based on the actual environmental impacts of a product but on the potential environmental impacts (Amadei et al., 2021). This creates a level of abstraction as the impacts have not occurred but could occur. Secondly, the environmental impacts found in a LCA occur over the different phases of the life cycle of the product thus over space and time (Pizzol et al., 2015). This creates a challenge for monetization because it is difficult to determine generalizable monetization factors (Pizzol et al., 2015). Thirdly, the system boundaries have not only to be determined for the technical system involving the inputs and outputs regarding the life cycle of the product, but also for the environmental system. That is to say that a boundary has to be fixed on which impacts to select, where and when (Steen, 2016). This is also related to the final challenge of environmental valuation because the boundaries of the environmental system must determine which indicators are used and how far in the future the impacts are being taken into account. The valuation of LCAs is challenging because of the level of abstraction of impact indicators. Indeed, a LCA commonly uses both midpoint and endpoint indicators. Midpoint indicators tend to be measured bottom up as they are commonly a causal effect of the emissions and the effect. Endpoints indicators are however involving impacts over longer time periods and are more complex such as human health (Pizzol et al., 2015). The monetization factors for these two types of impacts often need different monetization methods because of their differences in nature (Pizzol et al., 2015). Therefore, LCA monetization can be challenging due to its level of abstraction, the aspect of space and time as well as the differences between indicators.

True costs are used as a valuation of externalities in LCAs and commonly used as a tool to (1) calculate the financial compensation for externalities; and (2) assist decision making. True costs can indeed monetize externalities to estimate the environmental damage from a financial point of view. This can be used by stakeholders as a quantification for the compensation for externalities. Previous research elaborated on the purpose of true costing in decision making. The valuation of impact categories is a tool for decision making. Indeed, monetizing impact categories permits the financial comparison (in euros in this study) between and across impact categories that originally had different units. This method facilitates understanding for practitioners and non-practitioners (Schneider-Marin & Lang, 2020). Moreover, monetization factors give weights to the impact categories which can help to evaluate the magnitude of environmental impacts and ultimately to make tradeoffs (Durão et al., 2019; Pizzol et al., 2015).

The true cost method is composed of four stages (Future Diaper Project, 2022). The first stage is framing where the goal and audience of the true cost analysis is identified. Then, in the scoping stage, the product is described, and its lifecycle and relevant environmental impacts are specified. During the third stage, measuring and valuing, the impacts are quantified, valued and integrated. Reporting is the last stage; the interpretation of the results take place and are reported to the intended audience (Future Diaper Project, 2022). The present study focuses mostly on the scoping and valuing stage in which the impact categories and monetization factors are selected.

The valuation of environmental impacts also brings some challenges. For instance, ethical concerns include the objection of monetization on intangible aspects such as human life. However, Pizzol et al. (2015) argues that marginal changes are considered which would include a small change in a human's life for instance. Also, social impacts are monetized which creates an ethical debate. The scope of this study however limits itself to environmental impacts. Besides ethical challenges, monetization includes social, political and economic aspects which can be seen as controversial (Eldh & Johansson, 2006). For instance, the assumption that valuation based on hypothetical markets is representative of the actual value is challenged which questions the usage of true costing for decision making (Shabman & Stephenson, 2000). Despite the critics the monetization of environmental externalities in the form of true costs is becoming an agreed upon method for decision making and communication.

2.2. Impact categories and monetization factors

The selection of impact categories is dependent on the scope, stakeholders, and available information, but the consequences of this selection on true costing remains unknown. Impact categories can be identified by conducting a LCA of the product where special attention is given to the quantification of environmental damages (Matthews & Lave, 2000). The selection of impact categories depends on the scope of the study and the stakeholders involved (Souza et al., 2015). Besides the fact that stakeholders can be difficult to contact and involve, the lack of information on some impact categories also hinders the selection (Amadei et al., 2021).. For instance, the impact category global warming is universally used whereas land use is often disregarded (Amadei et al., 2021). Therefore, the selection of impact categories is difficult, resulting in the usage of different impact categories per study. It remains however unknown if the selection of impact categories influences the true cost results.

The monetization factors vary substantially within an impact category (Amadei et al., 2021) and the effect of the selection of monetization factor on true costing calculation remains unclear. Valuation of LCA approaches use various cost calculations thus establish different monetization factors for the same impact category, even when using the same calculation method there is a large variability in monetization factors (Arendt et al., 2020; Durão et al., 2019; Pizzol et al., 2015). The review on monetary valuation of LCAs by Amadei et al. (2021) provides an overview of the many monetization factors associated to impact categories. The author mentions the lack of universality and consensus on the usage of monetization factors which may affect the true cost calculations both within an impact category and between impact categories (Amadei et al., 2021). Monetization factors are selected by the practitioners to calculate their true cost, but the selection of monetization factors lacks guidance. There is a need of guideline in terms of what monetization factors to use in a specific situation and how to use them (Amadei et al., 2021). These missing guidelines result in the sometimes-arbitrary selection of monetization factors. This is for instance the case for the Future Diaper Project because the TP uses their own monetization factors without argumentation or questioning this decision. Arendt et al. (2010) highlights the importance of the selection of monetization factors and recommends varying monetization methods for one product in order to evaluate the robustness of the analysis.

It is argued that the three following factors affect monetization factors: (1) cost calculation; (2) geographical context; and (3) discount rate (Arendt et al., 2020).

Depending on the type of externality different monetization calculation methods can be used, sometimes one method fits best, in other cases a combination of methods is used (True Price, 2021). The TP differentiates restoration, compensation, prevention of re-occurrence cost and retribution. Other methods include collective consent to pay (Antheaume, 2004), travel and hedonic costs (Pizzol et al., 2015), abatement cost, budget constraint, averting behavior, contingent valuation and market price (Amadei et al., 2021). This exhaustive list of cost calculations can be completed with revealed and stated preferences of individuals (Eldh & Johansson, 2006), or the usage of taxes (Eldh & Johansson, 2006) or environmental priority strategies (Arendt et al., 2020). Using different calculation methods often entails the usage of different variables and data which results in differences of monetization factors per impact category.

Other differences of monetization factors may occur because environmental impact may in some cases be impacted by the geographical context. The TP mentioned that geographical adaptation was a limitation to their results because they only provide one global average monetization factor per impact category (True Price, 2021). In fact, it is stated that monetization factors should be adapted to regional data due to differences in impact per location (True Price, 2021). An example of the importance of location in the establishment of monetization factors is pointed out by the World Wide Fund for Nature as they state that water scarcity is very dependent on the watershed and the location (WWF Germany, 2023). Also, the local density of population, pollution level and pollution limits can impact the monetization factor (*Environmental Prices Handbook EU28 Version*, 2018). This causes monetization factors to differ when analyzing the impacts in different geographical contexts (True Price, 2021).

Another factor affecting monetization factors is the discount rate. The discount rate chosen can influence the value of the monetization factor. In literature two approaches can be found to estimate a discount rate: the prescriptive and descriptive approach (Perman et al., 2011). The prescriptive approach entails that an approximation of the utility discount, the elasticity of marginal utility and the consumption growth rate (Perman et al., 2011). These estimations can be difficult to calculate, therefore economists may disagree on the discount rate emerging from these variables (Perman et al., 2011). The descriptive approach relies on the preferencesatisfaction utilitarianism and economists following this way of reasoning tend to come up with higher discount rates than when following the prescriptive approach (Perman et al., 2011). The descriptive approach estimates the discount rate by finding an interest rate that aligns with both the discount rate of individuals' consumption and the marginal return rate of investment (Perman et al., 2011). Economists using these two approaches tend to come up with positive discount rates but often disagree on the fundamentals of the discount rate calculations (Perman et al., 2011). It is even argued that positive discount rates should not be used for environmental impact cases because it gives less weight to the future generations, but all generations should be taken into account evenly (Perman et al., 2011). Moreover, Weitzman (1998) argues that for the far future the lowest possible discount rate should be used in order to correct for the uncertainties involved in identifying future discount rates especially for environmental impacts. The choice of a discount rate is a challenge as it compares damages in the future and the current damages (Baltussen et al., 2021). High discount rates assign low value to future impacts (Clark, 2022).

The usage of different monetization factors in true cost calculations in terms of the aforementioned three categories could therefore help in finding the influence of the monetization factors on true costing.

