Evaluation of the sustainability of controlling diffuse water pollution in urban areas on a life cycle basis

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By

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ADP elements: Abiotic Depletion Potential (elements)
ADP fossil: Abiotic Depletion Potential (fossil fuels)
AP Acidification Potential
BMPs: Best Management Practice
BOD: Biochemical oxygen demand
CNA: Comision Nacional del Agua
COD: Chemical oxygen demand
CSOs: Combined sewer overflows
DOC: Dissolved organic compounds
EA: Environment Agency (England and Wales)
EMC: Event mean concentration
EP: Eutrophication Potential
EPA: Environmental Protection Agency (USA)
FAETP: Freshwater Aquatic Ecotoxicity Potential
FIOs: Faecal indicator organisms
FU: Functional Unit
GWP: Global Warming Potential
HTP: Human Toxicity Potential
ILCD: International Life Cycle Database
LCIA: Life Cycle Impact Assessment
MAETP: Marine Aquatic Ecotoxicity Potential
MPN:Most probable number
MRC: Magdalena river Catchment
NPV: Net present value
OFMSW: Organic fraction municipal solid waste
ODP: Ozone Layer Depletion Potential
PAHs: Polycyclic aromatic hydrocarbons
POC: Particulate organic carbon
POCP: Photochemical Ozone Creation Potential
PVC: Polyvinyl chloride
SETAC: Society for environmental toxicology and chemistry
sLCA: Social life cycle assessment
SPMV: Sustainability program of the Mexico City Valley
SSI: Sites of scientific interest
SSOs: Separate sewer overflows
TETP: Terrestrial Ecotoxicity Potential
TN: Total Nitrogen
TP: Total phosphorus
TSS: Total suspended solids
UHI: Urban heat island

Abstract

Diffuse water pollution in urban areas is growing due to polluted runoffs. Therefore, there is a need to treat this kind of pollution. Different structural treatment practices can be used for these purposes. However, little is known about their environmental, economic and social impacts. Therefore, the aim of this study has been to develop an integrated methodology for sustainability evaluation of structural treatment practices, considering environmental, economic and social aspects. Both environmental and economic evaluations have been carried out on a life cycle basis, using life cycle assessment and life cycle costing, respectively. For social evaluation, a number of social indicators, identified and developed in this research, have been used.

The methodology has been applied to the case of the Magdalena river catchment in Mexico City. Three structural treatment practices have been analysed: bio-retention unit, infiltration trench and porous pavement.

Based on the assumptions and the results from this work, the bio-retention unit appears to be environmentally the most sustainable option for treatment of diffuse water pollution. It is also the second-best option for social sustainability, slightly behind the porous pavement. However, if the costs of treatment are the priority, then the porous pavement would be the cheapest option. If all the sustainability aspects evaluated here are considered of equal importance, then the bio-retention unit is the most sustainable option.

Therefore, trade-offs between the different sustainability aspects are important and should be considered carefully before any decisions are made on diffuse water pollution treatment. This also includes the trade-offs with the additional life cycle impacts generated by the treatment options compared to the impacts from the untreated runoff. The decisions can only be made by the appropriate stakeholders; however, some recommendations are given, based on the outcomes of this research.

Declaration

No portion of the work referred to in this thesis has been submitted in support of an application for another degree or qualification of this or any university or other institute of learning.

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1 Introduction

Urban areas are growing, but their capacity to meet the needs of their inhabitants is not following at the same pace (UN, 2011). Among these, clean water supply is a basic human need and a requirement for the sustainable development of cities (UNEP, 2012).However, in many urban areas water is polluted and needs to be treated before it is fit for consumption. Thus far, much of the water treatment has been focused on pollution from point sources, with little attention being given to diffuse pollution, which is increasingly affecting water quality in urban areas.

Unlike water pollution coming from point sources, which has both defined origin and point of discharge, diffuse water pollution comes from multiple sources and does not have a defined point of discharge (DEFRA, November 2012). Although there is no consensus in regard to the sources considered as diffuse, in the United Kingdom D'Arcy et al., (2000 cited in Ellis & Revitt, 2008) offer a clear definition of these sources and of diffuse pollution itself: "pollution arising from land use activities (urban and rural) that are dispersed across a catchment, or sub catchment and do not arise as a process industrial effluent, municipal sewage effluent, deep mine or farm effluent discharge". During rainfall events pollutants coming from these activities are washed off from the surface where they have built up, finding their way to both surface and groundwater sources (DEFRA, 2004; Trauth & Xanthopoulos, 1997).In urban areas this pollutant transport process is emphasised due to paved surfaces (Ellis & Revitt, 2008), which increases runoff generation about five times in comparison with natural ground cover (EPA, 2003). Additionally to paved areas, activities inherent to the urban environment also increase the generation of diffuse pollutant sources; according to Novotny (2003) these activities encompass car wear (e.g. tyre and brake wear, exhaust emissions, oil leaks), faecal matter of urban animals, wrong connections and leaves falling from the trees.

Although tangible, the potential damage coming from diffuse pollutant sources in the urban context might be underestimated (Ellis & Revitt, 2008).This is due to the high cost and complexity of monitoring these pollutants (Novotny, 2003). Most of the available information on the effects of diffuse pollution comes from research in the developed world, where it has been identified as responsible for: 31% of seriously polluted rivers in Scotland (Ellis & Revitt, 2008), water quality detriment of at least 1000 water bodies in England (DEFRA, November 2012) and risk of illness caused from contact with polluted water (EPA, 1994 cited in Gaffield et al., 2003). It is also in the developed world that guidelines for controlling diffuse water pollution have been developed.

Due to the diffuse nature of the pollutant sources the corresponding treatment is not as straightforward as the installation of wastewater treatment plants to control point water pollution, but encompasses the installation of either structural or non- structural treatment practices, and sometimes a combination of both (Novotny, 2008). Although there is no standard definition of structural treatment practices, according to EPA (1999), they are defined as: constructed installations which aim to provide water quality/quantity control. Some examples of this kind of treatment practice are: infiltration trenches, bio-retention units, porous pavements, street trees or sand filters. On the other hand, non-structural treatment practices have been defined as a set of educational/regulatory controls designed to help in controlling diffuse water pollution (Novotny, 2003).In regard to regulatory controls, they are being developed and implemented mainly in the European Union (EU) and the United States (US); in this context the Water Framework Directive applies to the EU region (EA, 2012), while the Clean Water Act applies in the U.S. (through The National Pollutant Discharge Elimination system (phase I and II)) (Ellis & Revitt, 2008).As to the educational programs, they vary from place to place which makes them difficult to implement (Novotny, 2003). In spite of this difficulty, the effort might be worthwhile if they help people to understand that each individual contribution makes a real impact in dealing with diffuse water pollution.

The selection process for the most appropriate treatment practice is determined by the guidelines provided in the applicable regulation, if this is available for the analysed area. It is mainly based on the downstream effects of the treatment practices, giving little attention to the whole life cycle impacts (Kirk, 2006). However, information about these impacts is required for different stakeholders, who require information about the long term impacts not only from the environmental but also from the economic and social point of view. In this regard life cycle assessment plays an important role, however its implementation is a growing area with few practical examples. This is mainly due to the lack of life cycle data. The results of the literature review in regard to the application of LCA in the analysis of the environmental impacts of structural treatment practices shows that it has been used for the analysis of the greenhouse gas emissions and energy requirements of porous pavement and street trees in a high density urban area of New York (Spatari et al., 2010).The results from this study show that these treatments may be a potential option to reduce both greenhouse gas and energy from the municipal control pollution water facilities. Moreover, Andrew & Vesely (2008) have used LCA to analyze the CO_2 emissions and life cycle energy of a rain garden and a sand filter, identifying the rain garden as the best option with 20% less CO_2 emissions and 30% less life cycle energy requirements than the sand filter. Other studies such as Kirk, (2006) have also analyzed environmental impacts of four treatment practices[1]: advanced drainage system (ADS) subsurface water quality and infiltration device, bio-retention cell, gravel wetland and wet pond. The results from this analysis show that in regard to the life cycle environmental impacts wetland is the best option. Considering global warming potential as a basis for comparison, bio-retention cell is the best option with 1620 kg CO_2 eq./functional unit (fu)(management and treatment of runoff coming from 0.4 ha of parking lot), having 15%, 17% and 4% less CO_2 emissions than the ADS subsurface water quality and infiltration unit, wet pond and gravel wetland respectively.

[1] See (Kirk, 2006) for a complete definition of each treatment practice.

In addition to the information related to the life cycle environmental impacts, designers and storm water managers also require life cycle costing information. Unlike capital cost, this information would help in order not to underestimate hidden costs related to the maintenance and operation along the life cycle of the treatment option (Lampe et al., 2005). Information about life cycle cost (LCC) is scarce, being the main challenge identified by Lampe et al. (2005) when life cycle cost estimation is conducted. In addition to the economic results this author also summarizes the challenges encountered in order to estimate the LCC of structural treatment practices, such as the difficulty of collecting data due to the preliminary stage of the runoff treatment as an industry, as well as the fact that available information belongs either to private individuals or municipalities who consider this information private.

Controlling diffuse water pollution leads not only to economic and environmental impacts, but also to social impacts. However, even when these impacts may be the most important area of sustainability to be evaluated, there is no information about the social impacts along the life cycle of structural treatment practices.

1.1 Aim of the study

The aim of this study is to help to understand the life cycle environmental, economic and social impacts of controlling diffuse water pollution. In order to achieve this goal the following individual objectives have been established:

- Development of a life cycle model for each of the considered treatment options to assess their environmental and economic sustainability;
- Implementation of life cycle assessment methodology for the environmental assessment of the selected treatment options;
- Implementation of environmental life cycle costing for conducting the economic evaluation;
- Identification of social impacts associated with the considered treatment practices; and

- Identification of most sustainable option(s) and recommendations to relevant stakeholders, including local authorities and policy makers.

The novelty of this study is the implementation of a life cycle approach for the evaluation of environmental, economic and social impacts of controlling diffuse water pollution using structural treatment practices in the urban environment.

1.2 Justification

This study focuses only on the analysis of structural treatment practices, specifically considering three of them: bio-retention units, porous pavements and infiltration trenches. It has been considered that these treatment practices are installed in the area of the Magdalena River Catchment (MRC) in Mexico City. This city has been selected because little is known about controlling diffuse water pollution in urban areas in the developing world. Although the National Water Commission (NWC, 2013) has started to consider the importance of controlling this kind of pollution only in rural areas, the analysis of the case of urban areas such as Mexico City might be considered in the near future. Like many cities in the developing world, Mexico City faces the challenge of treating waste water, considering not just point sources but also diffuse pollutant sources.

This leads to an explanation of the selection of these structural treatment practices. Since in Mexico City diffuse pollution is not yet treated, there is neither regulation nor guidance for the selection process. Therefore, this information has been sourced from the Maryland guide (MDE, 2000), which as explained in Chapter 3 has been selected based on the similarities in the annual rainfall with Mexico City, specifically in the Magdalena River Catchment. The MDE, 2000 guideline establishes the following criteria for selecting the treatment practices: watershed and terrain factors, storm water treatment suitability, physical feasibility, community and environmental factors, location and permitting factors and space constraints (MDE, 2000). However, according to SMA (2011), this information is not available in the case of Mexico City. Hence, the criterion for selecting these treatment practices is the space constraint, which is considered as a key element not only for Mexico City but also for other metropolises. The selection has been narrowed down to structural practices, since according to SMA (2011), there is no information available to conduct the analysis of non-structural treatment practices.

1.3 Dissertation content

In order to set the scene, a description of the concept of diffuse water pollution is given in Chapter 2, together with an overview of diffuse pollution sources and structural and non-structural treatment options. Chapter 3 describes the integration of LCA, LCC and social impacts for the sustainability evaluation of structural treatment practices. This is followed in Chapter 4 with the description of the case study used in this work (Magdalena River Catchment) as well as a general description of the information required for the design of the selected treatment options. The results of both the environmental and economic sustainability assessments of the bio-retention unit, the infiltration trench and the porous pavement are provided in Chapters 5, 6 and 7, respectively. The analysis of these results, as well as the discussion of the social impacts associated with these treatment practices is given in Chapter 8. Finally, conclusions and recommendations for future work are detailed in Chapter 9.

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2 Diffuse water pollution and treatment practices

This chapter establishes the concept of diffuse water pollution, describing both pollutant sources and treatment options to control this kind of pollution. Moreover, it also illustrates the problem of controlling diffuse water pollution in urban areas, to this end a summary of the literature review is also shown in this chapter.

2.1 Diffuse water pollution

Water can be polluted from two sources: point and diffuse (Novotny & Olem, 1994; Tsihrintzis & Hamid, 1997). The difference between these pollutant sources is their origin; while the former has a defined origin such as sewage discharge, the latter has been defined as "pollution arising from land-use activities (urban and rural) that is dispersed across a catchment or sub-catchment, and does not arise as a process effluent, municipal sewage effluent or farm effluent discharge" (Darcy et al., 2000, cited in Ellis & Revitt, 2008). In addition to their origin Ice (2004) has identified other differences between point and diffuse sources, which are described in Table 1.

Table 1 Characteristics of point and non-point sources of pollution (Ice, 2004)

Point sources	Nonpoint – diffuse sources-
End of pipe, easier to identify source	Not well defined, diffuse source
Pollutants may be product of manufacturing	Pollutants usually natural (e.g. sediment) and essential to stream at some level
Loads far in excess of natural loads	Loads relatively low from any single source
Stationary sources, easier to set up representative monitoring sites	Sources along the catchment area, difficult to determine representative monitoring sites.
Pollution discharge may be less closely tied to weather and hydrology	Pollution discharge strongly influenced by weather and hydrology
Pollution controlled using process controls or effluent treatment under established control parameters	Pollution controlled with treatment practices through voluntary incentive, or regulatory nonpoint source control programs

Although diffuse pollutants from agricultural sources have received more attention in the past (Panda & Behera, 2003), the constantly growing city has increased the contribution from urban sources to diffuse water pollution. As runoff travels from the urban surface, where runoff is increased about five times in comparison with natural ground cover (see Table 2), it washes out pollutants that have built up on this surface. The amount of pollutant (known as pollutant load) transported through the urban runoff is determined by a complex relationship among different factors, which according to Hewett et al. (2009) include soil type, climate conditions, hydrology, topography, land use and regulation of land use activities. The amount of pollutant load is estimated through different models that relate these factors to each other (Marsalek, 1991).

Table 2 Effects of imperviousness on runoff (Chester & Gibbons, 1996)

Imperviousness (%)	Runoff (%)	Shallow infiltration (%)	Deep Infiltration (%)	Evapotranspiration (%)
0 (Natural ground cover)	10	25	25	40
10-20	20	21	21	38
30-50	30	20	15	35
75-100	55	10	5	30

Although the effect created by different pollutant loads is still under analysis, mainly in areas of the developed world (EA, 2007; EPA, 1999a), an increasing effort to standardize diffuse pollutants and identify the activities that produce these kinds of pollution has been carried out as explained in next section.

2.2 Diffuse water pollution sources

Diffuse pollution sources can be categorised as agricultural and urban sources. The former includes pollution due to the application of fertilisers, pesticides and livestock operations, but excludes farm effluent discharge (Novotny & Olem, 1994;EA, 2007). A brief description of both agricultural pollutant sources and their effects is provided in section 2.2.1; however, since the aim of this work is to analyse the effect of diffuse water pollution in the urban context, more emphasis has been given to urban pollutant sources, which as shown in section 2.2.2 include roads, commercial activities, as well as pollution from sewer overflows and untreated human excreta.

2.2.1 Agricultural sources of diffuse water pollution

Agricultural land use is considered as the main source of diffuse water pollution (Collins & McGonigle, 2008) because it occupies the majority of the land area that reaches river catchments (Whiters et al., 2001). Agricultural activities around the catchment that have been found to be the cause of these diffuse pollution sources are: use of fertilizers and pesticides (Ribbe et al., 2008) livestock operations (Kurz et al., 2006) and the implementation of new irrigation technologies (Collins & McGonigle, 2008; Chen et al., 2009; Withers & Lord, 2002).

According to Mainstone & Stewart (2008a), both phosphorus and nitrogen have been identified as the main diffuse pollutants of concern from agricultural activities. An excess of nutrient content in water bodies leads to eutrophication, which impairs water quality either for supply use or for recreational activities, (Novotny, 2005). According to the Organization for Economic Co-operation and Development (OECD), 65% of the lakes in the world are eutrophic, from pollution presumably coming from diffuse sources (Mitchell, 2005). In the United States (U.S.) diffuse pollutant sources from agricultural activities have led to anoxia in 53% of the estuaries in the country (Novotny, 2005). In addition to environmental cost, diffuse pollution from nutrients also represents an economic cost. For example, in the United Kingdom (UK), annual treatment of drinking water for pesticides costs £120 million, for phosphate and soils about £55 million, for nitrates around £16 million and for microorganisms £23 million (Pretty et al., 2000).

In addition to nutrients, sediment is another form of diffuse pollution that results from polluted agricultural runoff. Eroded land is the main source of fine sediment in American reservoirs and aquatics systems (Pimentel et al., 1995). An excessive amount of fine sediment causes an accumulation of sediment in the river beds, which is known as siltation (Greig et al., 2005). Siltation also has a detrimental effect on the physical properties of water bodies, since it causes less light penetration, an increase in turbidity and a decrease in the fish population (Mainstone et al., 2008b). Silt also leads to a lack of stability in the river bed, which in turn loosens the roots of the submerged plants and decreases oxygen diffusion (Greig et al., 2005). One species that has been highly affected by the accumulation of silt is salmon, because silt causes a decrease in the rate of oxygen transport from the water to the egg membrane (Greig et al., 2005). Another impact of silt accumulation is the reduction in energy production by hydropower plants. This is due to fine sediment accumulated in dams and reservoirs which decreases water storage capacity. The dam located in Aba-Samyuel, Ethiopia is an example of this effect, whose useful lifetime could be reduced from 70 to 24 years due to silt accumulation (Pimentel et al., 1995).

2.2.2 Urban sources of diffuse pollution

2.2.2.1 Pollution from urban motorways and roads

According to Kayhanian et al. (2007), average daily traffic is higher in urban highways than in rural ones, becoming a source of diffuse water pollution in urban hubs. Among the pollutants coming from urban highways are heavy metals (Sorme & Lagerkvist, 2002) (see Table 3), oil and grease (from leaks or improperly discarded use) and debris material (from construction or erosion) (Nixon & Saphores, 2007). In regard to heavy metals, Hulskotte et al. (2006) found that brake wear from road traffic vehicles amounts to 2.4 ktonnes per year of copper emissions, which could become a diffuse pollutant source. In addition to copper, another metal coming from road traffic is zinc, which comes from tyre wear (Davis et al., 2001, cited in Nixon & Saphores, 2007).

Another pollutant source from urban highways is oil, which in the UK, according to Ellis & Chatfield (2006), contributes to about 24,000 tonnes of oil per year transported through urban runoff. Although it is hard to track the origin of the oil found on the highway surfaces, four ways have been identified by Bohemen & Janssen Van De Laak (2003): lack of maintenance – or absence - of oil separators, spillages from tanks or from the delivery of overfilled storage tanks, leakage from storage tanks and illegal discharge.

Further to heavy metals and oil, another pollutant source from highways is debris material (Kay & Falconer, 2008). For example aluminium, mercury, and benzothiazole (an organic compound released from crumb rubber to roads) have been identified as significant pollutants arising due to highway construction activities (Gaffield et al., 2003).

Table 3 Heavy metal sources in highways (Nixon & Saphores, 2007)

Source	Cd	Co	Cr	Cu	Fe	Mn	Ni	Pb	Zn
Gasoline	*			*				*	*
Exhaust							*	*	*
Motor oil and grease		*			*		*	*	*
Antifreeze	*		*	*	*			*	*
Undercoating								*	*
Brake				*	*		*	*	*
Rubber	*							*	*
Asphalt							*		*
Concrete									
Diesel oil	*								
Engine wear				*	*	*	*	*	*

Finally, another pollutant coming from urban highways is polycyclic aromatic hydrocarbons (PAHs), which according to Napier et al., (2008) come as a result of incomplete combustion processes, oil leaks and tyre and brake wear.

2.2.2.2 Faecal matter on urban streets

Faecal material from urban animals is considered a major source of bacterial pollution in urban runoff due to the growing number of domestic and wild animals. O'Keefe et al., (2003) have found that dogs, cats, pigeons and rats are the main source of faecal matter in the urban environment. The densities of these animals in the urban environment are shown in Table 4. During rainfall events, faecal matter is transported from the urban surface to the receiving water bodies, which is why it is considered a potential health risk. For example, Dunk et al., (2008) found that faecal matter is a source of pathogens that can cause gastrointestinal illnesses.

Faecal Indicator Organisms (FIOs) from domestic and wild animals are used to monitor the presence of faecal material in urban water bodies (Ellis, 2004) and are used to establish water quality parameters for European and U.S. legislation. The most representative FIOs are coliforms, faecal coliforms and faecal streptococci (Kim et al., 2007). Within these *E. Coli* in surface water has been found to be a predictor of gastrointestinal illness. In Cornwall, southwest England, an outbreak of *E. Coli* was found to be the cause of gastrointestinal illness for people who were in contact with polluted water from cattle which was then spread by surface runoff (EPA, 2001; Novotny, 2003).

Table 4 Sources of FIOS in the urban environment (O'Keefe et al., 2003)

Non-human sources species	Example urban density	Location
Cats	160 cats/km^2	Bristol
Dogs	260 dogs/km^2 1 dog/10 people	Dunfermline UK and urban U.S.
Pigeons	10-250/flock	at least 200000 breeding pairs in UK
Rats	Similar to human population	60M rats in UK

The contribution of dog fouling to bacteriological load is significant, with a study conducted in North London (EPA, 2001) determining that 6.5-7.4 million dogs accounted for some 1,000 tonnes per day of fouling, and in Melbourne, dog fouling contributed to pollution equivalent to the untreated sewage from 90,000 people (EPA, 2003;Eganhouse & Sherblom, 2001). Additionally, birds also contribute some 25-30% of FIO to urban runoff (Gibson et al., 1998). For example, the Figgate Burn urban catchment in Edinburgh possessed surface water discharge containing some 500,000 faecal coliforms MPN (most probable number)/100 ml (Gaffield et al., 2003). The origin of this pollution was traced to a pigeon roost located beneath a railway. Cats, waterfowls and rats also contribute to bacterial water pollution, with rats representing a major concern since they typically live in storm water systems, which lead to direct faecal material being discharged into the water bodies (Gaffield et al., 2003). The estimations of FIO per gram of faecal matter produced by some urban animals and the contributions of faecal matter to surface drainage system are shown in Table 5 and Table 6, respectively.

Table 5 Faecal indicators per gram of host faeces modified from (O'Keefe et al., 2003)

Source	Faecal coliform (density/gram)	Faecal streptococcus (density/gram)
Cats	$7.9x10^6$	$2.7x10^7$
Dogs	$2.3x10^7$	$9.9x10^8$
Rats	$1.6x10^5$	$4.6x10^7$
Ducks	$3.3x10^7$	$5.4x10^7$

Table 6 Bacteriological load surface drainage from (Ellis, 2004)

Source	FC (faecal coliforms)	FS (faecal streptococcus)
Dogs	$2.3x10^7$/100 ml	$9.8x10^8$/100 ml
Pigeons	$0.5x10^{6/}$100 ml	-------------------
Waterfowl	$3.3x10^7$/g host faeces	$5.4x10^7$/g host faeces

2.2.2.3 Sewers overflows

A separate drainage system is divided into two parts: one that conveys storm water from roads and roofs (surface drainage) and another that conveys waste water from household supplies, sinks, wash machines, toilets and dishwashers (foul drainage) (Novotny, 2003). Conversely, in the combined sewer systems all the sewage is conveyed in the same system.

When a sewage system's capacity of wastewater treatment plants has been reached – or exceeded - excess water volumes are sent to surface water bodies (Dunk et al., 2008). Overflows from separate sewer systems are known as separate sewer overflows (SSOs). Since separate sewer systems are designed to convey only surface rainfall, water that comes from these sources is sent without treatment to rivers and streams (Dunk et al., 2008). However, as discussed in the next section, due to "wrong connections" this sometimes contains polluted water that impairs the water quality of the receiving water bodies. Based on water quality analysis conducted in water bodies throughout the U.S. EPA has determined that SSOs impair five designated water uses: aquatic life support, drinking water supply, fish consumption, shellfish harvesting and recreation (EPA, 2001).

In contrast, if overflows occur in combined sewer systems they are known as combined sewers overflows (CSOs) and are considered an important source of polluted water (EA, 2007). Even though they are regulated as a point source of pollution, the proportion of water that is sent to the receiving water bodies without any treatment is considered a diffuse pollutant source because the origin of its pollution remains unknown (Dunk et al., 2008). In the U.S., the EPA has determined that 40,000 SSOs and thousands of CSOs take place every year (EPA, 2001). Untreated sewage water can result in an increase in the FIO content that can result in impairment of water bodies and disease (Wang et al. 2008). Even though contributions from CSO as a non-point source of pollution are poorly understood (EPA, 2001) wastewater derived from these sources has been found to be a source of bacteria, viruses, protozoa (i.e. parasitic protozoa including Giardia, *Cryptosporidium*, and *Entamoeba Giardia*, helminths, roundworms, hookworms, tapeworms, and whipworms), biochemical oxygen demand (BOD), toxic compounds that include heavy metals, hydrocarbons and synthetic organic chemicals, nutrients and suspended solids (Novotny, 2003). During dry periods the concentrations of *Cryptosporidium* oocysts and *Giardia* cysts were 5-105 oocysts/100 l and 13-6,579 cysts/100 l respectively; whilst in wet weather these increased to 250-40,000 oocysts/100 l and 9,000-283,000 cysts/100 l respectively, which was attributed to CSO discharge (Caspar et al., 2009). Polluted water with *Cryptosporidium* and *Giardia* was found to be a cause of 17% to 32% of approximately 99 million people that contracted gastrointestinal illness as a result of contaminated drinking water (Marsalek et al., 2008; Walsh, 2000). In Spain, in order to control the effect of CSO, the analysis of three treatment technologies (vortex, Densadeg clarifier and screening) shows that implementing this control is beneficial, since the offset between the impacts derived from such implementation is positive, due to the eutrophication reduction generated by the CSO treatment (Aldea et al., 2012).

2.2.3 Wrong connections

Wrong connections between foul and surface drainage, transporting wastewater without any treatment from foul drainage into rivers are considered a diffuse pollutant source (Marsalek, 1991). They occur due to washing facilities and toilets wrongly connected to the rainfall drainage system (Mills, 2009). In the Thames region, in southeast England, the proportion of different causes of wrong connections was determined from the analysis of 28,000 outfalls in the region (Dunk et al., 2008); this distribution is shown in Figure 1. Through bacteriological breakdown organic matter contained in polluted water coming from wrong connections leads to ammonia, which has been found to be the main pollutant that affects rivers (Marsalek, 1991). Furthermore, Izurieta-Davila et al., (2002) and Anda et al., (2000) have identified water polluted with phosphorus and nitrogen coming from these wrong connections as the main source of pollution for Cuitzeo Lake and Chapala Lake respectively.

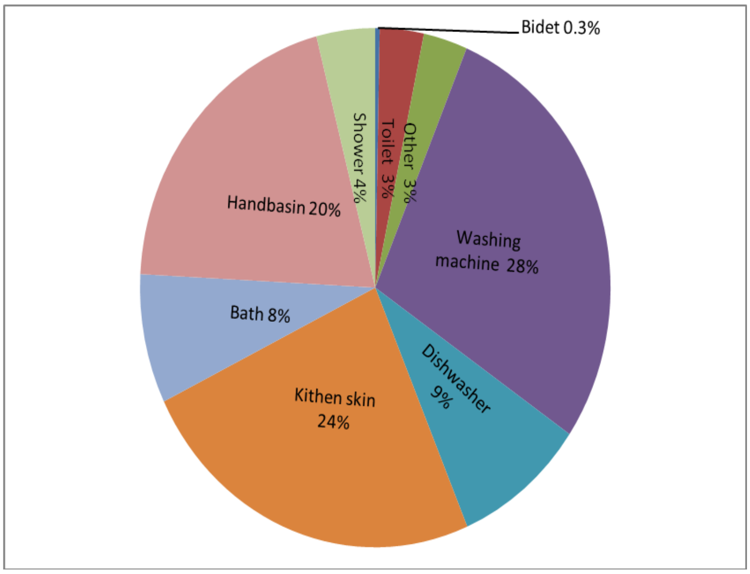

Figure 1 Percentage distribution of wrong connections (Dunk et al., 2008)

The impacts of urban diffuse sources are hard to monitor, since it is an expensive and time consuming activity. However these impacts have been identified on a general basis as described in the section below.

2.2.4 Impacts from diffuse water pollution

As explained above, monitoring of diffuse water pollution is mainly carried out in the U.S. and UK, since in these countries the control of this kind of pollution is regulated. In the U.S, polluted runoff coming from urban highways contribute a sixth of the hydrocarbons and up to half of suspended solids that reach water streams (EPA, 1996). Moreover, due to its trans-boundary transport, diffuse pollution affects water bodies located thousands of kilometres from the original pollution source. Large water bodies such as the Adriatic Sea, the Black Sea, Chesapeake Bay and the Gulf of Mexico are affected by this trans-boundary effect (Novotny, 2005). These water bodies present nutrient pollution coming from farming activities that take place in distant urban centres.

As shown in Table 7, in England and Wales, when compared with point sources of pollution, diffuse pollutant sources generally represent the main cause of water impairment for the receiving water bodies.

Table 7 Comparison of point and diffuse sources as a cause of water bodies impairment (Modified from (Taylor et al., 2005)

Point sources	Diffuse sources
39% rivers	87% rivers
38% lakes	50% lakes
84% estuaries	35% estuaries
24% coastal water	20% coastal waters
3 % groundwater	68% groundwater

According to the Environment Agency, about 50% of the rivers are considered to be at risk of high nutrient concentration from diffuse sources (EA, 2007). In addition to rivers, in England the impact of diffuse pollution can also be seen in areas known as sites of scientific interest (SSI). Around 22,000 hectares of SSI are affected by the effects of diffuse pollution coming from agricultural activities (EA, 2007).

2.3 Control measures

The set of control measures that are designed to reduce diffuse water pollution is known as best management practices (BMPs) in the U.S. (Ice, 2004); and sustainable urban drainage systems in the UK (EA, 2007);

however, in this study they are referred to as treatment practices. There are two kinds of treatment options: non-structural and structural. Non-structural treatment options represent development and implementation of regulations and educational programmes, while structural treatment practices refer to practices that are implemented through the construction of treatment systems. A description of these practices is outlined in this section.

2.3.1 Non-structural treatment practices

Non-structural treatment practices are defined as a set of institutional pollution prevention practices (Bulova & Davis, 1996). Although there is no unique categorization, there are four common categories of non-structural treatment practices (Wong & Taylor, 2002):

i. town planning controls;
ii. pollution prevention procedures;
iii. educational programs; and
iv. regulatory controls.

2.3.1.1 Town planning controls

Town planning controls are plans focused on promoting urban growth in equilibrium with receiving water bodies in the urban environment. Thus, the cornerstone of these plans is that runoff generated by new development does not become a diffuse pollutant source(Novotny, 2005). The application of town planning controls varies in relation to the site characteristics. For example, in Pennsylvania, town control practices include “cluster use of urban areas”, promoting open space availability and decreasing the disturbed area (PSBMP, 2006). Another example of planning control is the use of narrow streets; this practice is aimed at decreasing the impervious area of a residential development. In U.S., for instance the normal width of streets ranges from 10-12 m; town planning laws can reduce this to 4 m. Other examples of town planning controls are open space and green parking designs.

2.3.1.2 Pollution prevention procedures

Pollution prevention procedures include maintenance to avoid diffuse pollution reaching the receiving water bodies (Wong & Taylor, 2002). An example is maintenance of the drainage system to control the amount of sediment during combined sewer overflows. Furthermore, other preventive procedure to control diffuse water pollution are pest control, bridge and roadway maintenance, illegal dumping control and street sweeping, which is described below.

It has been found that the efficiency of street sweeping depends on the size of the particles as well as the frequency of cleaning. For example, heavy metals and toxic compounds tend to be adsorbed on the smaller particles (0.125-0.5 mm) (German & Svensson, 2002). However, most street sweepers are designed for removal of large particles; hence they are ineffective in removing toxic substances. Therefore, an appropriate design of the sweepers brush is essential for removal of fine particles (Vanegas-Useche et al., 2010), an example of a common mechanical sweeper is shown in Figure 2.

Figure 2 A mechanical brush street sweeper (German & Svensson, 2002)

In addition to brush design, the frequency of sweeping is another parameter to control (Muhammad and Hooke, 2005). For example, it was found that in Portland, U.S. bimonthly street sweeping practice reduces up to 80% of total annual suspended solids (Sutherland & Jelen, 1997). However, a drawback of the use of street sweeping is the high cost, which could reach AUS$1 million per year, based on the results from different field studies in Australia (Walker & Wong, 1999).

2.3.1.3 Educational programs

Educational programs are equally important as an implementation of the source control measures discussed above. They are based on promoting a change in behaviour through economic incentives and information campaigns. Specific areas of education are related to animal waste collection, car washing, landscape and lawn care, pest control and automobile maintenance (SWCM, 2010). Nevertheless, there is no defined pattern in the implementation of educational programs, since they vary in relation to the site characteristics. Implementation of these practices is not expensive; and although it has been found to be an effective measure for pollution control in the United States, New Zealand and Australia (Taylor et al., 2005), there is no standard evaluation parameter that enables a performance evaluation on the same basis (Wong & Taylor, 2002).

2.3.1.4 Regulatory controls

This category includes perhaps the most important non-structural treatment practices. Unlike other non-structural treatment practices, regulatory controls are not voluntary programmes, because they are part of environmental legislation. However, legislation to control diffuse water pollution has been developed mainly in developed countries, including the European Union (EU) and U.S. (Ellis & Revitt, 2008). For example, in the EU, the Water Framework Directive (WFD) addresses diffuse sources of pollution through articles 5 and 11 (WFD as cited in Ellis and Revitt, 2008). Article 5 requires the characterization of diffuse sources by land use activity, while article 11 specifies advancement of permitting systems through the development of programmes of measures. In the U.S., the regulation of diffuse sources of pollution is via the Clean Water Act through The Total Maximum Daily Load program, which specifies the maximum load that a water body can receive without affecting its water quality standards (Novotny, 2003). As mentioned above, another programme by which diffuse water pollution is controlled is the National Discharge Elimination Programme (EPA, 2000). It applies to operators of construction activities that disturb more than 0.4 ha, operators of municipal separate storm sewer systems located in an urbanized area with a density of either at least 1,000 persons per 2.5 square kilometres or a total population of 50,000 inhabitants, and municipalities including cities, towns and villages (Woelkers, 2000).

In developing countries, water quality regulation focuses only on the control of point sources of pollution and diffuse pollution is largely unregulated. Isolated efforts have been made to understand the impact of diffuse sources in urban centres in some developing countries (Campbell et al., 2004), including the Yamuna river in Delhi city (Jamwal et al., 2008), the Reconquista river located in Argentina (Olguin et al., 2004) and in Isfahan, Iran (Taebi & Droste, 2004). Even though these efforts represent a first step in understanding the problem of diffuse pollution in areas of the developing world, it has not yet been translated into official legislation.

2.4 Structural treatment practices

Structural treatment practices are techniques and processes that involve construction and operation of treatment systems (Ice, 2004). These treatment practices include but are not limited to bio-retention units, infiltration trenches, porous pavements grass filters and water ponds, which are discussed below.

2.4.1 Bio-retention units

Bio-retention units are landscaped depressions with a vegetative cover (Davis et al., 2006) used for treating runoff from residential areas, parking lots and traffic islands; (EPA, 1999b; Davis et al., 2006). Examples of bio-retention units in a parking and residential area are shown in Figure 3.