2.3. Impact and true cost estimates of the disposable diapers and cloth diapers

Two important studies have been carried out to compare disposable diapers and cloth diapers. The first study is the Future Diaper Project conducted by TP and evaluates the true costs of both disposable and cloth diapers in 2022, making it one of the most recent and relevant European studies on cloth diapers (Future Diaper Project, 2022). The second study, conducted by the Environmental Agency (EA), is a LCA that was utilized by the United Nations in March 2021 as the sole Europea LCA for their meta-analysis on diaper alternatives (United Nations, 2021). Therefore, the 2008 LCA by EA is considered a reliable source of data for research on European disposable and cloth diapers.

The Environmental Agency (2008) estimated that the difference in true costs between disposable and cloth diapers is very small. Many assumptions are made in this study; therefore, the author performed several sensitivity analyses to have more information on the impact of these assumptions. This matters as the assumptions that have a major influence on the result may be re-examined. Previous research by the Environmental Agency (2008) and the True Price (2022) has elaborated on the several sensitivity analyses of several factors; (1) the weight of the diapers; (2) the temperature at which they are washed and dried; (3) whether the diaper is used on a second child and (4) waste management. In terms of weight of the disposable diaper, the EA has calculated the externalities of the disposable diaper if it would be 10% lighter which was a prediction for the future. The lighter the disposable diaper gets, the less material was needed in the production stage and the more the impact across all impact categories decreased (Environmental Agency, 2008). Regarding washing and drying, the TP has calculated several scenarios when varying washing temperatures (Future Diaper Project, 2022). The baseline used in the Future Diaper Report is 60 degrees. When varying from 40 degrees to 90 degrees the true cost of the cloth diapers increased but stayed lower than the true cost of the disposable diapers (Future Diaper Project, 2022). When tumble drying is used, true costs became almost equal between the cloth and disposable diapers (Future Diaper Project, 2022). The EA specifies that if the diapers were washed in a fuller load and line dried, the global warming impact decreased by 40% (Environmental Agency, 2008). The impact of the user in terms of pre-washing, washing and drying can have an impact on the true cost ranking of the disposable and cloth diaper mainly due to electricity and water use (DEFRA Science, 2023). The reuse of cloth diapers is also an important factor in true cost calculations. The usage of the cloth diaper for a second child decreased the externalities across all impact categories (Environmental Agency, 2008). In terms of waste management, when recycling is increased from 2% to 50%, the true cost of the disposable diapers decreased by 6% and the cloth diapers by 4% (Future Diaper Project, 2022). The EA confirmed that waste management options such as landfilling had less impact than the manufacturing of disposable diapers (Environmental Agency, 2008). The results in terms of true cost of the disposable and cloth diaper remain close which suggest that other factors may have a decisive impact on the diaper trade-off.

While both studies analyzed the impact of diapers on climate change and fossil fuel depletion, they differed in their selection of other impact categories. TP's analysis included land occupation, scarce blue water usage, water pollution, and underpayment, while EA studied acidification, eutrophication, freshwater aquatic ecotoxicity, human toxicity, and photochemical oxidation. These variations in impact categories suggest that further research is needed to fully understand their impact on true cost results.

The TP used their own monetization factors in the Future Diaper Project. The TP has established one monetization factor per impact category (True Price, 2021). The monetization factors of the TP are based on the costs of restoration, compensation, prevention of recurrence and/or retribution throughout the lifecycle of the product (True Price, 2021). The choice of the type(s) of costs is dependent on its relevance for the externality. Restoration cost assumes that it is feasible to achieve the same state as before the externality took place. When restoration is impossible, the cost of compensation is used to compensate economically for the environmental or social impact that occurred. Then, there is the cost of prevention of recurrence, which corresponds to the cost of aversion, avoidance, or prevention to make sure that the environmental impact cannot be repeated. The last type of cost that can be used in the calculation of the monetization factor is the cost of retribution which compensates for the breaking of law or norms to society (True Price, 2021). The selected types of cost are chosen by the TP, thus subjective to their understanding of the problem, its context, and priorities. As aforementioned, there is a large variety of monetization factors to choose from. Therefore, it is necessary to evaluate the impact of selecting different monetization factors on true cost results.

3. Methodology

This research has employed a combination of qualitative and quantitative methods to facilitate the description and quantification of impact categories and monetization factors. The study was exploratory in nature, as it aimed to examine the influence of impact categories and monetization factors on the trade-off in terms of true cost between disposable and cloth diapers. Both descriptive and numerical secondary data were collected from a range of sources, including reports and websites from gray literature as well as academic literature retrieved on Google Scholar, Scopus Connected Papers. A summary of the methodology for both research questions is presented in Table 1. Each step of this methodology is further explained in the Section 3.1. and Section 3.2..

Table 1: Summary of the methodology with the steps, research methods, data source and data analysis per research question

Research question	Steps	Research methods	Data Source	Data Analysis
1	Variations of impact categories	LCA study selection	TP and EA reports	-Alignment of functional units and assumptions -Presenting the externalities based on EA's LCA considering the alignment of assumptions
		Monetization of EA LCA	TP monetization factors and Scopus	-Reporting monetization factors for the impact categories of the EA
		Presentation of the true costs of the diaper alternatives for the EA	EA report, TP monetization factors	-Multiplying the monetization factors by the corresponding externalities -Adding all monetized externalities per diaper alternative to calculate the total true costs
	The ranking in true costs when varying impact categories	Comparison of the true cost of the disposable and cloth diaper in both studies	TP and EA reports	-Ranking the true costs of the diaper alternatives per LCA -Comparing the two rankings
2	Alternative monetization factors	LCA study selection	EA report	-Gathering the true cost results of research question 1 and impact category selection for the variations of monetization factors
		Report alternative monetization factors	Gray literature and Academic literature retrieved from Google Scholar, Scopus and Connected Papers	-Reporting the alternative monetization factors categorized in calculation method, geographical context, and discount rate
	Sensitivity analysis	Sensitivity analysis	Monetization factors presented in the previous step	-Creating a range of the monetization factors presented before per impact category as an input for the sensitivity analysis -Performing a sensitivity analysis of the monetization factors on the absolute difference between the true cost of the disposable and cloth diapers

3.1. Research question 1

To achieve differences in impact categories, two LCAs were chosen that examined the externalities of the disposable and cloth diapers based on one toilet trained child. The LCAs by the TP and EA were selected for the resent research (Environmental Agency, 2008; Future Diaper Project, 2022). The LCA by the TP is an external report commissioned by the Future Diaper Project and was retrieved from the True Price website. The LCA by the EA was found while examining the most recent European report on diaper alternatives by the United Nations (United Nations, 2021). To compare the two LCAs, all data was normalized to the same functional unit: one toilet trained child. Indeed, the functional unit of one toilet trained child accounts for the differences un number of disposable and cloth diapers used. To facilitate the comparison of the two LCAs, the assumptions of both studies were compared and aligned. Necessary adaptations were made to the data of the EA LCA to align assumptions on the reduction of weight of the disposable diaper and the increased efficiency of washing of the cloth diaper. The data for these adaptations were taken from the predicted changes mentioned in the EA LCA (Environmental Agency, 2008). The externalities per impact category by the EA, while taking into account the previously mentioned changes in assumptions, were presented in a table.

After the selection of these two LCAs, monetization factors were selected to monetize the impact categories of the EA LCA. Indeed, the TP had already monetized their LCA. Because the focus lied on the variation of impact categories, there was a correction for variation in monetization factors. Indeed, one source of monetization factors was chosen to monetize all impact categories. The selection of monetization factors limited itself to the monetization factors established by the TP. The TP monetization factors were chosen because they were used by the TP for the monetization of their impact categories and that they were also applicable to the impact categories used by the EA. For the impact categories that only occurred in the LCA by the EA, the TP monetization factors were also used. It is assumed that by using the same source for monetization factors, the impact of the monetization factors on the comparison in true costs is minimal. For the impact categories human toxicity and abiotic resources, the monetization factors of the TP could not be used due to differences in units used to quantify the impact categories. To monetize these impacts a meta-analysis by Amadei et al. (2021) was retrieved from Scopus by searching on “monetary valuation”.