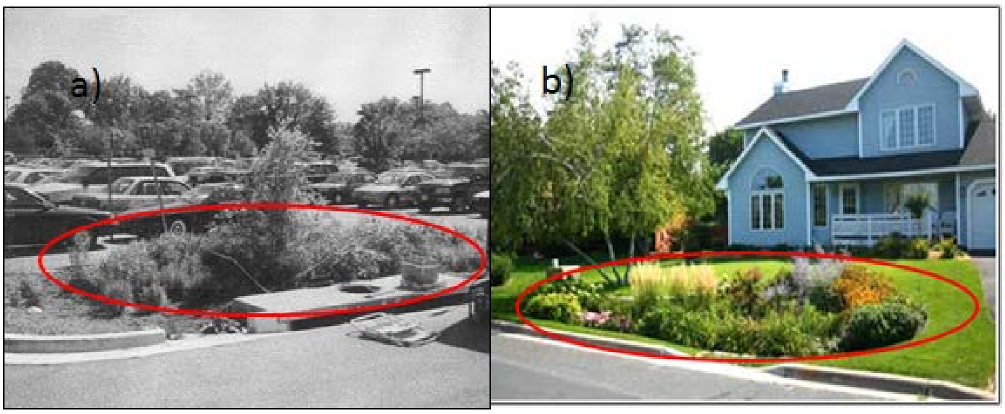

Figure 3 Bio-retention unit: a) in parking area (Hunt et al. 2006), b) in a residential area (Bar, 2006)

As shown in Figure 4 bio-retention units typically consist of two components: pounding area and treatment area. Additionally to the treatment area, filters are used to reduce the sediment load to avoid clogging of the unit (EPA, 1999b). The treatment area is buried in the soil; it is filled with layers of mulch, gravel and soil on which plants grow. These act as an adsorbent, taking up the pollutants and cleaning up the water (EPA, 1999b). The treated water is discharged through a pipe at the bottom of the unit and is sent either to the sewage system or to the groundwater. To control mosquitoes and other insects, the maximum retention time for water is 72 hours (Lampe et al., 2005). During high rainfall events when the treatment volume of the bio-retention unit is exceeded, the runoff is sent to the drainage system through an overflow pipe. Bio-retention units are designed to remove different pollutants, as discussed below.

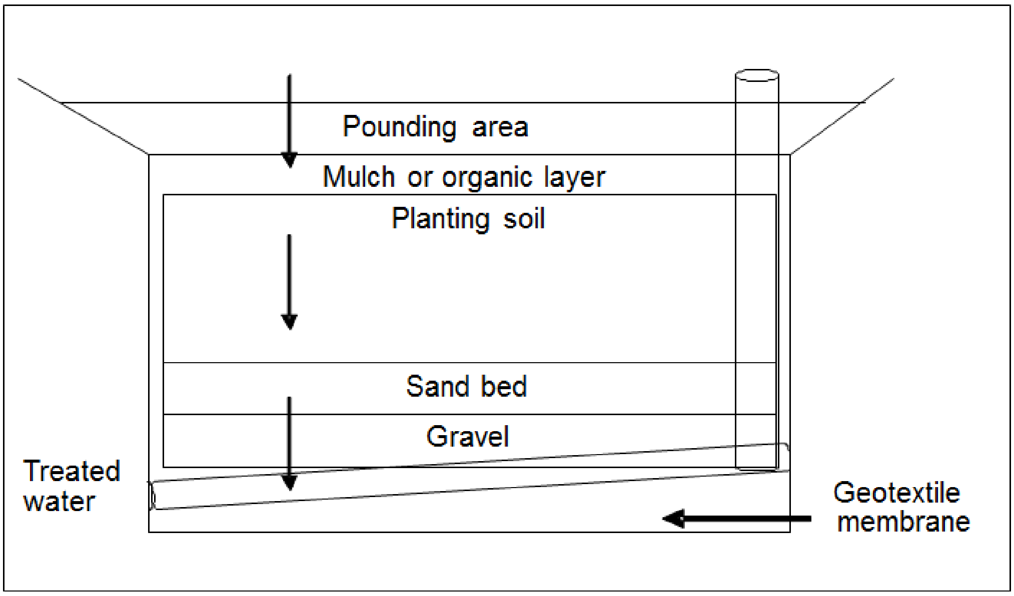

Figure 4 Treatment sequence in the bio-retention unit

Metal removal: This is affected by two design parameters: plant type and soil depth. However, the data on how either of the parameters influences the efficiency of bio-retention units are scarce. The work conducted by Li & Davis, (2008) indicated that metal accumulation in grasses was low, accounting for 0.5-3.3% of the total accumulation rate. Furthermore, Davis et al., (2003) showed that shallow metal accumulation occurred in the first 10 cm of the bio-retention cell, proving that most of the metal accumulation occurred where the organic material was situated. Additionally, according to the analyses of monitoring of field studies, which are shown in Table 8, bio retention units show high metal removal efficiencies. Metal removal efficiencies are around 90%-99%, except for the case of the bio retention located in Largo Maryland, U.S. where metal removal was low due to the kind of material used for the cell, which was made of construction sand and soil. In contrast, in the other cells the material used for the filtration media was fine soil (Davis et al., 2003). Therefore from these analyses it was concluded that the kind of material used in the treatment cell plays a determinant role in metal removal.

Table 8 Bio-retention pollutant removal efficiencies in field and laboratory analysis

Type of analysis	Cu (%)	Zn (%)	Pb (%)	TSS (%)	TN (%)	TP (%)	NO_3N (%)
Laboratory studies							
(Christianson et al., 2004)[a]	91 u 98 l	93 u 97 l	95 u 97 l				
(Kim & Davis, 2003)			>98			47-68	-16[b]
Field studies							
Greenbelt, Maryland (Kim & Davis, 2003)	97	>95	>95				
Largo, Maryland (Kim & Davis, 2003)	43	64	70				
Chapell Hill, North Carolina (Hunt et al., 2006)					40	65	13
Greensboro, North Carolina (Hunt et al., 2006)	99	98	81	-170[c]	40	-240[d]	75

a Samples were taken from the upper (u) and lower (l) ports in Maryland University.
b Negative value means nitrate increase due to the leachate from the soil .
c Negative value means that cell material released TSS.
d Negative value means phosphorus increase due to the leachate from the soil.

Although the bio-retention units can remove heavy metals from the runoffs, metal accumulation is a main concern for the implementation of this treatment practice. In this regard the unique quantitative analysis has been conducted by Davis et al., (2003), as explained in section 5.1.2.2. Another study (Li & Davis, 2008) demonstrated that exposure to lead and other metals accumulated in bio-retention units might represent a health risk to children. However, due to the lack of further studies, there is no conclusive evidence on the long-term effects of the exposure to metals accumulated in bio-retention units.

Nutrient removal: Ammonia is removed by adsorption in the soil (Hsieh et al., 2007). The results found by Davis et al., (1998) show that ammonia can be transformed to nitrate and released from the unit during rainfall events. These released nitrates can be destroyed by bacteria that thrive in the soil (Hunt et al., 2006). Phosphorus is also removed by adsorption on soil (Davis et al., 2006).Phosphorus removal varies in relation to the soil depth and phosphorus content in the soil. The phosphorus removal can be increased by up to 80% for the soil depth between 60 and 80 cm. Phosphorus content in the soil is measured through a P-index, and should be between 10-30 in order to reach around 60% of P removal (NC, 2006).

2.4.2 Infiltration trenches

Infiltration trenches are designed for both runoff treatment and water quantity control (EPA, 1999c). They are appropriate for treating runoffs from residential, commercial and highway land use (Lampe et al., 2005). Due to their design characteristics (large and shallow), they are used where there is limited land available, such as traffic islands (Figure 5). An infiltration trench is the treatment area, and as a separate part of the treatment area filters such as a buffer strip, grass swales or a detention basin are usually used (EPA, 1999c).

Figure 5 Infiltration trench used in a traffic island (LID, 2010)

The treatment area comprises six elements: a geo-textile membrane , sand, soil, stone and a PVC pipe and a rebar anchor (a rebar anchor is a metal component used to fix the PVC pipeline to the soil). The elements of the treatment area are shown in Figure 6.

The geotextile membrane is located around the base and walls of the trench, overlapping the trench walls and the layers of sand and soil (Lampe et al., 2005). It is used to avoid sand and gravel being polluted from the surrounding soil. Another layer (the upper layer in Figure 6) is designed to store the runoff and can be made from stone (EPA, 1999c). When this layer is clogged it can be easily removed to restore full performance of the trench, avoiding the removal of the entire unit. A (polyvinyl chloride) PVC pipe is inserted in the trench to monitor both water quality and volume (EPA, 1999c).The stone layer usually provides around 30%-40% of void space, where the runoff is stored before being treated. The water then goes through the soil and sand layers, where pollutants are removed.

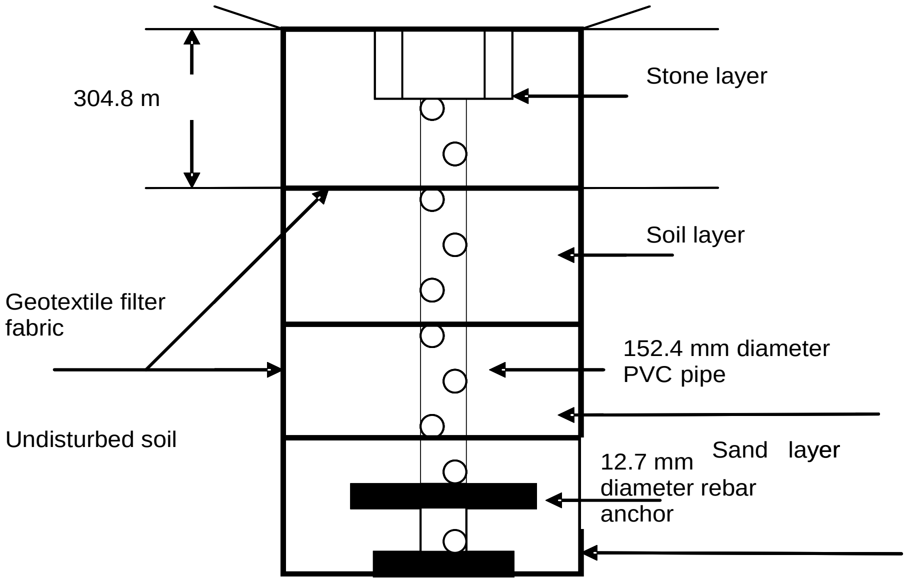

Figure 6 Typical infiltration trench (Southeastern Wisoconsin Regional Planning Commission, 1991 cited in (EPA, 1999c))

Since the mechanisms by which pollutants are removed are not clearly defined, research has been conducted into two aspects of infiltration trenches: heavy metal adsorption and its potential risk to groundwater pollution. With regard to heavy metal capture, it has been found that dissolved metals are adsorbed in the soil, i.e. the treatment layer in the infiltration trench (Sansalone & Buchberger, 1995). However, Murakami et al., (2008) found that Mn, Zn and Cd can be released untreated as free ions, reaching groundwater sources, which could increase pollution instead of controlling it. Hence, the infiltration trench location and design have to be evaluated carefully to avoid the potential risk of groundwater contamination. The removal efficiencies for infiltration trenches presented in Table 9 show high efficiency in total suspended solids (TSS), biochemical oxygen demand (BOD), chemical oxygen demand (COD), dissolved organic carbon (DOC) and heavy metal, ranging from about 89%-93%. In addition, infiltration trenches show high removal efficiencies, up to 100%, for oil and greases, which is why they are used in the treatment of runoff coming from roadways (Sansalone & Buchberger, 1995). On the other hand, the use of soil and stone native to the construction site leads to a variation in both TN and TP removal efficiencies (Lampe et al., 2005), which, as shown in Table 9, ranges from 30% to 84%.

Table 9 Pollutant removal efficiencies for infiltration trenches

Information source	TSS	BOD	COD	DOC	O&G	Pb,Zn,Cd	TN	TP
(Lampe et al. 2005)	NA*	NA	NA	NA	NA	80	30	60
(EPA, 1999b)	90	70-80	NA	NA	NA	90	60	60

* NA: not available

There is no defined life span of the infiltration trenches, and depending on the source, it varies from three or four years (CASQA, 2003) to 100 years of operation (Emerson et al., 2010). This remarkable difference in the infiltration trench life time is attributed to poor maintenance or changes in the original design parameters. For example, a survey of the infiltration trenches in Maryland, U.S., found that 53% were operated differently compared to the original design (Lindsey, 1991).

Maintenance activities play a key role in the long-term performance of infiltration trenches and especially in clogging control. These activities are divided into two categories: corrective and preventive maintenance. Among corrective activities are the removal of the upper layer and the PVC pipeline, while in the preventive activities debris removal and mowing play a determinant role in order to avoid clogging (Lampe et al., 2005). Information about the decommissioning of infiltration trenches is not documented, due to the lack of monitoring campaigns to evaluate the long term performance of this treatment practice (Lampe et al., 2005).

2.4.3 Porous pavements

Porous pavements have pores smaller than 60 µm which makes them porous (Field et al., 1982). Porous pavements are different from permeable pavements in the sense that the latter are made of impermeable materials, but water infiltrates through gaps that have been specifically designed for water infiltration (Myers et al., 2007). On the other hand, porous pavements allow water to flow through the material itself by acting as membranes (Figure 7).

Figure 7 Permeable pavement (left : ((Myers et al., 2007))) Porous pavement (right : (Cahill et al., 2003))

As shown in Figure 8, a typical porous pavement has four layers (Shirke & Shuler, 2009):

- Porous pavement layer: made of asphalt or concrete, this is aimed at providing an area to capture oil and chemical pollutants. Typical infiltration rates in this layer range from 635-762 cm/hour.
- Top filter layer is made of crushed stone; and is used to provide a base for the porous pavement layer and to act as a filter, providing an area to filter out oils and chemical pollutants
- Reservoir layer: This layer is made of gravel; and it stores the infiltrated runoff.
- Bottom filter layer: This layer is the connection between the reservoir layer and the geotextile membrane; a geotextile membrane is a membrane made of non-biodegradable synthetic fibber. This is aimed

at preventing movement of soil particles while allowing water to drain into the soil.

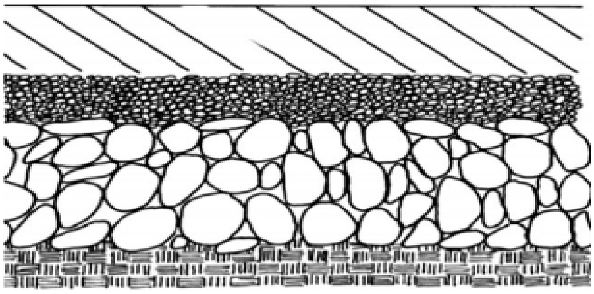

Figure 8 Cross section of porous asphalt pavement (Shirke & Shuler, 2009)

Porous pavements do not require special materials or skills and are therefore easy and inexpensive to install, in car park areas (Sutherland & Jelen, 1997; Cahill et al., 2003; Partland, 2001). Although they have been used for about 20 years (Field et al., 1982), information about their efficiency is scarce. Nevertheless, as seen in Table 10, existing data show that they are an effective measure to control diffuse water pollution, removing on average 70% of pollution.

Table 10 Pollutant removal efficiencies for porous pavements (Cahill et al. 2003)

Parameter	**Removal efficiency**
TSS	91%
TP	66%
TN	72%
TOC	86%
Pb	74%
Zn	81%
Metals	90%
Bacteria	90%
BOD	75%
Cd	33%
Cu	42%
TKN	53%
Nitrate	27%
Ammonia	81%

As shown in Table 11, results found by Dreelin et al., (2006), from a study conducted in the parking lots in Athens, Georgia, show that pollutant concentration from runoff passing through a porous pavement is smaller than passing through asphalt.

Table 11 Runoff composition after passing through porous pavement and asphalt (Dreelin et al., 2006)

Kind of pavement	Ca (mg/l)	Zn (mg/l)	TP (mg/l)	TN (mg/l)
Porous pavement	6.91	0.01	0.41	5.17
Asphalt	8.29	0.05	0.46	2.96

However, the low durability of 20 years of porous pavements is a major drawback (EPA, 1999d) This is mainly due to the clogging (Shirke & Shuler, 2009), which reduces their efficiency and prevents water infiltration (Myers et al., 2007). Regular maintenance can minimize the clogging and improve their longevity (EPA, 1999d). For example, regular sweeping has been found an effective measure to control dust particles that lead to clogging (EPA, 1999d). Nevertheless, since the use of the sweepers can contribute to soil stripping due to the water used to prevent fluidisation of dust, other options have been considered. Among these, Shirke & Shuler (2009) analysed the use of reverse flushing for cleaning of porous pavements. This involves removing particles from the bottom of the porous pavement layer instead of from the top layer. In this experiment water pressure was applied through a bell shaped funnel connected at the bottom of an experimental model, which was done considering the four layers described in Figure 8 . Results showed that water pressure of 21 kPa removes around 80% of the clogging particles (Shirke & Shuler, 2009). Even when this technique might represent a good option for increasing porous porosity, results from this research are not conclusive, since they do not provide information of the possible increase in the porous pavement's lifetime.

2.4.4 Grass filters

Grassed areas can be used to filter polluted runoffs. They retain suspended particles while dissolved pollutants are infiltrated into the soil (Claytor & Shueler, 1996). They are normally sited in side slopes so that the runoffs can be captured before reaching the receiving water bodies (Delectic & Fletcher, 2006). Since little contact time with the grass contributes to their poor performance (Claytor & Shueler, 1996), grass filters should not be located where high runoff volumes are expected.

Grass filters can be categorized into grassed waterways, swales and filter strips (Claytor & Shueler, 1996). Grass swales (see Figure 9) are channels that promote runoff infiltration through dense vegetation which is use to trap the pollutants (Delectic & Fletcher, 2006). A study of the performance of swales in the cities of Taiwan and Virginia found that they could achieve an average reduction of suspended solids of 30-97%, phosphorus of 29-99% and nitrogen of 14-24% (Yu et al., 2001). Grass swales present higher nitrogen removal when compared with these pollutant removal efficiencies for porous pavements, while total suspended solids and phosphate are similar.