The true costs of the disposable and cloth diaper per toilet trained child based on the LCA by the EA were presented. These true costs were calculated by multiplying the monetization factors of the TP by the externalities in the LCA. This method was already used by the TP to monetize their externalities of their LCA on disposable and cloth diapers. For the TP LCA, the social impact category “underpayment” was deducted from the calculated total true cost because this study’s scope limits itself to environmental impacts, thus does not include social impacts.

In order to answer the first research question, the true costs of the disposable and cloth diaper were ranked and compared for both LCA studies. If the rankings in true cost of the diaper alternatives between the two LCA studies were identical, there were no considerable influence of the selected impact categories on the ranking in true costs of the disposable and cloth diaper.

3.2. Research question 2

For the variation of monetization factors, one LCA was used. The focus lies on the LCA by the EA. Indeed, for this second part of this research the LCA by the TP is not used because the data on externalities is presented in different functional units such as per kilogram of raw material or per wash instead of per toilet trained child which makes the reproduction of the true cost calculation unattainable. Therefore, the LCA of the EA and the true cost calculation established in research question 1 were used. The impact categories chosen for the variations of monetization factors were global warming, acidification, and eutrophication. These impact categories were chosen because for one of them the disposable diapers had lower true costs than the cloth diaper, for the other impact category the disposable diapers had a higher true cost and for the third both diapers had almost equal true costs according the first research question.

The alternative monetization factors were found through (1) Google Scholar; (2) Scopus; and (3) Connected Papers. In fact, through Google Scholar the search on "monetary valuation in life cycle assessments" led to the study by Amadei et al (2021) which builds on the research of Pizzol et al. (2015). The relevant papers cited by Pizzol et al. (2015) and Amadei et al (2021) were added the selected papers by searching them through Scopus. In order to find other sources for alternative monetization factors, the keywords "monetization factors", "monetization", "life cycle assessment", "valuation of externalities, "environmental costs", "calculation methods", "location", "geographical context" and "discount rate" were used to collect literature on Google Scholar and Scopus. As an additional selection of data, the study by Amadei et al. (2021), being the most relevant for this research, was entered in Connected Papers. Connected Papers is a tool to visualize graphically similar papers to the one entered in the program. This tool is used to check for any relevant papers that were missed during the previous data selection methods. For instance, via this tool, the study by Schneider-Marin and lang (2020) on the "Environmental costs of buildings: monetary valuation of ecological indicators for the building industry" was found. The alternative monetization factors are researched using three categories according to the study by Arendt et al. (2020): cost calculation, geographical context, and discount rate. This background information as found together with the monetization factors in the selected papers. When this information was missing, this was noted with N/A meaning not available.

After the presentation of alternative monetization factors a sensitivity analysis was conducted. As inputs for the sensitivity analysis a range of monetization factors per impact category was provided based on the previously mentioned alternative monetization factors. Therefore, a minimum and maximum monetization factor was presented per impact category. These values were included in a sensitivity analysis on the absolute difference in true cost of the disposable and cloth diaper per toilet trained child. The true costs per diaper alternative was calculated by adding the true costs presented in research question 1 and changing the monetization factors for global warming, eutrophication, and acidification by the alternative monetization factors. The absolute difference in true costs is the difference in total true cost of the disposable diapers and cloth diapers. If the absolute differences were positive, it was concluded that, based on this study, the monetization factors did not influence the ranking on true costing of the disposable and cloth diapers.

4. Results

4.1. Impact categories and their influence on true costs

4.1.1. Variations of impact categories

The functional unit was essential in the comparison of the LCA by TP and the EA. The TP used “one toilet trained child” as a functional unit (Future Diaper Project, 2022). The EA established their LCA based on the functional unit "the use of nappies during the first two and a half years of a child's life” (Environmental Agency, 2008). The EA concluded from a survey that they conducted that 95% of children in the UK are out of diapers at age 2.5 (Environmental Agency, 2008). It was therefore assumed that “the use of nappies for the first two and a half years of a child’s life” is equivalent to “one toilet trained child”. Therefore, the functional unit for this research is “one toilet trained child”.

Besides the importance of the functional unit, other assumptions made in the LCAs were compared and aligned. The LCA was developed by the EA in 2005 (Environmental Agency, 2005) was revised in 2008 (Environmental Agency, 2008). However, due to advancements in design and technology, the data was somewhat outdated compared to the LCA by TP. Therefore, this study assumed some predicted changes mentioned by the EA in their LCA. Regarding disposable diapers, it is assumed that they have become 10% lighter over time. The EA has provided data for a diaper that is 10% lighter than their baseline. This was a predicted change according to the EA because the weight of the diaper had already decreased by 13,5% from 2001 to 2008 due to modifications in terms of production and design (Environmental Agency, 2008). Manufacturers have indeed decreased the weight of disposable diapers since 2008, resulting in economic and environmental benefits due to the reduced use of resources and waste production (EDANA, 2019). For cloth diapers, this thesis assumed a higher energy efficiency for the washing machine and dryer compared to the baseline calculations of the LCA by the EA (Environmental Agency, 2008). Since 2008, energy efficiency has become a more significant consideration in limiting costs and environmental impact. The higher energy efficiency is a predicted change by the EA and data was taken from their report (Environmental Agency, 2008).

With the aligned functional unit and assumptions, Table 2 shows the externalities per impact category of the LCA by the EA. The disposable diaper had higher environmental impacts than the cloth diaper on the following impact categories: abiotic resources, acidification, eutrophication and photochemical oxidation. For freshwater aquatic ecotoxicity, global warming potential and human toxicity the disposable diaper had lower environmental impacts than the cloth diaper. The impacts of the disposable and cloth diapers were difficult to compare across impact categories because of the different units used and the differences in weight assigned to impact categories. In order to compare the total environmental impact of the disposable and cloth diaper, the impact categories of the EA were monetized.

Table 2: Impact assessment of disposable diapers and cloth diapers per toilet trained child (EA, 2008)

Externalities	Disposable diaper	Cloth diaper
Abiotic resource (kg Sb-eq [a])	4.1	3.8
Acidification (kg SO-eq [b])	3.1	1.9
Eutrophication (kg PO_4-eq [c])	0.37	0.3
Freshwater aquatic ecotoxicity (kg 1,4-DB-eq [d])	1.9	4
Global warming (GWP100, unit kg CO_2-eq [e])	509	519.7
Human Toxicity (kg 1,4-DB-eq [d])	56	68.4
Photochemical oxidation (kg C_2H_4 [f])	0.19	0.1

[a] g Sb-eq: kilogram of Antimony equivalent
[b] kg SO-eq: kilogram of Sulphur Oxide equivalent
[c] kg PO_4-eq: kilogram of Phosphate equivalent
[d] kg 1,4-DB-eq: kilogram of 1,4 Dichlorobenzene equivalent
[e] kg CO_2-eq: kilogram of Carbon Dioxide equivalent
[f] kg C_2H_4: kilogram of Ethene

To calculate the true cost of the disposable and cloth diaper, monetization factors were selected among the monetization factors of the TP (True Price, 2021) when possible and presented in Table 3. The TP monetization factors were used for (1) acidification; (2) eutrophication; (3) freshwater aquatic ecotoxicity; (4) global warming; and (5) photochemical oxidation. For acidification the TP established the monetization factor 4.70 EUR/kg SO_2-eq (True Price, 2021). Concerning eutrophication it was assumed that 3.07 kg of phosphate (PO_4) corresponds to 1 kg of phosphorus (P) (Iheagwara et al., 2013). The eutrophication level for the disposable diapers is 0.37 kg PO_4 thus 0.121 kg P-eq and for the cloth diapers 0.3 kg PO_4 resulting in 0.098 kg P-eq. The TP monetization factor of 203.00 EUR/kg P-eq was used for the monetization of eutrophication (True Price, 2021). With regards to freshwater ecotoxicity the TP used the monetization factor 0.041 EUR/kg 1,4-DB-eq (True Price, 2021). Concerning global warming the TP established the monetization factor of 0.157 EUR/kg CO_2-eq (True Price, 2021). For photochemical oxidation, it was assumed that the TP monetization factor in EUR/ kg Non-Methane Volatile Organic Compounds (NMVOC) can be used because photochemical oxidation is presented in Ethene (C_2H_4) by the EA and Ethene is a NMVOC (Guenther, 2000).

For human toxicity and abiotic resources, the TP monetization factors were not used due to differences in unit between the impact category and the monetization factor. This resulted in the usage of other monetization factors. The monetization factor by the TP for human toxicity was expressed in Disability Adjusted Life Years (DALY). In the LCA by the EA, the unit kilograms of 1,4 Dichlorobenzene equivalent (1,4-DB-eq) were used to quantify human toxicity. So, for human toxicity, the monetization factor by the TP was unsuitable for the valuation of the LCA by the EA. Therefore, the monetization factor of 0.123 EUR/kg 1,4-DB-eq from the meta-analysis by Amadei et al. (2021) was used. Furthermore, the impact category abiotic resources corresponded to fossil fuel extraction (True Price, 2021) and is measured in kilograms of Antimony equivalent. However, the monetization factor for the extraction of oil, gas, coal reserves is given in euros per kilogram of oil equivalent by the TP which cannot be converted into kilograms of Antimony equivalent. Thus, the monetization factor by the TP was not applicable for the impact category abiotic resources in the LCA of the EA. No alternative monetization factor was found in the unit "kilograms of Antimony equivalent". Therefore, it was assumed that it was relevant to use the true cost of fossil fuel extraction by the TP in the Future Diaper report (Table 4).

Table 3: Baseline monetization factors per impact category

Impact category	Baseline Monetization Factor	Unit Baseline Monetization factor
Acidification	4.70	EUR/kg SO_2-eq [a]
Eutrophication	203.00	EUR/kg P eq [b]
Freshwater aquatic ecotoxicity	0.041	EUR/kg 1,4-DB-eq [c]
Global warming	0.157	EUR/kg CO_2-eq [d]
Human Toxicity	0.123	EUR/kg 1,4-DB-eq [c]
Photochemical oxidation	0.83	EUR/kg NMVOC [e]

[a] EUR/kg SO-eq: euros per kilogram of Sulphur Oxide equivalent
[b] EUR/kg P-eq: euros per kilogram of Phosphorus equivalent
[c] EUR/kg 1,4-DB-eq: euros per kilogram of 1,4 Dichlorobenzene equivalent
[d] EUR/kg CO_2-eq: euros per kilogram Carbon Dioxide equivalent
[e] EUR/ kg NMVOC: euros per kilogram of Non-Methane Volatile Organic Compounds

The monetization factors of Table 3 were multiplied to the corresponding externalities of Table 2 to calculate the true cost of the disposable and cloth diapers per impact category and in total in Table 4. The total true cost was the summation of all true costs presented. The total true cost for one toilet trained child for the disposable diaper was 201.05 euros whereas for the cloth diaper it was 161.01 euros.

Table 4: True costs (EUR) of disposable diapers and cloth diapers per toilet trained child based on the impact categories of the LCA by the EA

True Cost (EUR)	Disposable diaper	Cloth Diaper
Abiotic resource	99.40	61.80
Acidification	14.57	8.93
Eutrophication	0.04	0.03
Freshwater aquatic ecotoxicity	0.08	0.16
Global warming	79.91	81.59
Human Toxicity	6.89	8.41
Photochemical oxidation	0.16	0.08
Total	201.05	161.01

4.1.2. The ranking in true costs when varying impact categories

The TP had established that the true cost of the disposable diaper is higher than the cloth diaper as presented in Appendix 1 (Future Diaper Project, 2022). The TP also selected a social impact category which was underpayment. This impact was not considered in the present study as this study limited itself to the environmental impacts. Therefore, the true costs calculated by the TP based on environmental impacts were 350 euros for the disposable diaper and 240 euros for the cloth diaper. As a result, the disposable diaper has a higher true cost than the cloth diaper. This ranking in true cost was consistent with the findings using the LCA of the EA. Indeed, in both studies the disposable diaper has a higher true cost than the cloth diaper. Despite that the ranking in true costing remained the same between the two studies, the gap between the true cost of the disposable and cloth diaper has become smaller for the EA. The TP has calculated a gap of 110 EUR between the disposable and cloth diaper. For the EA this difference is 40.04 EUR. Also, the total true costs by the EA of the disposable and cloth diapers are lower than those calculated by the TP.

4.2. Alternative monetization factors

Several monetization factors for global warming, eutrophication and acidification are presented in Table 5 and categorized into differences in calculation method, geographical context, and discount rate. Global warming, acidification and eutrophication are being defined as follows. Global warming occurs through the emission of greenhouse gases, principally by human activities, causing worldwide problems (IPCC, 2023). The greatest contributors in terms of human activities are the unsustainable energy use, land use, lifestyles and consumption (IPCC, 2023). The increase in temperatures do not only affect the environment and the ecosystems but also human health and economies (IPCC, 2023). To help limit the emissions the Paris Agreement has set the 2-degree scenario and is pursuing a 1.5-degree scenario of temperature increase (United Nations Framework Convention on Climate Change, n.d.). Eutrophication is a natural phenomenon where the aquatic ecosystems' evolution boosts the production of organic materials (Centre Nationale de la Recherche Scientifique, 2017). This water pollution is mostly caused by phosphorus and nitrogen. Eutrophication also affects air and soil quality. This has severe consequences for the environment, the economy and human health. Acidification is a form of air pollution generally expressed in sulphur dioxide. Acidification affects ecosystems therefore damaging mostly biodiversity and the functioning of coral reefs (Miller & Spoolman, 2021).

4.2.1. Cost calculation

Several monetization factors for global warming were presented as the monetization of this impact category involves a lot of uncertainty and varies substantially. The two commonly used calculation methods for global warming are Marginal Abatement Cost (MAC) and Social Cost of Carbon (SCC). A range for these two-calculation method was given for global warming. The TP monetization factor for this impact category was 0.157 EUR/kg CO_2-eq (GW6) based on the MAC (True Price, 2021). This value is calculated based on a meta-analysis on MAC by Kuik et al. (2009). This meta-analysis considered 68 MAC estimates and presented them with a minimum of 1.40 EUR/ton CO_2-eq (GW2) and a maximum of 209.40 EUR/ton CO2-eq (GW7). Another method commonly used to value the expected damages from carbon dioxide equivalent is the SCC (Ricke et al., 2018). The SCC is an economic valuation of the marginal impacts of climate change and can be used for the global welfare perspective. A meta-analysis based on 58 studies established a ranged of SCC from -13.36 USD/ton CO_2-eq to 2386.91 USD/ton CO_2-eq (Wang et al., 2019). In euros this translated into -12.54 EUR/ton CO_2-eq (GW1) and 2239.69 EUR/ton CO_2-eq (GW9) (Guagenti, 2023). The other monetization factors by US EPA (2017) were located within the aforementioned range: 10 EUR/ton CO_2-eq (GW3), 40 EUR/ton CO_2-eq (GW4), 60 EUR/ton CO_2-eq (GW5).

To establish the monetization factor for eutrophication of 203 EUR/kg P-eq (E7), the TP used a combination of restoration and compensation costs (True Price, 2021). Restoration costs were also used in combination with willingness to pay and prevention costs which resulted in the range of monetization factors between 8 EUR/PO_4-eq (E3) and 20 EUR/PO_4-eq (E6) (OVAM, 2017). An alternative to establish the monetization factor for eutrophication was usage of contingent valuation (Ahlroth, 2009; Ahlroth & Finnveden, 2011). Converted into euros, this gives a monetization factor of 18.52 EUR/PO_4-eq (E5) (Ahlroth, 2009; Schneider-Marin & Lang, 2020). Another author calculated monetization factors for eutrophication by using damage costs and abatement costs which resulted respectfully in 1.8 EUR/PO_4-eq (E2) and 11 EUR/PO_4-eq (E4).

The monetization factors for acidification of 4.7 EUR/kg SO-eq (A7) by the TP were calculated in terms of compensation cost where data was used from the value of ecosystem services (True Price, 2021). Another method for the establishment of monetization factors for was contingent valuation and resulted in 2.73 EUR/kg SO-eq (A6) (Ahlroth, 2009). Bruyn et al (2010) however identified higher monetization factors for acidification that were based on abatement cost and damage cost and respectfully resulted in 5 EUR/kg SO-eq (A8) and 15.4 EUR/kg SO-eq (A9). In contrast, the lowest value reported in this research was reported by Arendt et al. (2020) and was calculated using Environmental Priority Strategies. OVAM (2017) found slightly higher monetization factors being 0.5 EUR/kg SO-eq (A3) and 2.02 EUR/kg SO-eq (A4) and were based on damage costs and restoration costs. Another, more unconventional, calculation method was used by Eldh & Johansson (2006) which is the usage of the tax on Sulphur content in fossil fuels, this resulted in a monetization factors of 2.26 EUR/kg SO-eq (A5).