Figure 9 Grass swale used in a residential area in Australia (Wong & Taylor, 2002)

On the other hand, filter strips are areas with abundant amounts of grass that act as a bed that can facilitate the infiltration of small drainage areas (Novotny, 2003). They have been found an effective measure in reducing pollution in the runoff. For example, in a U.S. study where filter strips were located between two highways, they reduced 85% of sediments, 68-93% of turbidity, chemical oxygen demand (COD), zinc and iron and 31-61% of phosphorus, total nitrogen, lead and metals, (Barret et al., 1998).

Although pollutant accumulation might be expected after pollutant treatment in grassed swales and filter strips, quantitative information is not documented. This is mainly due to the fact that these treatment options are not considered a primary treatment option, due to the lack of improvement in the pollutant removal.

2.4.5 Water ponds

There are two kinds of ponds for capturing runoffs: a permanent pool of water known as a wet retention pond, and dry ponds that retain water over a period of 24-48 hours (Barret, 2008). Wet ponds are a good option for TSS removal and bacteria, having a removal efficiency of 80% and 70% removal respectively (CWP, 2007). They can also remove 52% of phosphorous and 31% of nitrogen; heavy metal removal can reach 57% for Cu and 64% for Zn (CWP, 2007). Pollutant removal efficiencies are affected by the design parameters, specifically pollutant retention time and wet pond dimensions (Von-Sperling et al., 2003). In this regard, results from a study conducted in Sweden showed that the pollutant removal efficiency of three wet ponds failed owing to incorrect dimensioning (Lundbeg et al., 1999). The implementation of complementary equipment has been found a useful way of increasing the performance of wet ponds. Another factor that also affects pollutant removal efficiency in wet ponds is the accumulation of sediment, since pollutants are accumulated in settled sediments. Among metals found in settled sediments are copper and zinc (Yousef et al., 1996). For this reason sediment removal is required as part of the maintenance activities. Sediment removal every 25 years is suggested by Yousef et al., (1996) in order to maintain the performance of ponds.

In relation to dry ponds, polluted runoff treatment exhibited a reduction of 71% for TSS, 45% for particulate organic carbon (POC) and particulate nitrogen (PN), 33% for particulate phosphorus (PP), 26-55% for heavy metals, and loads from dissolved pollutant did not show any change (Barret, 2008). In regard to heavy metal removal efficiency, results from a dry pond implemented in Surrey were low, accounting for 7% (Hares & Ward, 1999). However, as mentioned above, an increase in heavy metal removal was achieved due to complementary equipment implementation. The silt settlement implementation led to 52% total heavy metal burden removal.

In an evaluation of a dry pond used as a control from runoff coming from highways, it was found that oil and grit traps removal efficiency was 21% and from grit settlement was 52% while the pond efficiency was only 7% (Hares & Ward, 1999). For both dry and wet ponds sediment removal represents an ecological concern due to the pollutant content in the sediment, however, results from sediment analysis specifically for heavy metal, from a wet pond located in Ontario U.S., showed that sediment can be considered as non-hazardous waste. Results for Cd are 1.3 mg/kg, for Cr 102 mg/kg, Pb 122 mg/kg and Ni 9 mg/kg, which were compared with the EU waste management directive values cited in Lampe et al., (2005). Values presented in this directive are: Cd 2500 mg/kg, Cr 2500 mg/kg Pb 2500 mg/kg and Ni 100 mg/kg.

2.5 Summary

There are many different options for managing and controlling diffuse water pollution. These include both preventative and treatment practices. However, it is still not clear how sustainable they are on a life cycle basis from the economic, environmental and social points of view. This work focuses on structural practices and aims to apply the life cycle assessment methodology for their economic and environmental evaluation, evaluating the social aspects on a qualitative basis. This is discussed in the next chapter.

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3 Sustainability assessment

This chapter presents the methodology applied for evaluating the sustainability of structural options for treatment of diffuse water pollution. The methodology is based on a life cycle approach and considers economic and environmental impacts. In addition, social aspects have also been considered and included in the analysis of the treatment options but these considerations are not on a life cycle basis but instead related to direct social impacts of the treatment options. Ranking of the options on the considered sustainability criteria is also part of the methodology to help identify the most sustainable practice and provide recommendations. The steps are outlined in Figure 10 and discussed below.

3.1 Definition of the goal of the study

This step identifies the goal of the study and the service provided by the treatment practice; the latter will define the method used for the design of such treatment practice. The goal of this study is to evaluate the sustainability of different structural treatment options for controlling diffuse water pollution. The service provided by these treatment options is improving runoff quality by reducing the pollutant load in the treated runoff.

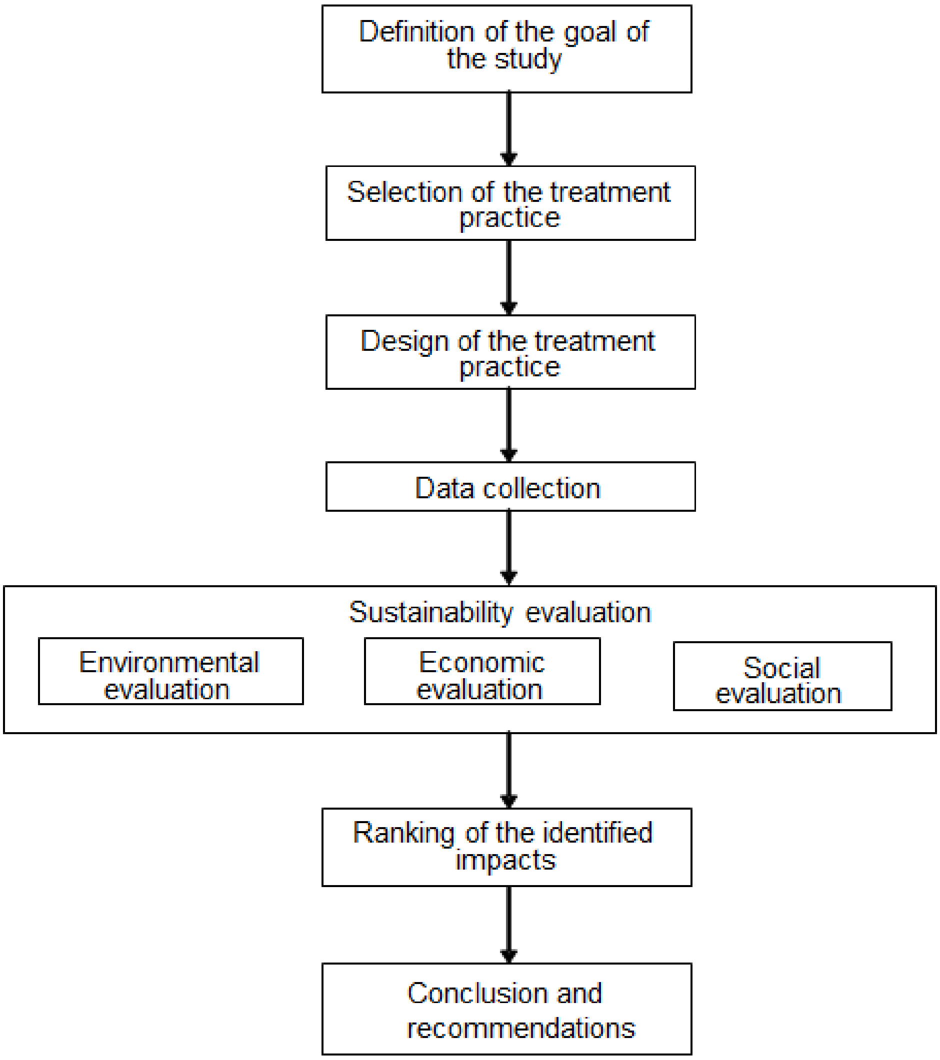

Figure 10 Steps for the sustainability evaluation of options for controlling diffuse water pollution

3.2 Selection of the structural treatment practice

Different structural treatment practices can be used to meet the above-defined service. Selection of the most appropriate option will depend on many different factors. If treatment of diffuse pollution is subject to regulation, then there are defined criteria to be followed during the selection process of the structural treatment practice, such as watershed and terrain factors, storm water treatment suitability, physical feasibility factors, community and environmental factors, as well as location and permitting factors (MDE, 2000).

In this study, the focus is on urban areas in developing countries, where typically there is no guidance for either selecting or building structural treatment options. Therefore, as explained in Chapter 4, based on similarities in annual rainfall guidelines from Maryland, U.S. and Magdalena River Catchment, which has been selected as a case study, construction guidelines have been sourced from MDE (2000). According to these guidelines there are five criteria that should be followed as selection criteria: watershed and terrain factors, storm water treatment suitability, physical feasibility, community and environmental factors, location and permitting factors and space constraints. However, as explained in Chapter 1, according to SMA (2011), this information is not available for the case of the Magdalena River catchment in Mexico City. Therefore, the selection process for this City has been conducted on space constraint, which is considered as a common criterion, due to the lack of information on the criteria established in the selected guidelines. Thus, out of the five structural options discussed in Chapter 2, the following three treatment options have been selected for evaluation in this work: bio-retention unit, infiltration trench and porous pavement. Grass filters and water ponds are not considered due to large space requirements, which in densely inhabited cities such as Mexico City are not feasible.

3.3 Design of the treatment practice

The procedure for the design of treatment practice is determined by the availability of local guidance to carry out the design. As explained above, the MDE (2000) has been used in the present study. General assumptions considered for the desing of the selected treatment practices are provided in Chapter 4, while specific assumptions for the bio-retention unit, infiltration trench and porous pavement are given from Chapters 5 to 7 respectively.

3.4 Data collection

Data collection is one of the most challenging stages, since treatment of diffuse water pollution is an emerging industry, so there are no standard methods for the construction, maintenance and decommissioning of the treatment practices. Therefore, information requirements are defined by the method considered for the construction, as well as the guidelines considered for conducting maintenance. As explained in Chapters 5-7, the guidelines provided by the MDE (2000) and Lampe et al. (2005) have been used for identifying the construction and maintenance requirements, respectively.

3.5 Sustainability evaluation

This stage corresponds to the evaluation of the environmental, economic and social impacts of the selected structural treatment practice. Both the tools used and the methodology followed within each evaluation are described in this section.

3.5.1 Environmental evaluation

The environmental evaluation is carried out on a life cycle basis to ensure that all life cycle stages and related impacts are taken into account. Life cycle assessment (LCA) has been used as a tool for these purposes. The LCA methodology followed in this study is based on the methodology defined by the ISO standards 14040/14044 (ISO, 2006a; ISO, 2006b). This divides the LCA procedure into four phases : goal and scope definition, inventory analysis, impact assessment and interpretation (Figure 11). The purposes of an LCA as well as the intended audience are outlined during the goal definition (ISO, 2006a).

In this study the goal is to identify and quantify the environmental impacts of controlling diffuse water pollution using structural treatment practices. In addition to providing information on the life cycle impacts of each option, this information can be used to identify any trade- offs between the life cycle impacts of the runoff treatment compared to leaving the runoff untreated.

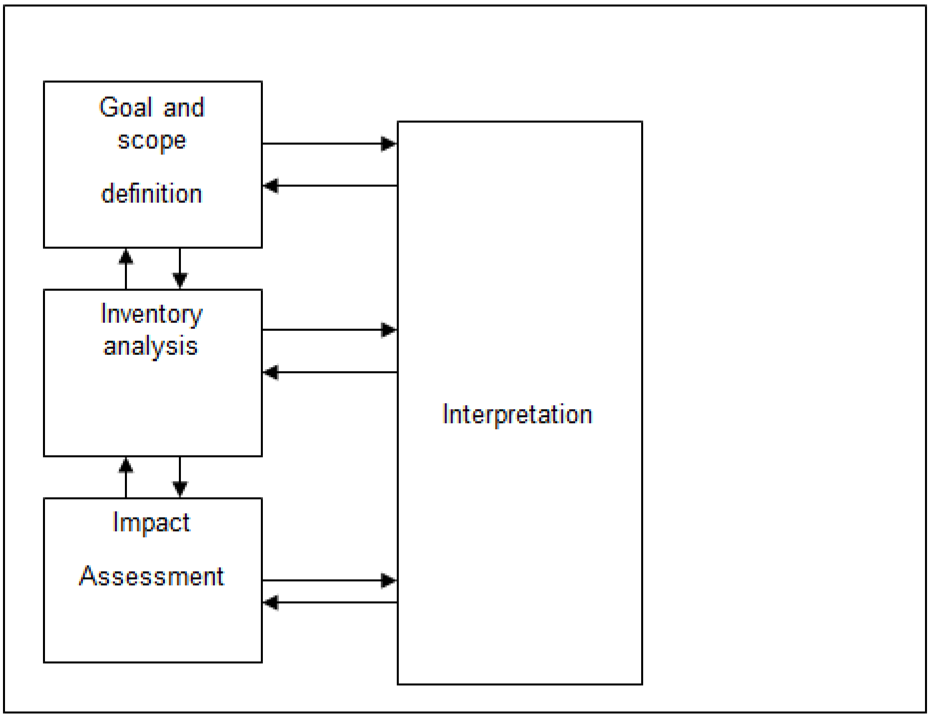

Figure 11 Phases in the LCA methodology

The functional unit is also defined in the goal and scope definition phase (ISO, 2006a). In this study, functional is defined as "treating 1 m^3 of runoff over 30 years". Treatment volume is considered as the functional unit to allow the environmental impacts to be easily scaled for known treatment volumes. In contrast, if another parameter such as drainage area is considered, then the land use, the infiltration rate, and the percentage of impervious surface should be uniform for every case being compared, i.e., the drainage area used as a functional unit must have the same characteristics as assumed in the LCA in order to enable a fair comparison between different treatment practices. As explained in the previous section, the treatment volume is calculated according to the guidelines provided by MDE (2000).

Finally, system boundaries are also established within the goal and scope stage (ISO, 2006a). In this study, the system is defined as the structural treatment practice and the system boundaries are from "cradle to grave ". A detailed explanation of the life cycle stages considered for each treatment practice is given in Chapters 5-7.

Tthe second phase of the LCA methodology is the inventory analysis (ISO, 2006a). In this phase, the calculation of environmental burdens, i.e. mass and energy inputs into the system and emissions to air, water and land is carried out according to equation 1 (Azapagic et al., 2004):

$$B_j = \sum_{i=1}^{i} bc_{j,i}\, x_i \qquad (1)$$

Where:

$bc_{j,i}$ = burden j from process or activity i

x_i = mass or energy flow associated with the activity i

The calculated environmental burdens are then translated into different environmental impact categories in the third phase of the LCA methodology: impact assessment (ISO, 2006a). For that purpose, different life cycle impact assessment (LCIA) methods can be used. The ISO standard (ISO, 2006b) does not specify an LCIA method to be used. The International Reference Life Cycle Data System (ILCD) Handbook suggests a problem-oriented approach with 14 impact categories (JRC, 2011). However, to date, the most widely used method has been the CML 2001 problem-oriented mid-point method (Guinée et al. 2001) which is also selected for use in this study. One of the advantages of this method is that, unlike end-point methods such as Ecoindicator 99 (Goedkoop et al., 1998), the impacts are not aggregated which provides a greater transparency of results and avoids subjective judgements (valuation) of impacts. In order to understand as much as possible about the impacts from controlling diffuse water pollution, the following impact categories have been selected for consideration: non-renewable resource depletion potential (elements and fossil), acidification potential, eutrophication potential, freshwater aquatic ecotoxicity potential, global warming potential, human toxicity potential, marine aquatic toxicity potential, ozone layer depletion potential, photochemical oxidant formation potential and terrestrial ecotoxicity potential. Although not all of the impacts are relevant directly to where the treatment practices are situated (i.e.

Mexico City), they are relevant on a whole life cycle basis, as some of the impacts will be generated elsewhere – not considering them would mean missing out these impacts. Their definition is given in appendix A. The impacts are calculated relative to a reference substance by multiplying the relative contribution $ec_{k,j}$ with burden B_j as shown in equation 2 (Azapagic et al., 2004):

$$E_k = \sum_{j=1}^{j} ec_{k,j} B_j \qquad (2)$$

The fourth and the final phase of the LCA methodology is interpretation, in which the potential for improvements in the system is identified. This stage encompasses four steps: identification of major burdens and impacts, identification of hot spots in the life cycle, sensitivity analysis and evaluation of LCA findings and final recommendations. In this study, these four LCA phases have been followed as shown in Chapters 5-7.

3.5.2 Economic evaluation

Like the environmental impacts, the economic impacts are also evaluated using a life cycle approach. However, unlike the well-defined ISO methodology for LCA (ISO, 2006a;ISO, 2006b), there is no such standard for the economic evaluation. Instead, there are a number of different life cycle costing (LCC) approaches, including that developed by the Society for Environmental Toxicology and Chemistry (SETAC), which enables the estimation of the life cycle cost following a similar approach as in LCA (Swarr et al., 2011). This approach, termed “environmental life cycle costing”, offers the advantage that it can be conducted in parallel to an LCA study, and hence it has been selected and adapted for use in this study.

The SETAC methodology for environmental LCC comprises four stages, which emulate the four phases defined in the ISO standard for the LCA methodology (Swarr et al., 2011): goal and scope definition, economic life cycle inventory, interpretation and review. During the goal and scope definition, the aim of this study is established. This stage could also include identification of the intended application, the reason for carrying out the study, the intended audience and the use of the results (Swarr et al., 2011). In this study, the goal of the economic evaluation is quantification of the life cycle costs and economic hot spots for different treatment options for diffuse water pollution treatment. Moreover, within the scope definition stage, the system and its boundaries, the function provided by the system and the functional unit are defined (Swarr et al., 2011). In this study, these are the same as for the LCA study, as discussed in the previous section.