4.2.2. Geographical context

An impact category that is not location dependent is global warming. Monetization factors are based on the environmental impacts; therefore, the location of environmental impact could be the same or different from the location where the externality occurs. The emissions of greenhouse gases can be more severe in some places than others (Ricke et al., 2018) but the environmental impact is global. Indeed, monetization factors were reported globally by Wang et al. (2009), Kuik et al. (2009) and True Price (2022). Other sources did not mention the geographical location for the monetization factors of global warming.

The monetization factors for eutrophication were dependent on the geographical context. The TP argued that to calculate country specific eutrophication factors, the water basin-level risk of eutrophication of that country would be used (True Price, 2021). Also, the country's GDP had been found to influence willingness to pay for water quality, in this case eutrophication (Ahtiainen, 2009). Different monetization factors have been established by a Belgium study which differentiated western Europe eutrophication and the rest of the world (OVAM, 2017). This resulted in 20.00 EUR/kg of PO_4-eq (E6) in Western European countries and 8.00 EUR/kg of PO_4-eq (E3) in the rest of the world. These values were based on existing literature as well as the differences in GDP per capita in order to have location specific data (OVAM, 2017). Despite the location specific monetization factors the TP established a global monetization factor for eutrophication which was 203 EUR/P-eq (E7).

Acidification is a form of air pollution thus can be seen as a transboundary environmental problem (*Environmental Prices Handbook EU28 Version*, 2018). Despite that this type of pollution is transported over long distances and thus does not only impact the location where the gases are emitted but also other locations, acidification can be location dependent. In Belgium a minimum and maximum were established in contrast with the European monetization factors as it was argued that differences occur in terms of impact of emissions due to wind direction and speed but also due to differences in land use and precipitation for example (OVAM, 2017). It was established that the Belgium monetization factors are between 0.5 EUR/kg SO-eq (A3) and 2.02 EUR/kg SO-eq (A4) (OVAM, 2017). Another approach on geographically dependent monetization factors is to use local taxing. For instance, in Sweden a monetization factor was derived from the Swedish tax on the sulphur content in fossil fuels which resulted in 30 SEK/kg SO-eq (Eldh & Johansson, 2006) which was equivalent to 2.26 EUR/kg SO-eq (A5). In contrast, the TP established a global monetization factor for acidification which was 4.7 EUR/kg SO-eq (A7) (True Price, 2021).

4.2.3. Discount rate

For global warming, discount rates of 2.5%, 3% and 5% were chosen, the monetization factors for global warming in 2025 were respectfully 68 $/ton of CO_2-eq, 46 $/ton of CO_2-eq and 14 $/ton of CO_2-eq (US EPA, 2017). These values correspond respectfully to 60 EUR/ton CO_2-eq (GW5), 40 EUR/ton CO_2-eq (GW4) and 10 EUR/ton CO_2-eq (GW3). As stated before, some authors argue for a 0% discount rate when addressing the environment to give an equal weight to the current and future generations. Bruyn et al (2010) found a monetization factor using 0% discount rate of 395 EUR/kg CO_2-eq (GW8) for global warming.

Concerning eutrophication, Bruyn et al. (2010) has elaborated on the differences of monetization factors when varying discount rates. In fact, when a 2.5% discount rate was used, the monetization factors found were 11 EUR/kg PO_4-eq (E4) and 1.80 EUR/kg PO_4-eq (E2). The differences in results also emerge from the different calculation method as the first monetization factor is based on abatement costs and the second on damage costs. When 0% discount rate is used, the monetization factor found was 1.78 EUR/kg P-eq (E1) (Bruyn et al., 2010).

For acidification, Bruyn et al. (2010) mentioned the monetization factors 5 EUR/kg SO-eq (A8) for abatement cost and 15.40 EUR/kg SO-eq (A9) for damage costs when using 2.5% discounting. When 0% discount rate is used, Bruyn et al. (2010) found the monetization factor 0.23 EUR/kg SO-eq (A2). In contrast, Arendt et al. (2020) used 0% discount rate and found 0.01 EUR/kg SO-eq (A1). The difference between the results of both studies may lay in the differences in calculation method being respectfully ReCiPe and Environment Priority Strategies.

Table 5: Alternative monetization factors for global warming, eutrophication and acidification ant their calculation methods, geographical contexts and discount rates (N/A: Not Available)

Impact category	Monetization factor	Monetization factor code	Source	Calculation method	Geographical context	Discount rate
Global Warming (EUR/ton CO2-eq)	-12.54	GW1	Wang et al. (2019)	SCC	Global	N/A
	1.4	GW2	Kuik et al. (2009)	MAC	Global	N/A
	10	GW3	US EPA (2017)	SCC	N/A	5%
	40	GW4	US EPA (2017)	SCC	N/A	3%
	60	GW5	US EPA (2017)	SCC	N/A	2.5%
	157	GW6	True Price (2021)	MAC	Global	N/A
	209.4	GW7	Kuik et al. (2009)	MAC	Global	N/A
	395	GW8	Bruyn et al. (2010)	N/A	N/A	0%
	2239.69	GW9	Wang et al. (2019)	SCC	Global	N/A
Eutrophication (EUR/ P-eq)	1.78	E1	Bruyn et al (2010)	N/A	N/A	0%
Eutrophication (EUR/ PO4-eq)	1.8	E2	Bruyn et al (2010)	Damage cost	N/A	2.5%
	8	E3	OVAM (2017)	Willingness to pay, restoration cost and prevention costs	Rest of the world (outside of western Europe)	N/A
	11	E4	Bruyn et al (2010)	Abatement cost	N/A	2.5%
	18.52	E5	Ahlroth (2009)	Contingent valuation	N/A	N/A
	20	E6	OVAM (2017)	Willingness to pay, restoration cost and prevention cost	Western Europe	N/A
Eutrophication (EUR/P-eq)	203	E7	True Price (2021)	Restoration cost and compensation costs	Global	N/A
Acidification (EUR/kg SO-eq)	0.01	A1	Arendt et al (2020)	Environmental Priority Strategies	N/A	0%
	0.23	A2	Bruyn et al (2010)	N/A	N/A	0%
	0.5	A3	OVAM (2017)	Damage cost and restoration costs	Belgium	N/A
	2.02	A4	OVAM (2017)	Damage cost and restoration costs	Belgium	N/A
	2.26	A5	Eldh & Johansson (2006)	Tax on sulphur content in fossil fuels	Sweden	N/A
	2.73	A6	Ahlroth (2009)	Contingent valuation	N/A	N/A
	4.7	A7	True Price (2021)	Compensation cost	Global	N/A
	5	A8	Bruyn et al (2010)	Abatement cost	N/A	2.5%
	15.4	A9	Bruyn et al (2010)	Damage cost	N/A	2.5%

4.2.4. Sensitivity analysis of the monetization factors

The list of aforementioned monetization factors (Sections 4.2.1., 4.2.2. and 4.2.3.) resulted in a range of monetization factors per impact category. The following ranges of monetization factors are being considered for the sensitivity analysis. For global warming the monetization factors -12.54 EUR/ton CO2-eq (GW1) and 2239.69 EUR/ton CO2-eq (GW9) are used as they presented the minimum and maximum of all alternative monetization factors. In terms of eutrophication the monetization factors 1.78 EUR/ P-eq (E1) and 203 EUR/ P-eq (E7) form a range. For acidification the range of monetization factors is between 0.01 EUR/kg SO-eq (A1) and 15.4 EUR/kg SO-eq (A9). A sensitivity analysis was conducted using the three ranges of monetization factors mentioned before. Table 6 presents the absolute difference in true cost of the disposable and cloth diapers per toilet trained child when varying the monetization factors for global warming, eutrophication, and acidification.

Table 6: Absolute difference (EUR) between the true cost of the disposable diaper and the cloth diaper per toilet trained child when varying from the minimum to the maximum the monetization factors global warming, eutrophication, and acidification

		GW1	GW9
E1	A1	36.25	12.15
E7	A1	40.84	12.33
E1	A9	48.20	30.62
E7	A9	59.31	35.21

When adopting the alternative monetization factors for the disposable and the cloth diapers, it was observed that no variation of monetization factors results in a change of ranking between the disposable and cloth diaper. Indeed, no combination of monetization factors resulted in a negative absolute difference between the true cost of the disposable and cloth diapers. The absolute difference in true costs between the two diaper alternatives for one toilet trained child was between 12.15 EUR, using GW9, E1 and A1, and 59.31 EUR, using GW1, E7, A9. Because all results from the sensitivity were positive values, it can be argued that regardless the monetization factors chosen for global warming, eutrophication and acidification the true cost of the disposable diapers remains higher than the one of the cloth diapers for one toilet trained child.

5. Discussion

The findings of this study are that the true cost ranking of the disposable and cloth diapers per toilet trained child did not change when varying impact categories and monetization factors introduced in this study. Indeed, the LCA of the EA replaced 4 out of 6 impact categories that were chosen by the TP. When monetizing the LCA by the EA, the disposable diapers still had a higher true cost than the cloth diapers. Furthermore, there was a large variation of monetization factors applied to the impact categories. These variations did not affect the true cost ranking of the disposable and cloth diapers.

The findings of this study are aligned with previous research on several aspects: (1) there is large variability of monetization factors per impact category; (2) the choice of monetization factors seems arbitrary; and (3) the impact of monetization factors on true cost rankings seems limited. Indeed, the findings of this study showed large variabilities of the monetization factors within one impact category. These differences were found when varying calculation methods, geographical contexts and discount rates as suggested by Arendt et al. (2020). Even while using the same calculation method, monetization factors were varying as argued by Amadei et al. (2021). For instance, the range of monetization factors for global warming varied between minus 12.54 EUR/ton CO_2-eq and 2239.69 EUR/ton CO_2-eq using the same calculation method. Furthermore, because of the large variability of monetization factors, it can be argued that studies performing a true cost analysis are being challenged by the lack of guidelines and information regarding the monetization factors (Amadei et al., 2021). In fact, during the present study many monetization factors were found but the information on calculation method, geographical context and/or discount rate was not available. The lack of transparency causes stakeholders to make arbitrary choices in terms of monetization factors. The high variability of monetization factors did however not impact the rankings of diaper alternatives. This is aligned with the results of Schneider-Marin & Lang (2020) who studied the variations of monetization factors when comparing the sustainability aspects of different buildings. These authors concluded that using minimum and maximum monetization factors in the monetization of externalities had no impact on the ranking of the most sustainable building in their study. Thus, the most sustainable building remained the same when varying monetization factors. The study by Schneider-Marin & Lang (2020) is relevant as an example to illustrate that there was no impact of monetization factors on true cost rankings. The most sustainable options remains even when weights attached to the externalities, by means of monetization, are varied substantially.

The novelty of this study lies in the evaluation of the true cost method and the influence that practitioners have on the result by using different impact categories and monetization factors for their true cost calculations. The results that were found may however be dependent on the methodology used in this study. This study has the following considerable limitations, presented in order of weight of impact on the results: (1) some impact categories were not considered in this study; (2) there might be monetization factors outside of the ranges provided; and (3) the uncertainties and assumptions in the LCAs made their comparison challenging. These limitations should be considered with regards to the usage of the information in this paper for further research.

The first limitation of this study is that few impact categories were subject to variations which could have affected the results and affect the external validity of this study. In fact, for the first part of this research 9 impact categories by the TP and EA were considered (Environmental Agency, 2008; Future Diaper Project, 2022). It is however known that outside of these LCAs, other impact categories exist and were not considered in this research. For instance, social impact categories were not included in this research. True costs are based on the monetization of social and environmental impact categories. Social impact categories involve much uncertainty, abstraction, and ethical challenges (Pizzol et al., 2015), but this study considered only environmental impact categories. Moreover, in the second part of this research, the variation of monetization factors was only applied to global warming, eutrophication and acidification. The selection of the three impact categories was made by choosing: one impact category where the disposable diaper had a higher true cost than the cloth diaper (acidification), one impact category where the cloth diaper had a higher true cost than the disposable diaper (global warming) and one impact category where the true costs of the diaper alternatives were close to equal (eutrophication) based on the first true cost calculation. Also, these three impact categories have units which are used universally which entailed that more monetization factors were available. The implication of other impact categories was unrealistic because of the lack of availability of monetization factors with the right unit and background information. For instance, the data regarding calculation methods, geographical context and discount rate for monetary valuations were often not available. This is consistent with previous research, as stated by Amadei et al. (2021) inconsistency, differences in units, missing calculation methods and the lack of availability of monetization factors created a challenge to find enough monetization factors across all impact categories. Indeed, an impact category can be quantified in a unit of choice, there is no consensus on a specific unit to use. If other impact categories were chosen for the variations in monetization factors, it is unclear whether this would be beneficial for the disposable and cloth diaper as the ranges of monetization factors have not been reported. The selection of two LCAs and some impact categories questions the external validity of this study as the findings cannot be generalized for other LCA studies or other impact categories. The selection of more impact categories and LCAs could make the findings of this study more generalizable and could lead to a more complete overview.

The second limitation of this research is that there might be monetization factors outside of the ranges provided. As aforementioned, monetization factors lack information on the background. The background information on calculation method, geographical context and discount rates is missing for many studies as can be observed by the frequent presence of N/A meaning not available in Table 5. Another barrier to finding more monetization factors is that the units vary substantially which entails that some monetization factors could not be included in this study. This is a limitation that could cause an underestimation of results. Indeed, if the ranges of monetization factors were to be extended, this could mean that the true costs of the products would vary more and that a change in ranking in terms of true costs of the diaper alternatives might take place.

A third limitation regards the inconsistencies in terms of assumptions when comparing the two LCA studies leading to an underestimation of results, and which might affect the reliability of the study. Indeed, this research used LCA data from previous studies by the TP and EA (Environmental Agency, 2008; Future Diaper Project, 2022). These LCAs were chosen because they were based on the same functional unit "one toilet trained child. Despite having the same functional unit, these LCAs had different interpretations, as the EA counted 2.5 years of diaper period for one toilet trained child resulting in 3796 disposable diapers (Environmental Agency, 2008) whereas the TP counted 3.5 years for one toilet trained child which is equivalent to 5968 disposable diapers (Future Diaper Project, 2022). The number of cloth diapers did not vary because these diapers can be washed and reused over time. As the difference in disposable diapers between studies is 2172 units, this might influence the comparison of the two studies. If the EA has considered an equal amount of disposable diapers as the TP, their true cost difference between the disposable and cloth diaper would be more substantial than the current findings, therefore not affecting the conclusion of this study because no change in ranking would occur. Another inconsistency in terms of assumptions between the two studies is the fabric used to make the cloth diapers. In fact, the EA considered nappies of 100% cotton whereas the TP considered diapers made of 29% cotton with viscose, bamboo and polyester (Environmental Agency, 2008; Future Diaper Project, 2022). As stated by the TP, 50% of the water consumption from the cloth diaper is related to the cotton cultivation. Therefore, if the diaper is made of 100% cotton the water usage would increase, affecting the true cost of the product. The differences in assumptions form inconsistencies and uncertainties in terms of the comparison of the studies thus challenges the reliability of the method used. Indeed, the uncertainties presented in the LCAs are then also applied to the true cost calculations.

In sum, it remains unknown if the limitations of this study create an underestimation or overestimation of the results, but alternative methods can be used to further examine the effect of the limitations. Firstly, an alternative to the method used in this study is to diversify the variations both in impact categories and monetization factors. In fact, not all impact categories were varied, and it is unclear which impact categories and monetization factors are most influential on the true cost results for the disposable and cloth diapers. Using more impact categories and monetization factors will lead to more conclusive results with regards to the aim of this study. A second alternative method proposed is to use externality data from one LCA source instead of two LCA sources to avoid differences in assumptions between studies. While finalizing this paper, a new LCA of the disposable and cloth diaper became available by Giraffe Innovations involving 18 impact categories (DEFRA Science, 2023). The data of this LCA could be used to reproduce the present study. The results would then show if the differences in assumptions of the TP and EA had an influence on the purpose of this research.