Once the goal and scope of the study have been defined, economic flows are calculated in the second stage of the LCC methodology: economic life cycle inventory (Swarr et al., 2011). The cost of each stage considered should be estimated, although there is no standardized definition of the cost categories that should be considered. In this study, the costs include the capital, operating and labour costs associated with each life cycle stage, including transport costs. The overall LCC costs are calculated as follows:

$$LCC = \sum_{i=1}^{n} x_i c_i \qquad (3)$$

Where:

LCC = total life cycle costs (US$/m^3)

x_i= material or energy used in the activity i

c_i= cost associated with the activity i

n = total number of life cycle stages.

Finally, the results are reviewed in the final stage of the LCC methodology, with the aim of identifying the hot spots, i.e. the sources of highest costs in the life cycle. The results are also validated by comparison with other related studies. For this reason, the LCC costs are expressed in US dollars ($) and can be found in Chapters 5-7.

3.5.3 Evaluation of social aspects

Social impacts are perhaps the most difficult dimension of sustainability to be evaluated. Unlike the LCA and th LCC methodologies, the development of a standardised methodology for the social life cycle assessment is in its infancy (Jorgensen et al., 2008). Nevertheless, in recent years different approaches to conducting Social Life Cycle Assessment (sLCA) have emerged (Benoit-Norris et al., 2011), increasing available information about of the impact categories as well as the information requirements that should be considered. However, unlike environmental LCA, social LCA is highly dependent on data geographically specific to the area under analysis (Hunkeler,2006;Dreyer,et al., 2006; Brent & Labuschagne, 2006 & (UNEP/SETAC, 2009). Unfortunately, due to this relationship between geographic data requirements and sLCA it is not possible to evaluate the social impacts of controlling diffuse water pollution under a quantitative approach, since according to SMA (2011) there is no social information available for the area of the Magdalena River Catchment. Hence the analysis of the social impacts of the treatment practices considered in this study has been limited to a qualitative analysis defining a set of impacts (employment and training, aesthetic aspect, flood control, water supply, heat island effect, mosquitoes breeding and preservation of ecosystem services), which were selected and developed based on issues considered as relevant in information available in literature. A description of each impact as well as the literature information is given in Chapter 8. It is worth stating that human toxicity potential, estimated as part of LCA, is considered as a social rather than an environmental impact.

It should be noted that the above social aspects do not represent a definitive set as they are based only on the treatment practices and the area under analysis. If there is the potential within a given urban area to consider other treatment options (for example a pond), the scope of the social sustainability assessment should be expanded to include their social impacts as well as those presented in the current study. Other social impacts applicable to water treatment practices in general (such as risks for children, pest or crocodile control) have not been considered in this study because they do not apply to the analysed treatment practices or the area considered.

3.6 Ranking of sustainability impacts

In this stage of the methodology, the treatment practices are ranked based on their environmental, economic and social performance, using the LCA impacts, life cycle costs and social aspects as the indicators of performance. In the absence of stakeholders' preferences, a simple ranking approach has been chosen for the purposes of this study, assuming equal importance of all the sustainability aspects considered. A scale from 1 to 3 has been used, with the former representing the best performance for each sustainability indicator and the latter the worst. For the environmental and economic impacts the ranking is based on the quantitative information obtained from the LCA and LCC studies. For the social aspects which are qualitative, the ranking is based on the potential of the treatment practices to contribute positively or otherwise to the social wellbeing. For example, in the case of mosquito breeding, the option that is more prone to contribute to this effect has the highest score of 3 indicating that it is the worst option for this criterion. Since there is a different number of environmental, economic and social indicators, to avoid bias towards any of them, the final ranking is arrived at in the following way:

- the scores are first summed up for each sustainability dimension - environmental, economic and social – respectively;
- the options are then ranked for environmental, economic and social performance, respectively, based on their scores;
- their individual scores for each of these dimensions are then summed up to give the final scores which then determines the overall ranking of the options.

The treatment option with the lowest total score is considered the most sustainable treatment option. The results of the scoring stage are given in Chapter 8.

3.7 Conclusions and recommendations

Finally, based on the analysis carried out, the conclusions are drawn and recommendations made for relevant stakeholders. These are given in Chapter 9.

3.8 Summary

The methodology applied for evaluating the sustainability of structural options for treatment of diffuse water pollution has been presented in this chapter. Its application is illustrated using the case of Mexico City in the remainder of the dissertation. First, the diffuse water pollution in Mexico City is described in the next chapter, followed by the LCA and LCC studies in Chapters 5-7 for each of the three treatment options selected for this region. The options are compared on their environmental, economic and social sustainability in Chapter 8.

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4 Case Study: Diffuse Water Pollution in Mexico City

This chapter provides an overview of the challenge that Mexico City faces in regards of water pollution control, illustrating that the control of diffuse pollutant sources is in its infancy. Additionally, it also explains the importance of treating this kind of pollution, showing that it is not an easy task due to the lack of a unique government body who could establish standard regulations covering the five entities where the Mexico City water catchment is located. Furthermore, a description of the Magdalena River catchment is given, explaining the reasons for selecting this area as a case study for implementing treatment options for controlling diffuse water pollution. Finally, a description of the procedure employed for the estimation of both treatment volume and pollutant load treated by the treatment options considered in this work is also provided.

4.1 Wastewater treatment in Mexico City

Like most urban centres in the developing world, Mexico City has to deal with the challenge of treating water pollution. Currently, efforts to treat wastewater (which includes polluted runoff) in Mexico City have been focused on controlling pollution from point pollutant sources, treating about 6% of the wastewater produced in the City (Tamargo, 2011). This has brought as a consequence a detriment in water quality of surface water bodies in the city (See Figure 12) (SMA-GDF, 2002). As shown in Figure 12, about 70% of rivers in the city are highly polluted, as most of them are used for sewage discharge.

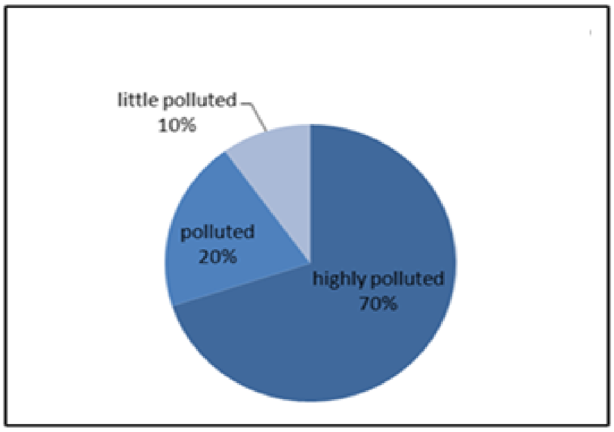

Figure 12 Water quality of the surface water bodies in Mexico City (SMA-GDF, 2002)

In addition, unculverted rivers also received polluted runoff that comes from the agricultural areas in Mexico City, where there are 8,000 ha intended for grasslands and 30,000 ha for livestock operations. In order to improve wastewater treatment, the National Water Commission (NWC) through the Hydric Sustainability Program of the Mexico Valley (SPMV), has started the construction of the Atotonilco wastewater treatment plant, which has a treatment capacity of 35 m^3/s (NWC, 2010). The construction of this plant will increase treatment coverage from 6% to 60% of the wastewater coming from Mexico City. This improvement will help not only to improve coverage of wastewater treatment in the city, but will also improve water quality of irrigation water, which otherwise is used without treatment in the Mezquital Valley (Jimenez, 2009). Thus in regard to wastewater treatment the SPMV, which is the most important programme in sustainable water management for Mexico, is totally dedicated to the control of point pollutant sources, leaving diffuse pollutant sources virtually ignored in the near future, since their treatment is considered as an objective for future treatment plans in 2030 (NWC, 2010)

Current research into diffuse water pollutant sources is at a preliminary stage, with isolated efforts to analyse the sources and effects of diffuse pollution in Mexico City. The first one is an unpublished study by Blanca Jimenez, the contents of which are unknown, as it is not available in the public domain. The second is by Jimenez (2009), who identified potential sources of diffuse pollution in Mexico City. Among these are septic tanks, non-treated municipal water discharges, the lack of a sewage system and combined sewer overflows. As a consequence of this early stage in the research in relation to diffuse water pollution, there are neither guidelines nor regulations for its control. However, in the context of Mexico City controlling this kind of pollution might be helpful, since treated water could represent a water resource which would help to reduce dependence on groundwater sources, which as shown in Table 12 is the main water source for the City.

Table 12 Water sources in Mexico City (NWC, 2009)

Source	Vol (million m^3)	Flow (m^3/s)	Percentage (%)
Aquifer	1876	59.5	72
Rivers and streams	92	2.9	4
Lerma catchment	151	4.8	6
Cutzamala catchment	464	14.7	18
Total	2583	81.9	100

4.2 Water management in Mexico City

Due to the geographical and administrative boundaries of the Mexico City catchment area, water management in Mexico City is fragmented. As shown in Figure 13, this catchment area encompasses five entities: Federal District, Hidalgo, Puebla, State of Mexico and Tlaxcala (SMA-GDF, 2002).

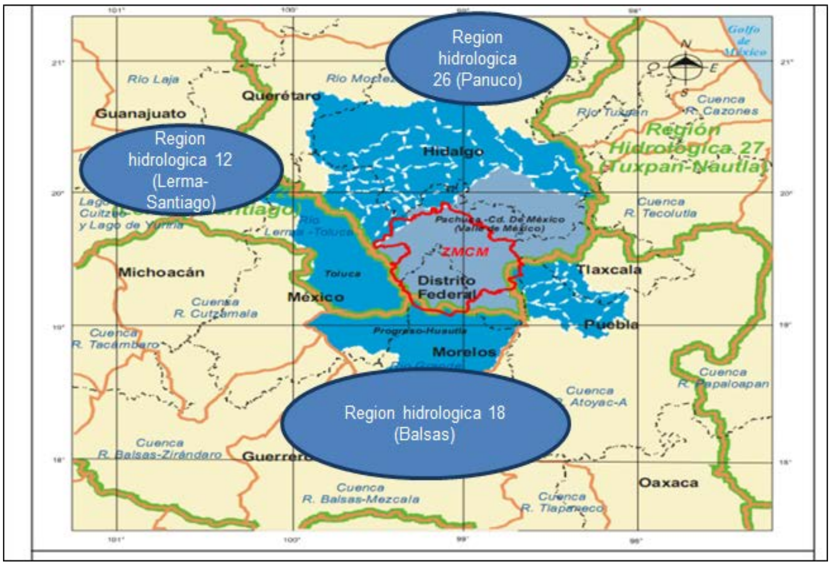

Figure 13 Regions, catchment and hydrological regions in the Mexico City catchment (INEGI, 2002)

The catchment is also divided into three hydrological regions (HR): Balsas (18), Lerma-Santiago (12) and Panuco (26) (SMA-GDF, 2002). Because of that, the National Water Commission defined administrative-hydrological regions where the catchment is the hydrological unit but the legal rights of the water are owned by the counties where the catchment is established. However, even with the definition of the administrative-hydrological regions it is difficult to establish who is in charge of dealing with water management in the Mexico City catchment area. This fragmentation in regard to water management is included in this section to illustrate an identified challenge for obtaining information to conduct this research. As explained below, it is due to this, and to the early stage of research into diffuse water pollution in Mexico City that alternative data sources had to be used.

4.3 The Magdalena River Catchment

In Mexico City, the Magdalena River is the ‘victim’ of both governance problems regarding water management and various sources of pollution. The river represents the heart of one of the few reserved areas in the City. As shown in Figure 14, it is located in the Magdalena River Catchment (MRC) with an area of 34.7 km^2 shared among five counties: Magdalena Contreras (70%), Cuajimalpa (11.82%), Alvaro Obregon (11%), Coyoacan (5%) and Tlalpan (2%) (SMA-GDF, 2006). The main stream in the catchment, the Magdalena River, has a length of 27.2 km and flows through the green and urban area.

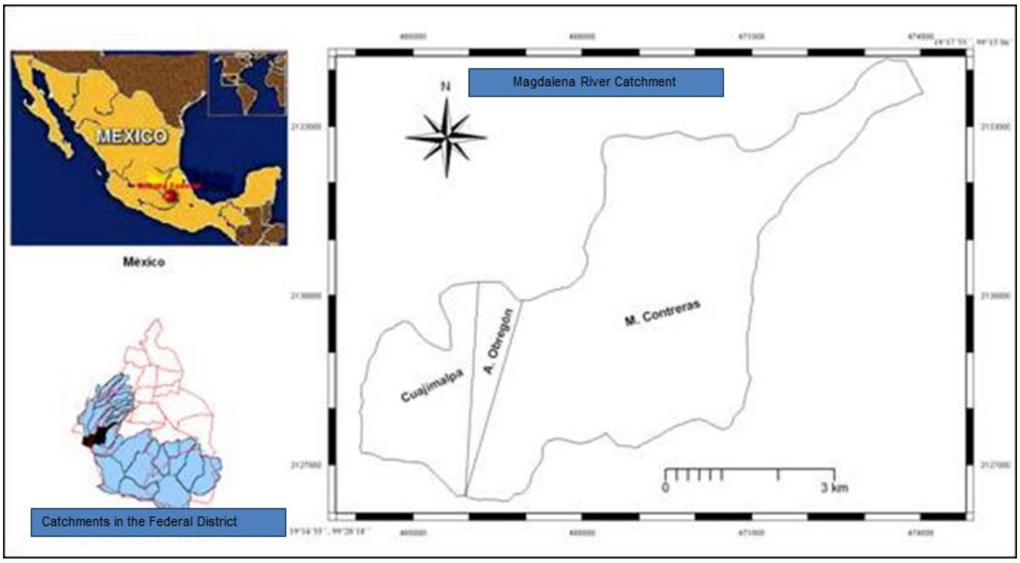

Figure 14 Magdalena River Catchment (Sanchez, 2007)

Due to its location, the Magdalena River receives both point and diffuse pollution from the urban and rural areas (Almeida et al., 2007). Point pollutant sources are well defined and identified, encompassing 19 point discharges from the sewage system in the area that corresponds to the urban area (NWC, 2009). As to the diffuse pollutant sources, they have not been fully identified; as mentioned above, information about this kind of pollution is not yet available in the public domain.

The government of the Federal District has launched a plan for improving the water quality of the MRC, known as “The Magdalena River Basin Comprehensive Management and Sustainable Usage Master Plan” (NWC, 2009). This project is aimed at the analysis of the water quality in the Magdalena River for both point and diffuse sources. Its importance lies in the fact that it represents a milestone in the analysis of water quality in rivers in the City, the results of which could be applied in improving the quality of other rivers in the City.

Both point and diffuse pollution need to be addressed in order to improve water quality in rivers and groundwater sources. Although the treatment for point sources might be clear, the selection of the most adequate and sustainable treatment options for controlling diffuse pollutant sources is yet to be carried out. Hence the area of the MRC has been selected for the research purposes of this study. However, since the control of diffuse pollution is at a preliminary stage in Mexico, there are no construction guidelines for the implementation of treatment options. Therefore, construction calculations are based on the construction guidelines from Maryland County U.S, which have been used because of the similarities in average annual rainfall with Magdalena River Catchment. Annual average rainfall in Maryland County is 1066 mm (NOOA, 2010) while annual average rainfall for Magdalena river catchment is 1100 mm (Mendoza, 2009). Despite these similarities, the climates of Mexico and Maryland are certainly different, but the Maryland data is the closest known available match. More accurate modelling of local conditions would be required before installing a treatment option, and a field test in the study area is recommended in the construction guidelines (MDE, 2000). Based on these guidelines the volume treated is calculated according to the design objectives of the selected treatment practice. In this work the design aim is to control runoff quality, therefore treatment volume is calculated based on the rainfall depth from a one year–24 hour duration storm[2] (MDE, 2000). This event has been selected because it is the storm event used in Maryland to treat the first flush, which the most polluted portion during a rainfall event (MDE, 2000).Volume required to store the polluted portion of a rainfall event is defined as water quality volume (WQv) which is calculated according to equations 4 and 5 (MDE, 2000):

$$WQv = P \times A \times Rv \quad (m^3) \qquad (4)$$

$$Rv = 0.05 + 0.009 \times I \quad (-) \qquad (5)$$

Where:

WQv = water quality volume (m^3)

P = annual rainfall depth (m)

A = catchment area (m^2)

Rv = runoff coefficient (-)

I = site’s impervious cover (percentage).

[2]1 year-24 hour refers to 1 year return period, 24-hour duration storm - a storm lasting 24 hours and depositing this amount of water takes place on average once in a year in the specified location.

Imperviousness (I) has been assumed as 100%, in order to consider the worst case where maximum runoff could be generated. Since the treatment options considered in this study are micro-scale treatment technologies, their corresponding drainage area has been assumed as 0.4 ha, as recommended in the design manual (MDE, 2000). Thus, the total WQv is equal to:

$WQv = P \times A \times (0.05 + 0.009 \times I)$

$WQv = (0.0254\ m) \times (4047^{*}\ m^2) \times [0.05 + (0.009 \times 100)]$

$WQv = 98\ m^3$ [3]

In relation to the operation stage, pollutant mass to be treated is calculated as pollutant load. The corresponding computation for pollutant load is performed according to equation 6 (Chandler, 1994):

$$L = R \times C \times A \quad (g) \qquad (6)$$

Where:

L = pollutant load (g)

R= average annual runoff (m)

C = pollutant concentration (mg/l)

A= area (m^2).

Average annual runoff R is calculated as:

$$R = P \times Pj \times Rv \qquad (7)$$

Where:

P = annual rainfall (m)

Pj = fraction of rainfall events that produce runoff (0-1)[4].

Pollutant load varies with the pollutant concentration, which has been assumed as event mean concentration (EMCs). This is useful when due to

[3] This value corresponds to 0.4 ha as referred above

[4] This value has been fixed as 0.9 in order to evaluate a scenario where 90% of the rainfall events in MRC produce runoff.

the lack of budget and time it is not possible to obtain local information, as in the case of the present study. In this work it has been assumed that the treatment option treats polluted runoff whose composition is mainly made of heavy metals; this composition has been considered because it constitutes a typical pollutant concentration from motorways or parking lots (Nixon & Saphores, 2007), which according to the National Water Commission might pose a pollutant source in the Magdalena river catchment (NWC, 2009).