6. Conclusion

This study has investigated the influence of the selection of environmental impact categories and monetization factors on the ranking in true costs of the disposable and cloth diapers for one toilet trained child. According to this study the true costs per diaper alternative were dependent on the selection of impact categories and monetization factors as the values of true costs changed when using another LCA and other monetization factors than the TP's. Therefore, it could be argued that the true cost results by the TP should not be presented as one single value but as a range of possible values. Indeed, much uncertainty exists, and this is not reflected in the Future Diaper Report. Despite the range of true costs that can be attributed to the diaper alternatives, the true cost of the disposable diapers was still higher than the true costs of the cloth diapers per toilet trained child. Indeed, when varying the selection of impact categories as well as during the sensitivity analysis with the usage of different monetization factors per impact category. The two selected LCA's involving different impact categories were monetized and, in both situations, the disposable diapers had a higher true cost than the cloth diapers over a duration of one toilet trained child. Moreover, when monetization factors from various secondary literature sources were used in the establishment of true cost calculations, the cloth diaper remained the diaper alternative with the lowest true cost compared to the disposable diaper. Therefore, it could be argued that according to this study, the cloth diaper has a lower true cost than the disposable diaper based on the TP and EA LCA's (Environmental Agency, 2008; Future Diaper Project, 2022) and regardless of the monetization factor chosen per impact category.

Future research is recommended to: (1) reproduce this study on other products and their sustainable alternatives; (2) study the effect of social impact categories on the ranking in true cost of the disposable and cloth diapers; (3) add more monetization factors per impact category to the databases of the TP; and (4) study the relation between true price and purchasing behavior as well as suggestions on how to use true pricing.

For further research, the present study can serve as a format to assess the robustness of true cost rankings between products. When analyzing the true cost of two alternatives, variations in impact categories and monetization factors can help to gain more insights into the robustness of the results established by a practitioner. The gathering of previous and future comparable studies, such as the study on the ranking of sustainable buildings by Schneider-Marin & Lang (2020), may lead to more evidence to support deductive reasoning towards the fact that the selection of impact categories and monetization factors do, or do not, impact the ranking of true cost of a sustainable and unsustainable product. For stakeholders to gain confidence in the true cost method, it is helpful to perform a deductive review on the causal relation and magnitude of influence of the impact categories and monetization factors on true cost results based on examples and quantitative research. Therefore, the present research cannot be seen as proof that the impact categories and monetization factors doesn't affects the ranking in true cost of the disposable and cloth diaper but as a hypothesis that future research can re-examine and replicate.

The present research limited its scope to the selection of 9 environmental impact categories from the TP and EA. It is however unclear what the influence of other environmental impact categories as well as social impact categories would be. It is recommended to research how the selection of impact categories takeq place and to analyze which impact categories have the biggest impact on the ranking of the disposable and cloth diapers by performing a sensitivity analysis.

As a recommendation for the TP, more monetization factors per impact category could be added to their database. Indeed, according to this research a wide range of monetization factors exists per impact category which impacts the true cost results. By adding alternative monetization factors practitioners can calculate the true cost of products according to their context. A condition for achieving this is the transparency in terms of background information on the monetization factor as well as guidelines to ensure that practitioners use the monetization factors correctly. This aligned with the recommendation by Amadei et al. (2021), to create more guidelines for the selection and usage of monetization factors.

Another path for further research would be the impact of true pricing (adding true cost to market prices) on purchasing behaviors of customers. Because the market price for one toilet trained child of disposable diapers is already 500 EUR higher than the market price of the cloth diaper and that most consumers still choose to use disposable diapers it can be questioned how influential true pricing is on purchasing behaviors of consumers. It could be studied if any mediators or confounders impact the relation between price and purchasing behavior in the case of disposable and cloth diapers. This could be done from a point of view of health economics but also for instance marketing and communication. It could be researched if there were a minimum price difference that would lead the consumer to choose the cloth diapers instead of the disposable diapers. Furthermore, it could be studied how true pricing should be implemented. Indeed, true prices can be paid by the consumer, or used as a communication tool to educate the consumer on the environmental impact without payment. It is however unclear which of the two previous strategies is most effective. The mechanisms of true pricing and purchasing behavior can be studied in more detail by future research to achieve more consensus on the usage of true pricing for sustainable purchasing.

References

Ahlroth, S. (2009). Valuation of environmental impacts and its use in environmental systems analysis tools. *Doctoral Dissertation, KTH Royal Institute of Technology*.

Ahlroth, S., & Finnveden, G. (2011). Ecovalue08–A new valuation set for environmental systems analysis tools. *Journal of Cleaner Production*, *19*(17), 1994–2003. https://doi.org/10.1016/j.jclepro.2011.06.005

Ahtiainen, H. (2009). Valuing international marine resources: A meta-analysis on the Baltic Sea. *MTT Discussion Papers*.

Amadei, A. M., De Laurentiis, V., & Sala, S. (2021). A review of monetary valuation in life cycle assessment: State of the art and future needs. *Journal of Cleaner Production*, *329*, 129668. https://doi.org/10.1016/j.jclepro.2021.129668

Arendt, R., Bachmann, T. M., Motoshita, M., Bach, V., & Finkbeiner, M. (2020). Comparison of Different Monetization Methods in LCA: A Review. *Sustainability*, *12*(24), Article 24. https://doi.org/10.3390/su122410493

Baltussen, W., Goglio, P., Janssen, B., Broekema, R., & Nuhoff-Isakhanyan, G. (2021). Contribution to climate change: Impact-specific module for true price assessment; true pricing method for agri-food products. *Wageningen University & Research*.

Bruyn, S., Korteland, M., Markowska, A., Davidson, M., de Jong, F., Bles, M., & Sevenster, M. (2010). *Shadow Prices Handbook Valuation and weighting of emissions and environmental impacts*. CE Delft. https://cedelft.eu/publications/shadow-prices-handbook-valuation-and-weighting-of-emissions-and-environmental-impacts/

Centre Nationale de la Recherche Scientifique. (2017). *Eutrophication causes, mechanisms, consequences and predictability*. https://www.cnrs.fr/sites/default/files/download-file/Eutrophication_short-version_ENG_0.pdf

Chekima, B., Syed Khalid Wafa, S. A. W., Igau, O. A., Chekima, S., & Sondoh, S. L. (2016). Examining green consumerism motivational drivers: Does premium price and demographics matter to green purchasing? *Journal of Cleaner Production*, *112*, 3436–3450. https://doi.org/10.1016/j.jclepro.2015.09.102

Clark, N. H. F., Lesley. (2022, November 21). *EPA floats sharply increased social cost of carbon*. E&E News. https://www.eenews.net/articles/epa-floats-sharply-increased-social-cost-of-carbon/

Dasgupta, P. S., & Ehrlich, P. R. (2013). Pervasive Externalities at the Population, Consumption, and Environment Nexus. *Science*, *340*(6130), 324–328. https://doi.org/10.1126/science.1224664

DEFRA Science. (2023). *Life Cycle Assessment of Disposable and Reusable Nappies in the UK 2023*. https://randd.defra.gov.uk/ProjectDetails?ProjectId=20622

Durão, V., Silvestre, J. D., Mateus, R., & Brito, J. D. (2019). Economic valuation of life cycle environmental impacts of construction products—A critical analysis. *IOP Conference Series: Earth and Environmental Science*, *323*(1), 012147. https://doi.org/10.1088/1755-1315/323/1/012147

EDANA. (2019). *Sustainability Reports*. https://www.edana.org/how-we-take-action/edana-sustainability-initiatives/sustainability-reports

Eldh, P., & Johansson, J. (2006). Weighting in LCA Based on Ecotaxes—Development of a Mid-point Method and Experiences from Case Studies. *The International Journal of Life Cycle Assessment*, *11*(1), 81–88. https://doi.org/10.1065/lca2006.04.015

Environmental Agency. (2005). *Disposable and reusable nappies in the UK: Life cycle assessment*. GOV.UK. https://www.gov.uk/government/publications/disposable-and-reusable-nappies-in-the-uk-life-cycle-assessment

Environmental Agency. (2008). *An updated lifecycle assessment for disposable and reusable nappies*. GOV.UK. https://www.gov.uk/government/publications/an-updated-lifecycle-assessment-for-disposable-and-reusable-nappies

Environmental Prices Handbook EU28 version. (2018). [CE Delft]. CE Delft - EN. https://cedelft.eu/publications/environmental-prices-handbook-eu28-version/

Future Diaper Project. (2022). *Future Diaper Project—True Price of Diaper Systems*. Future Diaper Project. https://www.futurediaperproject.com/projects/true-price-of-diaper-systems

Green, L., Myerson, J., & Mcfadden, E. (1997). Rate of temporal discounting decreases with amount of reward. *Memory & Cognition*, *25*(5), 715–723. https://doi.org/10.3758/BF03211314

Guagenti, C. (2023, April 13). *USD To EUR: Convert United States Dollar to Euro*. Forbes Advisor. https://www.forbes.com/advisor/money-transfer/currency-converter/usd-eur/

Guenther, A. (2000). Natural emissions of non-methane volatile organic compounds, carbon monoxide, and oxides of nitrogen from North America. *Atmospheric Environment*, *34*(12–14), 2205–2230. https://doi.org/10.1016/S1352-2310(99)00465-3

Iheagwara, O. S., Ing, T. S., Kjellstrand, C. M., & Lew, S. Q. (2013). Phosphorus, phosphorous, and phosphate: Phosphorus, phosphorous, and phosphate. *Hemodialysis International*, n/a-n/a. https://doi.org/10.1111/hdi.12010

IPCC. (2023). *Synthesis report of the IPCC sixth assessment report (AR6)*. https://www.ipcc.ch/report/sixth-assessment-report-cycle/

Matthews, H. S., & Lave, L. B. (2000). Applications of Environmental Valuation for Determining Externality Costs. *Environmental Science & Technology*, *34*(8), 1390–1395. https://doi.org/10.1021/es9907313

Mendoza, J. M. F., Popa, S. A., D'Aponte, F., Gualtieri, D., & Azapagic, A. (2019). Improving resource efficiency and environmental impacts through novel design and manufacturing of disposable baby diapers. *Journal of Cleaner Production*, *210*, 916–928. https://doi.org/10.1016/j.jclepro.2018.11.046

Milieu Centraal. (n.d.). *Luiers: Wegwerp of wasbaar*. Milieu Centraal. https://www.milieucentraal.nl/bewust-winkelen/uitgelichte-producten/luiers/

Miller, G. T., & Spoolman, S. (2021). *Living in the environment* (20E (Twentieth edition)). Cengage Learning.

OVAM. (2017). *Annex. Monetisation of the MMG method. Update 2017*. www.vlaanderen.be. https://www.vlaanderen.be/publicaties/annex-monetisation-of-the-mmg-method-update-2017

Pampers. (2023). *Pampers-producten, informatie over babyverzorging en ouderschap | Pampers*. Pampers-NL-NL. https://www.pampers.nl

Perman, R., Ma, Y., Maddison, D., & McGilvray, J. (2011). *Natural resource and environmental economics* (4th ed.). Pearson Education.

Pizzol, M., Weidema, B., Brandão, M., & Osset, P. (2015). Monetary valuation in Life Cycle Assessment: A review. *Journal of Cleaner Production*, *86*, 170–179. https://doi.org/10.1016/j.jclepro.2014.08.007

Ricke, K., Drouet, L., Caldeira, K., & Tavoni, M. (2018). Country-level social cost of carbon. *Nature Climate Change*, *8*(10), Article 10. https://doi.org/10.1038/s41558-018-0282-y

Rosenberger, R. S., & Loomis, J. B. (2003). Benefit Transfer. In P. A. Champ, K. J. Boyle, & T. C. Brown (Eds.), *A Primer on Nonmarket Valuation* (pp. 445–482). Springer Netherlands. https://doi.org/10.1007/978-94-007-0826-6_12

Schneider-Marin, P., & Lang, W. (2020). Environmental costs of buildings: Monetary valuation of ecological indicators for the building industry. *The International Journal* *of Life Cycle Assessment*, *25*(9), 1637–1659. https://doi.org/10.1007/s11367-020-01784-y

Shabman, L., & Stephenson, K. (2000). Environmental Valuation and Its Economic Critics. *Journal of Water Resources Planning and Management*, *126*(6), 382–388. https://doi.org/10.1061/(ASCE)0733-9496(2000)126:6(382)

Souza, R. G., Rosenhead, J., Salhofer, S. P., Valle, R. A. B., & Lins, M. P. E. (2015). Definition of sustainability impact categories based on stakeholder perspectives. *Journal of Cleaner Production*, *105*, 41–51. https://doi.org/10.1016/j.jclepro.2014.09.051

Statista. (2021). *Netherlands: Number of live births 2021*. Statista. https://www.statista.com/statistics/519994/total-number-of-live-births-in-the-netherlands/

Steen, B. (2016). Calculation of Monetary Values of Environmental Impacts from Emissions and Resource Use The Case of Using the EPS 2015d Impact Assessment Method. *Journal of Sustainable Development*, *9*(6), Article 6. https://doi.org/10.5539/jsd.v9n6p15

True Price. (2021). *Monetisation Factors for True Pricing*. https://trueprice.org/wp-content/uploads/2023/01/211108_MonetisationFactorsForTruePricing_v2_0_3.pdf

United Nations. (2021). *Single-use nappies and their alternatives: Recommendations from Life Cycle Assessments - Life Cycle Initiative*. https://www.lifecycleinitiative.org/library/single-use-nappies-and-their-alternatives/, https://www.lifecycleinitiative.org/library/single-use-nappies-and-their-alternatives/

United Nations Framework Convention on Climate Change. (n.d.). *Key aspects of the Paris Agreement | UNFCCC*. Retrieved May 14, 2023, from https://unfccc.int/most-requested/key-aspects-of-the-paris-agreement

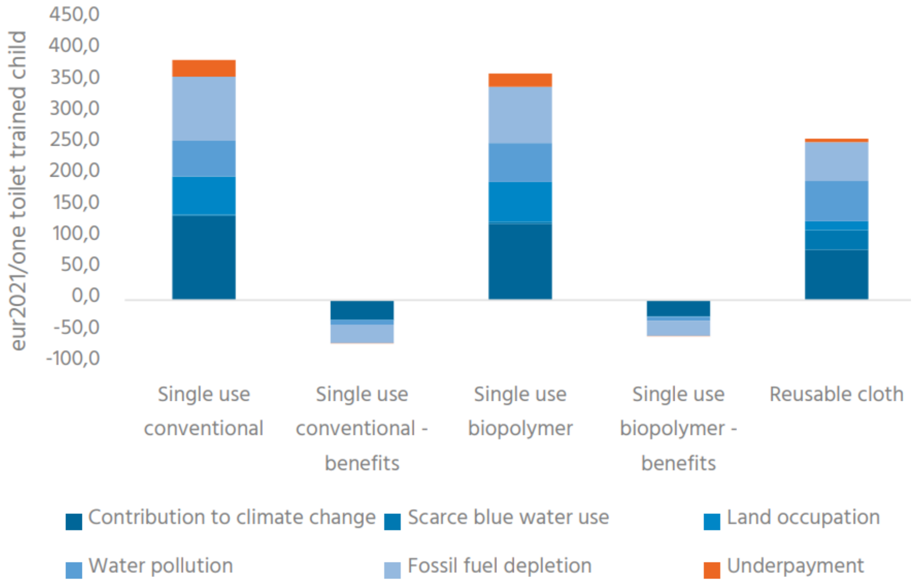
US EPA, O. (2017). *The Social Cost of Carbon* [Reports and Assessments]. https://19january2017snapshot.epa.gov/climatechange/social-cost-carbon

Wang, P., Deng, X., Zhou, H., & Yu, S. (2019). Estimates of the social cost of carbon: A review based on meta-analysis. *Journal of Cleaner Production*, *209*, 1494–1507. https://doi.org/10.1016/j.jclepro.2018.11.058

Wenzel, T. (2014). Consumer myopia, competition and the incentives to unshroud add-on information. *Journal of Economic Behavior & Organization*, *98*, 89–96. https://doi.org/10.1016/j.jebo.2013.12.002

WWF Germany. (2023). *WWF Water Risk Filter—Country Profiles*. https://riskfilter.org/water/explore/countryprofiles

Appendix



Appendix 1: True cost of the disposable (single use conventional diaper) and cloth diaper (reusable baseline diaper) by the TP (Future Diaper Project, 2022). The following data from this figure is irrelevant and is not used for this research: underpayment, single use conventional benefits, single use biopolymer and single use biopolymer benefits.