As stated before, in Mexico the analysis of diffuse water pollution is at an early stage, therefore information about EMC is not available in the public domain. In order to find an appropriate data set of pollutant concentrations an extensive literature review was conducted. As a result two data sets were found, one from the EMC from Beijing, China (Zhao et al., 2010) and another from California, U.S. (Kayhanian et al., 2007). From these two options, the California data have been selected. The reason for this is that the Californian data provide the highest range reported in the literature for most of the parameters apart from cadmium (see Table 13) and includes measurements from 34 highway sites, encompassing a wide variety of daily traffic levels (Kayhanian et al., 2007), which is considered a key factor in the generation of pollutants in parking lots (Kayhanian et al., 2003).

Table 13 Comparison of EMC among Beijing and California

City	TSS (mg/l)	Cd (mg/l)	Cr (mg/l)	Cu (mg/l)	Ni (mg/l)	Pb (mg/l)	Zn (mg/l)	Reference
Beijing	315-437	0.222-0.323	0.009-0.060	0.039-0.080	0.01-0.04	0.06-0.08	0.341-0.470	(Zhao et al., 2010)
California	1-2988	0.002-0.03	0.001-0.094	0.001-0.27	0.001-0.13	0.001-2.6	0.005-1.68	(Kayhanian et al., 2007)

Therefore, the use of the data set from California for the case of Mexico City is justified because it is representative of a variety of daily traffic levels taking place in an urban area. If in future information for pollutant concentrations for the area of the Magdalena River Catchment becomes available, this can be used for increasing the accuracy of the results obtained here. Thus, based on equation 6 and previously defined data sources, the pollutant load has been calculated as:

L_{TSS}= ((0.0254m x 0.9 x 0.95) x (2988 mg/l)x(4047 m^2))/1000 = 262kg/year
L_{Cd}=((0.0254m x 0.9 x 0.95) x (0.007 mg/l)x(4047 m^2))/1000=0.0006 kg/year
L_{Cr}=((0.0254m x 0.9 x 0.95) x (0.009 mg/l)x(4047 m^2))/1000 = 0.0008kg/year
L_{Cu}=((0.0254m x 0.9 x 0.95) x (0.033 mg/l)x(4047 m^2))/1000=0.0029 kg/year
L_{Ni}=((0.0254m x 0.9 x 0.95) x (0.011 mg/l)x(4047 m^2))/1000 = 0.0010 kg/year
L_{Pb}=((0.0254m x 0.9 x 0.95) x (0.047 mg/l)x(4047 m^2))/1000 = 0.0041 kg/year
L_{Zn}=((0.0254m x 0.9 x 0.95) x (0.187 mg/l)x(4047 m^2))/1000 = 0.0164 kg/year

The environmental benefits of controlling diffuse water pollution in cities of the developing world are not well understood; but this knowledge is important because, globally, most population growth takes place in developing countries (UNFPA, 2007). Therefore, this study could contribute to the development of sustainable storm water management perspectives which consider runoff as a useful water resource instead of as waste, not only in Mexico City but also in other megacities in the world.

In addition to pollutant load, another common element to all the case studies is the functional unit (unit of analysis), which has been defined as "treatment of 1m^3 of runoff over the 30-year life span of the treatment practice".

4.4 Summary

In Mexico City, as in other megacities in the world, wastewater treatment is a challenge that should be faced considering the treatment of both point and diffuse water pollutant sources. However as explained above, the analysis of the latter is difficult due to two factors: the early stage in research into diffuse water pollution, and the fragmentation of water management. In spite of these, treating pollutant sources with a diffuse origin could represent an option to transform polluted runoff into a useful resource. In this research three treatment practices have been considered in order to treat polluted runoff in the MRC, which has been selected as a case study. The analysis of both the environmental and economic impacts of these treatment options follows in Chapters 5-7.

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5 Bio-retention unit: Environmental and economic evaluation

As explained in Chapter 2, one of the most common treatment practices to control diffuse water pollution is the bio-retention unit. This chapter describes the life cycle evaluation of the environmental impacts and economic costs of this treatment practice.

5.1 Environmental evaluation

5.1.1 Goal and scope

The goal of this study is to quantify the environmental impacts and identify the hot spots in the life cycle of a bio-retention unit for treating diffuse water pollution. The scope of the study is from 'cradle to grave' and the following life cycle stages are included: construction, operation (runoff treatment), corrective and preventative maintenance and decommissioning (Figure 15). For research purposes other researchers (e.g. Kirk (2006) and Lampe et al. (2005)) have considered 30 years as the life span of bio-retention units, Therefore this study also considers the same life span. As shown in Chapter 4 the functional unit has been defined considering this life span.

The model developed in this work has been tested using the case study of the Magdalena river catchment in Mexico City (see Chapter 4), increasing knowledge about diffuse water pollution in big cities in the developing world. Additionally, the present study could help to understand the environmental impacts of implementation of a bio-retention unit from a life cycle point of view, since currently there is a lack of life-cycle data for storm water bio-retention units and only two previous LCA studies have been carried out: Kirk (2006) and Flynn & Traver (2012).

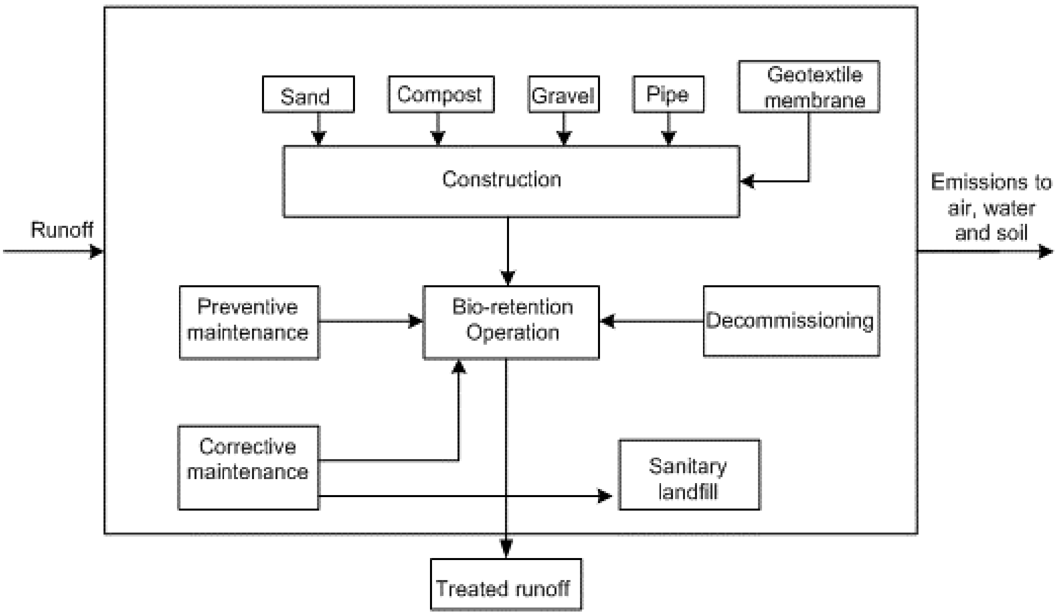

Figure 15 Life cycle diagram of a bio-retention unit

5.1.2 Inventory analysis

The inventory is divided into five stages: construction, operation, corrective maintenance and preventive maintenance and decommissioning. Data sources and assumptions considered in each stage are described below.

5.1.2.1 Data sources

As shown in Table 14 and Table 15 , data for the construction stage are divided into three groups, with different information sources. Information required and data sources for (i) calculating the runoff treatment volume and (ii) sizing the bio-retention are shown in Table 14. The third group encompasses data sources for the background and foreground life cycle inventories of the materials, machinery and transport used during the construction stage. These data sources, along with the system boundaries of the processes considered for each material, are shown in Table 15. In order to adapt these data as far as possible to Mexican conditions, the Mexican electricity mix has been used for all the processes with electricity requirements (Santoyo-Castelazo et al., 2011). All material transport has been modelled using a 40 tonne truck, while construction activities have been modelled considering a hydraulic digger. This is due to the impossibility of obtaining information for any company dedicated to the installation of bio-retention units.

Table 14 Information sources for the construction stage

(i) Information sources for runoff treatment volume		
Parameter	*Value*	*Data source*
P (annual rainfall)	25.4 mm	(MDE, 2000)
(ii) Information sources for bio-retention sizing		
Parameter	*Value*	*Data source*
Pooling depth	0.152 m	(MDE, 2000)
Soil media depth	0.46 m	(MDE, 2000)
Gravel depth	0.30 m	(MDE, 2000)

Table 15 Data sources and considered processes for materials used within the construction stage

Materials		
Material	*Considered processes*	*Data source*
Sand	Whole manufacturing process for digging, transport and machinery used for operation	(Kellenberger et al., 2007)
Gravel	Whole manufacturing processes, machinery and transport	(Kellenberger et al., 2007)
PVC	Whole production PVC	(PE International, 2010)
Geotextile membrane	Whole manufacturing process of geotextile membrane	(PE International, 2010)
Compost	Construction, operation, dismantling of the compost plant. Compost transport to the final user is not considered	(Nemecek & Kagi, 2007)
Transport		
Vehicle	*Considered processes*	*Data source*
40 tonne truck	Construction, use (including maintenance) and end of life	(Spielmann et al., 2007)
Machinery		
Machinery	*Considered processes*	*Data source*
Hydraulic digger	Materials, transport of the parts to the assembly point and energy and heat requirements during use phase.	(Kellenberger et al., 2007)

In relation to the corrective and preventive maintenance life cycle stages, information regarding the frequency of these activities has been sourced from Lampe et al. (2005). Frequency reported in this source corresponds to field study data from the UK and U.S. (see Table 16). Life cycle information about background processes considered for both vehicles and machinery used within the maintenance stages is given in Table 17.

Table 16 Frequency for the activities of corrective and preventive maintenance stages

Corrective maintenance stage	
Activity	*Frequency*
Inspection	Every four years
Sludge removal	Every four years
Preventive maintenance stage	
Activity	*Frequency*
Inspection	Every three years
Mowing	Once per year

Table 17 Data sources for life cycle data for corrective and preventive maintenance stages

Corrective maintenance stage		
Machinery	*Included processes*	*Data source*
Hydraulic digger	Materials, transport of the parts to the assembly point and energy and heat requirements during use phase.	(Kellenberger et al., 2007)
Vehicle	*Included processes*	*Data source*
Passenger car	Vehicle manufacturing, operation and disposal	(Spielmann et al., 2007)
Preventive maintenance stage		
Machinery	Included processes	Data source
Hydraulic digger	Materials, transport of the parts to the assembly point and energy and heat requirements during use phase.	(Kellenberger et al., 2007)
Vehicle	Included processes	Data source
Passenger car	Vehicle manufacturing, operation and disposal	(Spielmann et al., 2007)

As explained in Chapter 4, in relation to the operation stage, pollutant load has been calculated based on the simple method (Chandler, 1994). According to this method, the following information is required to conduct the estimation: pollutant concentration, annual rainfall, imperviousness and the percentage of events that produce runoff. Pollutant concentration data refer to event mean concentration (EMC[5]). The EMC data for Mexico City are not available, so as explained in Chapter 4, these data had to be sourced from the literature. Thus, the EMC data for the total suspended solids (TSS), Cd, Cr, Cu, Pb, Zn are based on the information in Kayhanian et al. (2007) and represent the average values as given in Table 18. Data sources for annual rainfall, imperviousness and percentages are given in Table 19.

Table 18 Average EMC values used in this study (Kayhanian et al., 2007)

Parameter	**EMC mg/l**
TSS	2988
Cd	0.007
Cr	0.009
Cu	0.033
Ni	0.011
Pb	0.047
Zn	0.187

[5]It is defined as a statistical parameter used to represent concentration of pollutant from a flow-weighted composite sample (Novotny & Olem, 1994).

Pollutant mass in the effluent has been calculated based on a simple mass balance approach. It is simple in the sense that neither chemical nor physical reactions have been considered.

Table 19 Information source for pollutant load calculation

Parameter	Value	Information source
Drainage area	0.4 ha	(MDE, 2000)
Annual rainfall depth	25.4 mm	(MDE, 2000)
Fraction of rainfall events that produces runoff	90%	(Chandler, 1994)
Imperviousness	100%	(MDE, 2000)

The mass balance calculation requires pollutant removal efficiencies data, which have been sourced from the latest version of the National pollutant performance database (CWP, 2007) and this is shown in Table 20.

Table 20 Information source for the operation stage (CWP, 2007)

Parameter	% removal
TSS	59
Cr	90
Cu	93
Ni	93
Pb	93
Zn	93

Finally, data source and the background process considered for the decommissioning stage are given in Table 21. They only include data about machinery, since this stage has been assumed to involve removing the bio-retention unit elements and using clean top soil to restructure the area where the bio-retention unit is built.

Table 21 Information sources for decommissioning stage

Machinery	Considered processes	Data source
Hydraulic digger	Materials, transport of the parts to the assembly point and energy and heat requirements during use phase.	(Kellenberger et al., 2007)

This work represents the first attempt to understand the environmental impacts in big cities in the developing world, and is also only the third life cycle assessment (in addition to Kirk (2006) and Flynn and Traver (2012)) of a bio-retention unit for storm water management purposes in any location, hence the data described above are considered adequate.

5.1.2.2 Assumptions

In the bio-retention design different assumptions are made in each stage considered in the life cycle. A description of these assumptions is given in this section.

It has been assumed that the bio-retention unit is built in the Magdalena river catchment (MRC) in Mexico City. As explained in Chapter 4, due to the lack of information for the case of Mexico in regard to installation of bio-retention units, the MDE (2000) guidelines have been used as a reference in this study. According to the MDE (2002) construction guide, the volume treated is calculated based on the design objectives of the bio-retention unit. In this work the design aim is to control runoff quality, the calculations for the treatment volume considering this criterion are shown in Chapter 4. Based on this estimation, treatment volume (WQv) amounts to 98 m^3, which is used to estimate the surface area of the bio-retention unit as follows:

$$SA = WQv/d \qquad (m^2) \tag{8}$$

Where:

SA= surface area of bio-retention unit (m^2)

WQv = water quality volume (m^3)

d = depth of the treatment area of the bio-retention unit (m)

The depth of the bio-retention unit was assumed at 0.76 m (MDE, 2000). This depth corresponds to the treatment area which is composed of two layers made of gravel and soil media (see Chapter 2 for design of bio-retention units). The soil media layer is composed of a mixture of 80% sand, 10% top soil and 10% compost according to the information given in the construction guide (MDE, 2000). The volume of soil media and gravel is calculated by multiplying the surface area by the depth of each layer (0.46 m and 0.30 m, respectively).

Thus:
SA = 98 m^3 /(0.46 m+ 0.30 m)
SA = 128 m^2

Finally, transport of material used during the construction stage has been modelled assuming a transport distance of 100 km. This distance has been selected in the absence of this information, assuming that construction materials can be sourced within this distance.

In relation to the operation stage, pollutant mass to be treated is calculated as pollutant load. The corresponding computation for pollutant load is performed according to equation 8 (Chandler, 1994), as explained in Chapter 4.

The maintenance stage is divided into two sub-categories: preventive and corrective maintenance (Lampe et al. 2005). Activities considered for preventive maintenance are inspection activities and mowing, which refers to cutting grass or trimming the canopies of trees. These activities are selected because they are required in order to keep the bio-retention unit working properly (Lampe et al. 2005). In regard to corrective maintenance, there is little information since the bio-retention unit has only recently been used as a treatment option (Lampe et al. 2005). Activities considered as corrective maintenance could encompass repair or replacement of the elements of the treatment area, which based on the selected design could include shrubs, trees, sand, gravel, soil, or geotextile membrane (EPA, 1999). Since it is not possible to anticipate which element of the bio-retention unit might need to be either replaced or repaired, in this study corrective maintenance activities have been reduced to inspection activities, which are required on a regular basis previous to the identification of any individual failure. Additionally, sediment removal is also considered as part of the corrective maintenance activities, in order to analyse its environmental impacts.

Sediment disposal is a major concern in relation to the implementation of bio-retention units, however it is barely conducted in field bio-retention units

(Lampe et al. 2005). Sediment quality has been analysed from the environmental perspective by Lampe et al. (2005) and Davis et al. (2003). In the first case sediment quality assessment was based on the analysis of 565 sediment samples from different storm water management options (Lampe et al. 2005). The parameters evaluated were As, Cd, Cu, Pb, Zn, Ni and Hg (mg/kg). Maximum concentration values were compared to the threshold levels established for hazardous materials in the European Hazardous Waste Directive (HWD Council Directive 91/689/EEC, cited in Lampe et al. 2005) and U.S. hazardous waste threshold values. Results from this comparison are shown in Table 22 and they are used to illustrate the fact that sediment removed from a bio filtration strip is not considered as hazardous material.

Table 22 Maximum concentration values in sediment from different storm water management options across US and EU (Lampe et al. 2005)

		Maximum concentration (mg/kg)						
Location	*Structure*	*As*	*Cd*	*Cu*	*Pb*	*Zn*	*Ni*	*Hg*
U.S.	Bio-filtration strip	2.90	1.2	60	144	337	13	0.05
EU threshold value for hazardous waste		30,000	25,000	NA*	2,500	NA	1,000	2,500
U.S. thresholds value for hazardous waste		5,000	1,000	25,000	5,000	250,000	20,000	NA

*NA: Not available

The analysis conducted by Davis et al. (2003) evaluates the metal content (specifically Cd, Cu, Pb and Zn) from sediment samples collected specifically from a bio-retention unit. Metal content in the samples was compared to the metal threshold values established for metal accumulation caused from the application of bio-solids from waste water treatment plants in agricultural land in the United States (Davis et al., 2003). It was found that hazardous levels from the sediment removed from the bio-retention unit would be reached after 20 years for Cd, 77 years for Cu and 16 years for Pb and Zn.

The analyses described above are focussed on the assessment of sediment quality - since neither Davis et al. (2003) nor Lampe et al. (2005) analysed the environmental impacts produced from sediment disposal. Therefore, in order to quantify these environmental impacts, sediment disposal (and its transport to the disposal site) are also considered here as part of the activities conducted during the corrective maintenance stage and sediment is assumed to be removed every four years.

Based on the above-mentioned studies (Davis et al. 2003; Lampe et al. 2005), the sediment is not considered as hazardous waste. Although the time reported by Davis et al. (2003) for Cd, Pb and Zn to reach hazardous levels is 20 and 16 years respectively, which is less than the life span considered, it is assumed that hazardous level is not reached, since sediment is removed every four years. Hence, it has been assumed that sediment is disposed of in a sanitary landfill.

It has been considered that the sanitary landfill works under Swiss conditions, which encompass leaching collection, leaching treatment and municipal incineration processes (Doka, 2009). These conditions are different to the Mexican sanitary landfill in regard to the treatment technology. In this study it has been assumed that the sanitary landfill considered is Bordo Poniente landfill, which has been selected because it is the only one under operation in Mexico City (GODF, 2010). The technology used in this landfill encompasses both leaching collection and leaching treatment (Najera et al., 2010). In developing countries such as Mexico, this treatment is reduced to evaporation and recirculation (Najera et al., 2010). Hence, in order to adapt as far as possible the Swiss sanitary landfill process to the conditions modelled in this study, two adaptations have been carried out. The first one is in regard to the sediment composition treated in the landfill which has been modified using the model developed by Doka (2009) in order to introduce the metal composition shown in Table 25. The second adaptation corresponds to the use of the Mexican electricity (Santoyo-Castelazo et al., 2011) to model the energy requirements in the sanitary landfill. The amount and composition of sediment washed into bio-retention units varies in relation to the watershed characteristics. However, in this study sediment has been assumed to be equal to the total suspended solids (TSS). In order to calculate the sediment volume that is removed, a sediment density of 1.5 kg/m^3 has been assumed (Lampe et al. 2005). Transport distance has been assumed as 100 km due to the lack of information.

Finally, the decommissioning stage has been modelled assuming that all the elements of the bio-retention unit are removed and transported from the installation area. Installation of new vegetation is not considered, since this depends on the local conditions of the removed bio-retention unit. However, it has been assumed that clean soil is located instead of this treatment option. Based on the above, the assumptions and inventory per stage for the bio-retention unit are summarised in Table 23 to Table 27. To address some of the uncertainties in the assumptions and data used, sensitivity analyses have been carried out, as discussed in Section 5.1.4. Prior to that, the results of the impact analysis are presented next.

Table 23 Summary of the assumptions for the bio-retention unit

(i) WQv		**(ii) SA**	
Parameter	*Value*	*Parameter*	*Value*
Annual rainfall depth	0.0254 m	WQv	98 m^3
Catchment area	4047 m^2	Bio-retention depth	0.76m
Runoff coefficient	0.95		

Table 24 Inventory for the construction stage

Construction Stage		
Material	*Quantity*	
Sand	47 m^3	
Top soil	6 m^3	
Compost	6 m^3	
Compost for pooling area	19 m^3	
PVC	13 m	
Geotextile membrane	128 m^2	
Gravel	39 m^3	
Transport		
Material	*Mass (tonne)*	*tonnexkm*
Sand	75	7508
Top soil	0.0058	0.58
Compost	15	1500
PVC	0.002	0.200
Geotextile membrane	0.131	13.1
Gravel	59	5900

Table 25 Inventory for the preventive and corrective maintenance stage

Preventive maintenance stage		
Activity	*Frequency*	*Quantity*
Inspection	Every three years	1000 p·km [a]
Mowing	Once per year	3844 m^2 [b]
Transport		
Personnel transport for mowing	Once per year	6000 p·km [c]
Corrective maintenance		
Activity	*Frequency*	*Quantity*
Inspection and removal	Every four years	961 m^2
Sediment composition		
Parameter	*kg accumulated over 4 years*	*mg [e] pollutant/kg sediment*
TSS	618.62	-
Cd	0.00221	3.57
Cr	0.00272	4.39
Cu	0.01093	17.67
Ni	0.00359	5.80
Pb	0.01534	24.79

The values shown have been calculated as: a) 100km* x 1person x 10trips; b) 128 m^2/year x 30 years; c) 100 km x 2 persons x 30 trips; d) 128 m^2 x 7.5 inspection trips; e) estimations for sediment accumulation are shown in Appendix B.

Table 26 Inventory for the runoff stage

Pollutant load in [a]	**Mass (kg/year)**
TSS	262.13
Cd	6.00E-04
Cr	8.00E-04
Cu	2.90E-03
Ni	1.00E-03
Pb	4.10E-03
Zn	1.64E-02
Pollutant load out	
TSS	107.47
Cd	6.14E-05
Cr	7.54E-05
Cu	2.05E-03
Ni	6.75E-05
Pb	2.90E-04
Zn	1.15E-03
Pollutant load accumulated	
TSS	154.65
Cd	6.00E-04
Cr	7.00E-04
Cu	2.70E-03
Ni	9.00E-04
Pb	3.80E-03
Zn	1.53E-02

[a] The pollutant load on the inlet of the bio-retention unit has been calculated based on the average EMC values from Kayhanian et al. (2007).

Table 27 Inventory for the decommissioning stage

Material removal		
Material	*Quantity*	
Sand	46 m^3	
Top soil	6 m^3	
Compost	6 m^3	
Compost for pooling area	19 m^3	
PVC	13 m	
Geotextile membrane	128 m^2	
Gravel	39 m^3	
Material transport		
Material	*Mass (tonne)*	*tonnexkm*
Sand	75	7508
Top soil	0.0058	0.58
Compost	15	1500
PVC	0.002	0.200
Geotextile membrane	0.131	13.1
Gravel	59	5900
Closure		
Material	*Quantity*	
Soil	117 m^3	

5.1.3 Impact assessment

The CML 2001 impacts characterisation method (Guinee et al. 2001) has been used to assess the environmental impacts. The impact assessment has been conducted using the GaBi software package version 4.4 (PE, 2010).

As shown in Figure 16 the construction stage is the largest contributor for most of the impact categories. Preventive maintenance is the second largest contributor, followed by corrective maintenance. The contribution from decommissioning is negligible. A detailed description of the contributions within each impact category is given below.

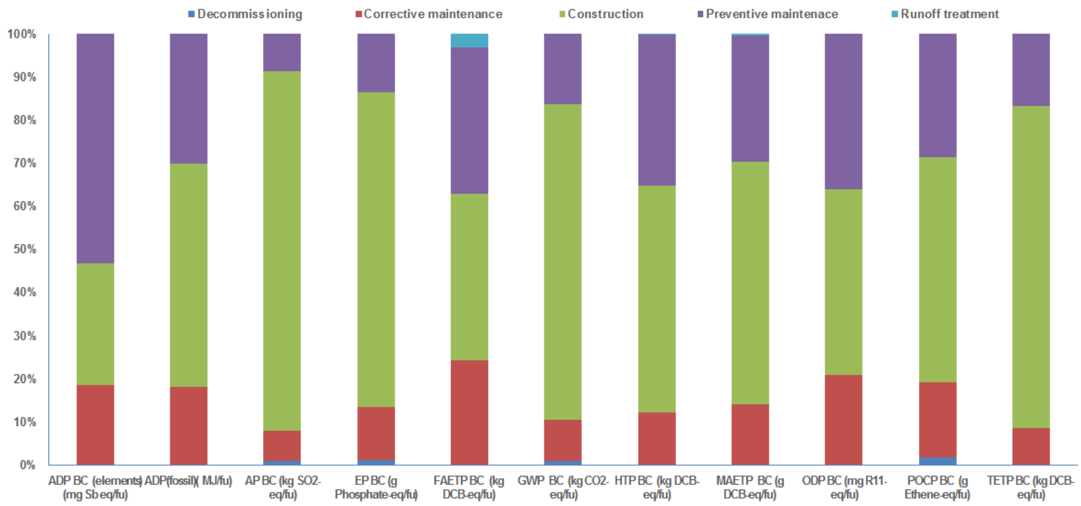

Figure 16 Environmental impacts and contribution analysis for the bio-retention unit
[BC: Base case; fu: functional unit = 1 m^3 of runoff treated]

5.1.3.1 *Abiotic depletion potential of elements (ADP elements)*

Total abiotic depletion (ADP elements) is estimated at 2.86 mg Sb per functional unit (fu). The largest contribution within this impact category comes from the preventive maintenance stage, accounting for 53% of the total ADP (Figure 16). Within the contribution from the preventive maintenance stage, 33% comes from transport activities (See Figure 17). Maintenance activities represent overall the main contributor to the transport activities.

5.1.3.2 *Abiotic depletion potential of fossil resources (ADP fossil)*

Total abiotic depletion potential (fossil) is about 27 MJ/fu. The largest contribution within this impact category comes from the construction stage (52%); see Figure 16. Within the construction stage, the largest individual contributions are from the geotextile membrane and transport, accounting for 13% and 10% respectively (Figure 18).

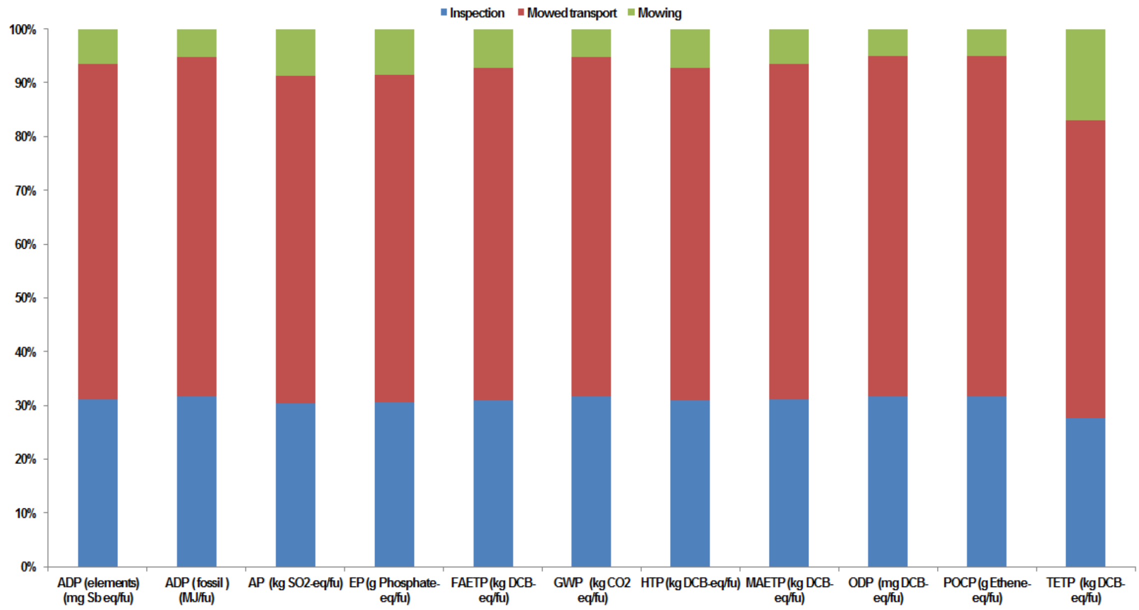

Figure 17 Contribution analysis for the preventive maintenance stage

5.1.3.3 Acidification potential (AP)

Total AP is estimated at 0.023 kg SO_2 eq/fu. The main contribution to this impact category comes from the construction stage (Figure 16), which accounts for 83% of the total AP. Within this stage, the largest contributor is the composting process, accounting for 62% of the AP from the construction stage (Figure 18).This is due to the emissions of ammonia (34%), hydrogen sulphide (18%) and nitrogen oxides (6%) to air. Activities conducted during the preventive maintenance stage represent the second largest contributor to the total AP, accounting for about 9%.

5.1.3.4 Eutrophication potential (EP)

This impact is equal to 4.9 g phosphate eq/fu. The construction stage is the major contributor, accounting for 73% of the total EP (Figure 16). As can be seen in Figure 18, compost accounts for 55% of this contribution, largely due to the emissions to air of ammonia, nitrous oxides and nitrogen oxides, which account for 36%, 8% and 6% respectively, and are produced during the composting process.

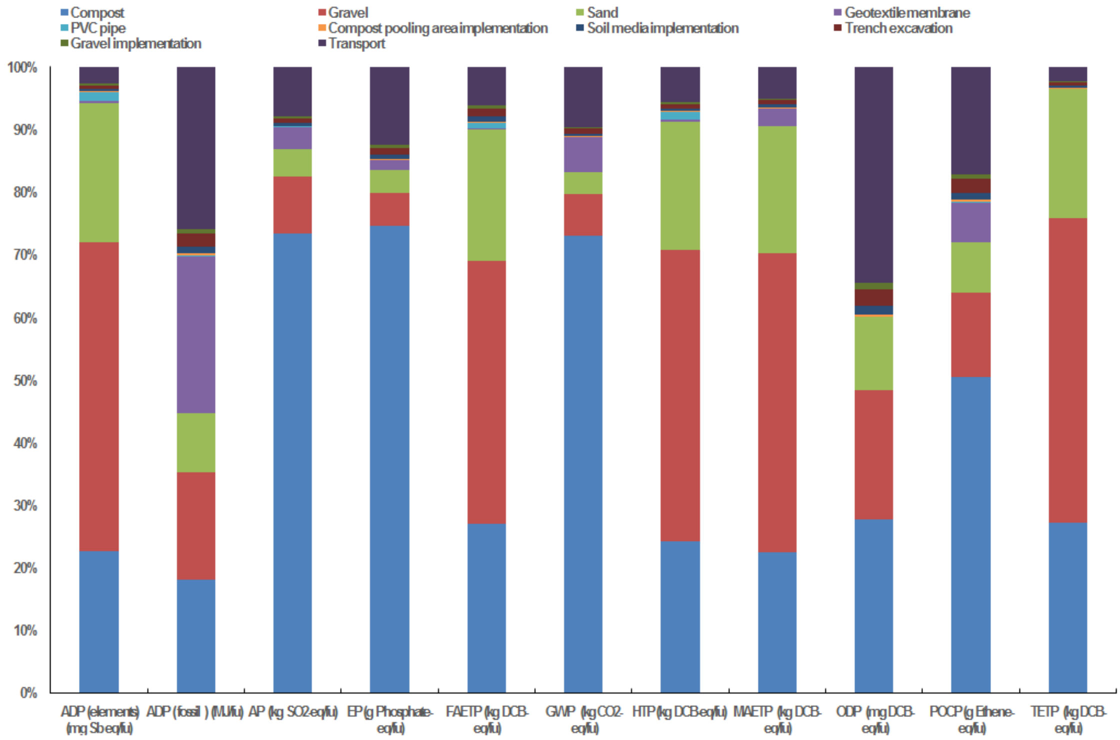

Figure 18   Contribution analysis from the construction stage

The second largest contributor to total EP is the preventive maintenance stage; it contributes about 13% of the total EP.

5.1.3.5 Freshwater Aquatic Ecotoxicity Potential (FAETP)

Total FAETP is estimated at 0.23 kg DCB eq/fu, with the construction stage contributing 39% and preventive maintenance 34% of the total FAETP (see Figure 16). As shown in Figure 18, the largest contributing burdens within the construction stage are from gravel extraction, accounting for about 16% of the total FAETP. This is due to the emissions to freshwater of vanadium (+III), beryllium and cobalt. The major contributing burdens in the preventive maintenance stage are from transport (21%), including vanadium, beryllium and cobalt.

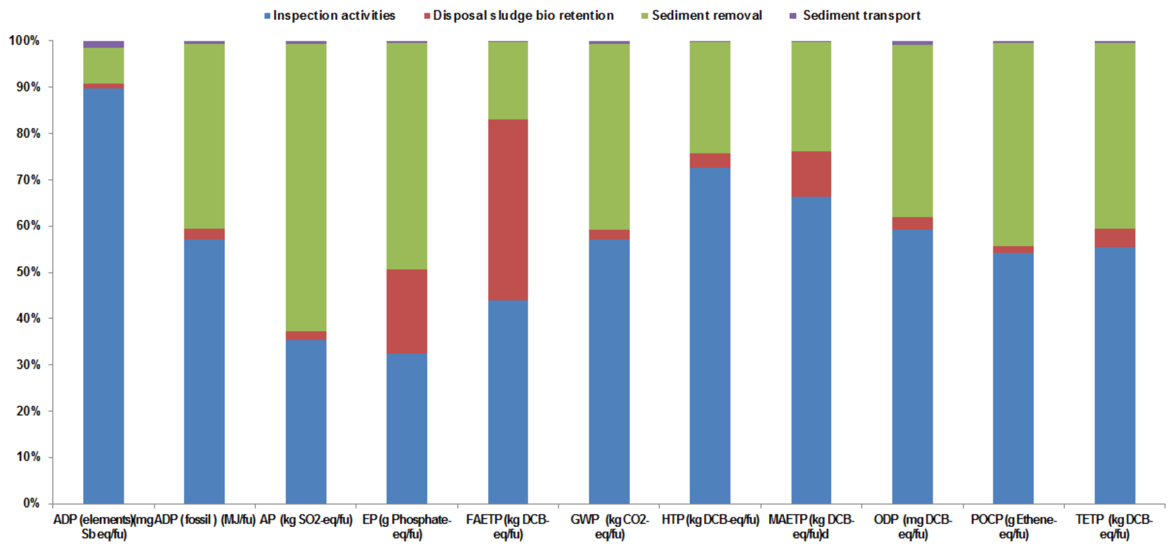

Figure 19 Contribution analyses from the corrective maintenance stage

5.1.3.6 Global warming potential (GWP)

The bio-retention unit has a GWP of 3.58 kg CO_2 eq/fu. This is again due to the construction stage, which contributes 73% to the total. About a half of this is from compost (Figure 18), due to the emissions of methane (36%) and nitrous oxide (12%), produced during the composting process.

Burdens from preventive maintenance are the second largest contributor to GWP, accounting for 16% of the total CO_2 eq per functional unit (see Figure 17).The main contributor is the emissions of CO_2 from transport which accounts for about 10% of the total GWP.

5.1.3.7 Human toxicity potential (HTP)

The largest contribution to HTP of 0.67 kg DCB eq/fu is from the construction stage, which accounts for 53% (Figure 16) of the total. This is largely due to the life cycle of gravel (25%), with the majority (21%) coming from nickel, vanadium and chromium emitted during the gravel extraction.

Preventive maintenance is the second largest contributor, accounting for 35% of the total HTP (Figure 17). Emissions to air from transport are mainly responsible for this contribution (22%), including volatile organic compounds and chromium (VI).

5.1.3.8 Marine aquatic ecotoxicity potential (MAETP)

MAETP is estimated at 652 kg DCB eq/fu. Similar to the other impacts, emissions from the construction stage are the major contributor, accounting for 56% of the total (Figure 16). Gravel production within the construction stage is the major contributor, accounting for about 27% of the total MAETP (Figure 18). From this, the largest contributing burdens are emissions of vanadium (+III) and hydrogen fluoride, which are emitted during electricity production.

Activities conducted during the corrective maintenance stage are the second largest contributor, accounting for 29% of the total MAETP (See Figure 19). Within this, the largest contributing burdens (18%) are from transport of personnel, they include emissions to freshwater of beryllium, and hydrogen fluoride to air.

5.1.3.9 Ozone layer depletion potential (ODP)

Total ODP amounts to 0.219 mg R-11eq/fu, with the construction stage accounting for 43% of the total. Around 12% of this is from compost (Figure 18), mostly due to the emissions of halon 1301 to air, coming from transport activities.

Another 36% of the total ODP is due to the emissions from the preventive maintenance stage, mainly from the emissions of halogenated compounds from transport.

5.1.3.10 Photochemical ozone creation potential (POCP)

This impact is equal to 2.1 g ethane eq/fu. The construction stage contributes 52% and preventive maintenance 29% to the total POCP. Compost represents about 28% of the contribution from the construction stage (Figure 18), due to the emissions of methane, nitrogen oxides and carbon monoxide, which are produced during the composting process. Transport in the preventive maintenance stage contributes 18% to the total photochemical ozone creation potential; this is due to the emissions to air of non-methane VOCs, carbon monoxide and nitrogen oxides.

5.1.3.11 Terrestrial ecotoxicity potential (TETP)

Total TEPT is around 0.015 kg DCB eq/fu. The largest contributor is again the construction stage (75%), around half of which is from the gravel's life cycle (Figure 18) and particularly from the use of electricity within the gravel extraction and the related emissions to air of vanadium and mercury. Preventive maintenance is the second largest contributor, adding 17% of the total ODP, with half of that being from transport due to the emissions of chromium and mercury.

5.1.4 Sensitivity analysis

Sensitivity analysis has been carried out considering two parameters which could affect the design of the bio-retention unit as well as the environmental impacts: pollutant concentration in the inlet of the bio-retention unit, and volume of the treated runoff. Furthermore, the composting process has been included as part of the sensitivity analysis, because as shown in section 5.1.3, compost is the main contributor to AP, EP, GWP, ODP and POCP. Since compost is a basic element in the bio-retention unit design (see Section 2.4.1), it is not possible to analyse different quantities or substitution of this material. Hence, this analysis has been focussed on different production processes for compost. The results of the sensitivity analysis are shown as follows.

5.1.4.1 Analysis of the variation in the pollutant concentration

The pollutant concentration has been assumed as EMC, based on the mean values of the EMC range presented in Kayhanian et al. (2007). The effect of the EMC variation in the environmental impact is analysed in this section, since this strongly influences the accumulation of pollutants in the sediment which is disposed of within the corrective maintenance stage. Furthermore, EMC also influences the composition of the effluent runoff sent to the receiving water bodies.

Maximum and minimum EMC values have been labelled as SA and SB scenarios respectively. In addition to these values, a third (SC) case is analysed, corresponding to a six times increase in the maximum EMC value. This scenario has been included in order to analyse what would be the effect on the environmental impacts if EMC values were much higher than the maximum values reported in the literature, to account for potential extreme pollution situations. Finally, another scenario has been included in order to analyse the impacts from polluted runoff without any treatment. This scenario (SW) is included to compare the impacts with and without the treatment of the polluted runoff, to find out whether any trade-offs exist. The scenario without the treatment assumes maximum pollutant concentration as in SA in order to consider the worst case scenario. Information about the EMC used for each scenario is given in Table 28. The pollutant concentration affects only the toxicity-related impacts, however since the variation in TETP is negligible (SA = 0.015 kg DCB eq/fu; BC = 0.0149 kg DCB/fu; SB = 0.0149 kg DCB eq/ fu), only FAETP, HTP, and MAETP are considered below.

Table 28 EMC range considered for each scenario

Parameter (mg/l)	Max (SA)	Average (BC)[a]	Min (SB)	6 times increase on max value (SC)
Cd	0.03	0.007	0.002	0.18
Cr	0.09	0.009	0.001	0.54
Cu	0.27	0.034	0.001	1.62
Ni	0.13	0.011	0.001	0.78
Pb	2.60	0.047	0.001	15.6
Zn	1.68	0.187	0.005	10.1

[a] Base case: these values correspond to the average EMC values that have been used to calculate the pollutant loads as shown in equation 6.

5.1.4.1.1 Freshwater aquatic ecotoxicity potential

FAETP ranges from 0.39 kg DCB eq/fu for SA to 0.20 kg DCB eq/fu SB, due to the variation in the EMC from the maximum to minimum values, respectively. By comparison, this impact for the base case is 0.23 kg DCB eq/fu. This is quite close to the results for the minimum EMC value (SB), despite the large difference between the EMC concentrations. This can be explained by the fact that the difference between the pollutants loads from SB and BC is small. For the case of SA, the highest contributions are from the corrective maintenance and operation (residual runoff) stages. For corrective maintenance, the largest contributing burdens are from sludge disposal, accounting for 33% of its total FAETP, due to the emissions of nickel and zinc.

Comparison between the SA and the SC shows that increasing the maximum EMC value by six times increases the FAETP by 340% (see Figure 20) due to the higher contribution from the corrective maintenance stage. Compared to the BC this represents an increase of 650%. Finally, comparison between SA and SW indicates that the runoff treatment by a bio-retention unit can reduce the FAETP by 73%, compared to no treatment (Figure 20). This is because the untreated runoff stream carries heavy metals which have a high contribution to the FAETP. Thus, the implementation of a bio-retention unit helps in reducing this impact.

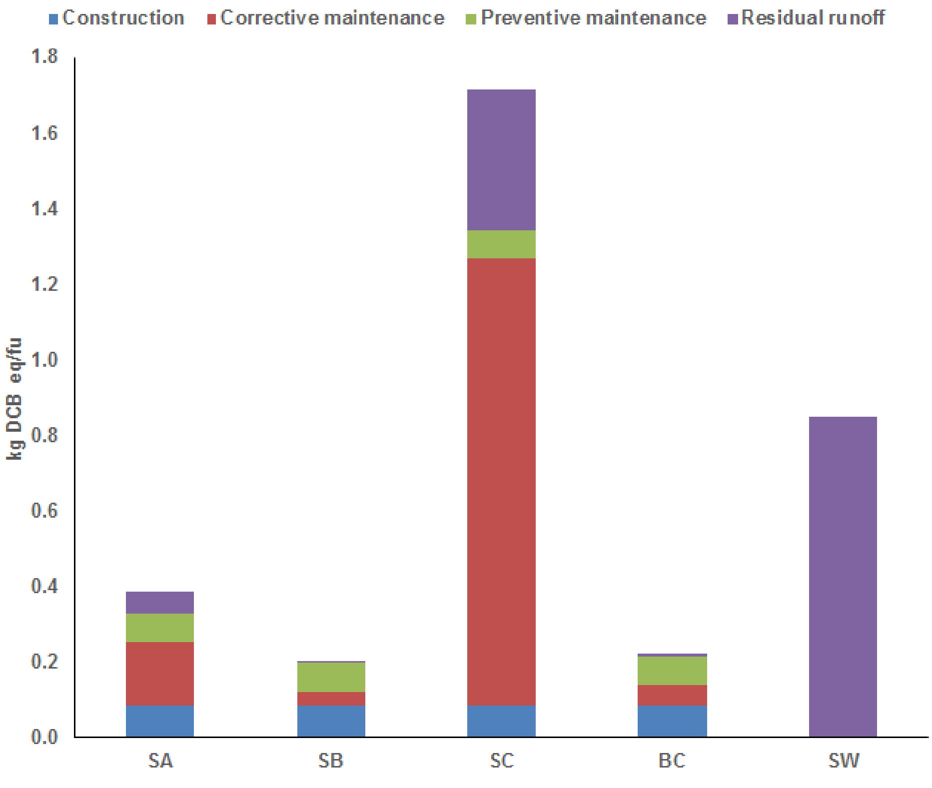

Figure 20 FAETP for different scenarios

5.1.4.1.2 Human toxicity potential

HTP variation between scenarios SA, SB and BC is small (see Figure 21). The main difference is in the contribution from the corrective and residual runoff stages. Among these, the highest HTP is for the SA, where the highest contribution comes from the construction stage, and in particular from the emissions to air of nickel and vanadium associated with gravel.

Comparison between the SC and SA shows that the HTP from the SC is 13% higher than HTP from the SA. As shown in Figure 21, this is due to the contribution from the corrective maintenance and the treatment stages.

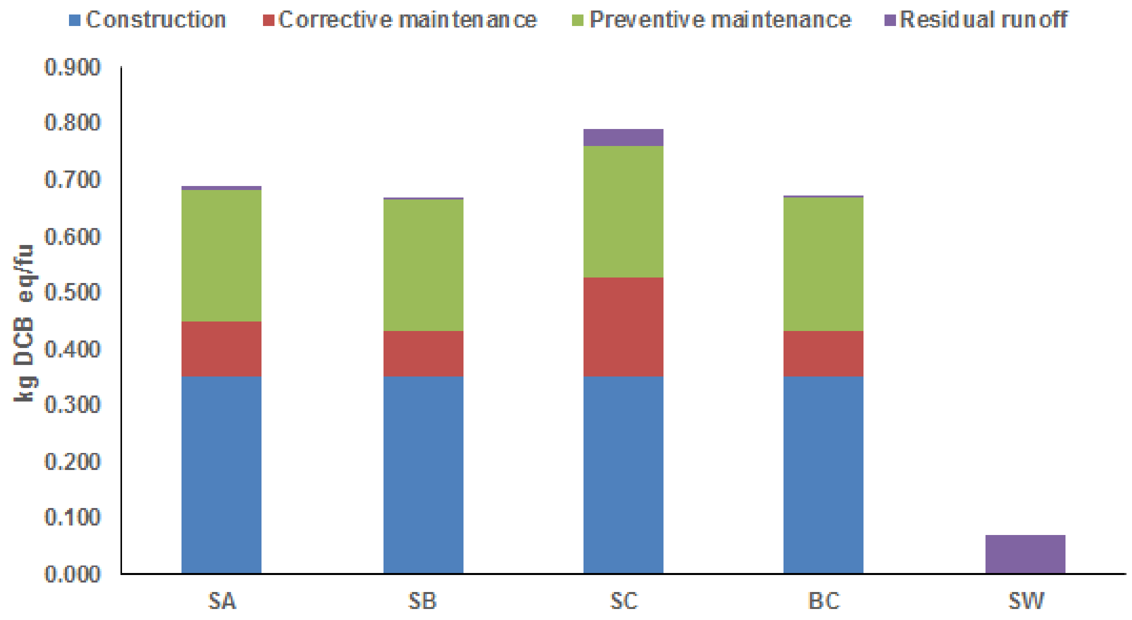

Figure 21 HTP for different scenarios

It is observed that HTP from the SW is 10 times lower than the HTP from the SA. This is due to the emissions in the construction stage, which are avoided if the runoff is left untreated.

5.1.4.1.3 <u>Marine aquatic ecotoxicity potential</u>

As shown in Figure 22, the MAETP ranges from 642 kg DCB eq/fu for SB to 1264 kg DCB eq/fu for SC. The latter is 42% higher than from the SA, largely due to the corrective and residual runoff stages, which are affected by the increase in the EMC.

As to the comparison between SA and SW, it can be observed in Figure 22 that the runoff treatment leads to a 113% increase in this impact. The reason for this is the construction stage and in particular gravel. Therefore, treating diffuse pollution by the bio-retention unit does not lead to the reduction of MAETP; in fact, it increases when compared to leaving the runoff untreated.

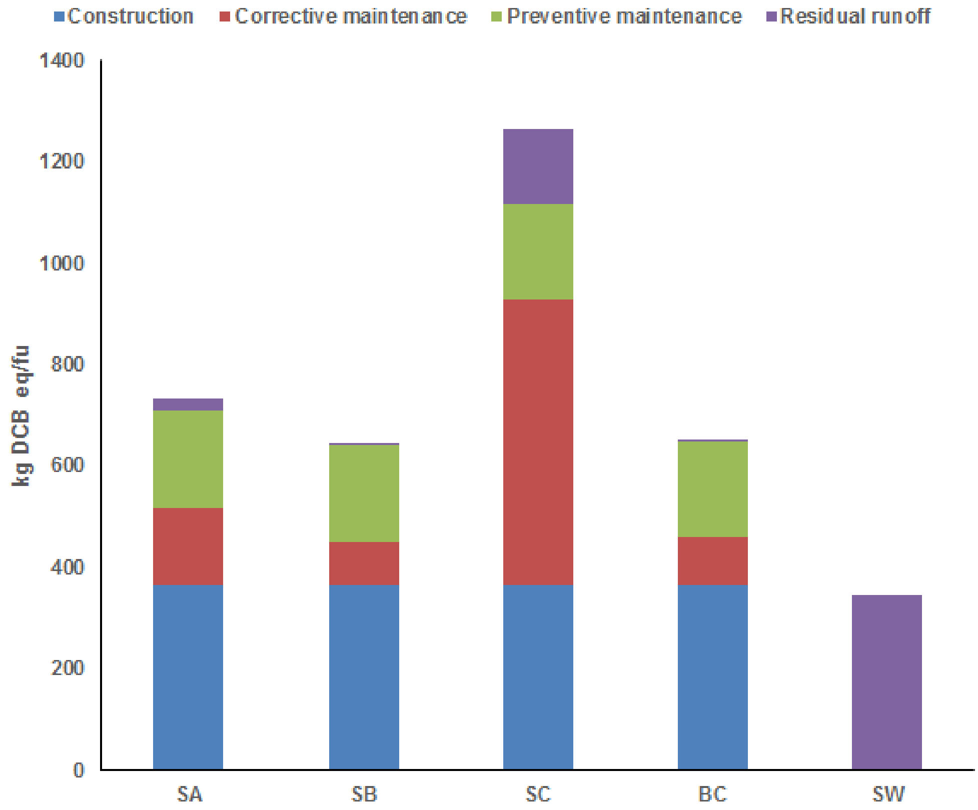

Figure 22 MAETP for different scenarios

5.1.4.2 Analysis of the variation in the treated runoff volume

The design of the bio-retention unit is based on a design storm, which varies in relation to the function to be provided by the bio-retention unit. As previously explained, since the aim of this study is to assess the control of diffuse water pollution, the design storm used was a 1-year-24 hour, as prescribed by the MDE construction guidelines (MDE, 2000). However, the rainfall depth associated with this event varies according to the geographical location. Since it was not possible to find information for the typical rainfall depth of a 1-year- 24-hour storm for the case of the Magdalena River catchment, a rainfall depth range is analysed as part of the sensitivity analysis. The rainfall depth considered in the base case was 25.4 mm. Therefore, in order to analyse the effect of increasing this, an increase of five times this rainfall depth has been considered (127mm) for the sensitivity analysis. Using the rainfall depth of 127 mm and the EMC values considered in the base case and scenarios SA and SC, the treatment volume is calculated using equations 4 and 5.Thus, three scenarios have been defined, whereby SD refers to maximum EMC values, SE refers to average EMC values and SF refers to minimum EMC values. The results from these scenarios are compared with their counterparts for the treatment depth of 25 mm assumed in the base case. The variation in the EMC mainly affects two impact categories: FAETP and MAETP. As to the cases with the highest pollutant concentration (scenarios SA and SD), a reduction in FAETP and MAETP of 20% and 47%, respectively, is found for the rainfall depth of 127 mm compared to 25.4 mm. Comparison of SE and the BC for the two different rainfall depths indicates that there is a reduction of 40% for FAETP and 54% for MAETP for the highest rainfall. Finally, for SB and SF, reductions of 40% for FAETP and 55% for MAETP are observed for 127 mm rainfall. The observed reductions in the impacts are because the increase in rainfall depth leads to an increase in the runoff volume to be treated. Thus, since the treatment volume has an inverse relationship with the impact produced, the calculated impact will be reduced as the treatment volume is increased.

5.1.4.3 Analysis of the compost production

In this study, compost is produced through open windrows using biogenic waste, which includes green waste coming from gardens (Nemecek & Kagi, 2007). The composting process includes the following stages: construction, operation and decommissioning of the treatment area, excluding the transport to the final user (Nemecek & Kagi, 2007). This process is considered as a base case scenario (BC). Additionally three scenarios have been defined, which correspond to each treatment technology considered. Scenario A refers to in-vessel tunnels with a curing phase in turned windrows in an enclosed building (Blanco et al., 2010), Scenario B refers to closed tunnels (Cadena et al., 2009) and Scenario C refers to confined windrows

(Cadena et al., 2009). A description of each compost technology is provided as follows.

The composting plant defined for SA treats 15,000 tonne of organic waste a year. Organic waste is defined as the organic fraction of municipal solid waste (OFMSW) (leftovers of raw fruit and vegetables, food scraps and raw fish or meat) and pruning waste (PW) (tree cuttings, branches, grass and wood) as a bulking agent. The decomposition processes take place in vessels, while the curing phase takes place in turned windrows in an enclosed building (Blanco et al., 2010). Gaseous emissions from both the composting tunnels and curing area are treated with bio filters, while leachate is reused as part of the composting process (Blanco et al., 2010). The system defined by this treatment technology encompasses six stages: transport of OFMSW and PW and kitchen bin production, building of the treatment facility and other infrastructure, diesel, electricity and water consumption, transport and management of solid waste fraction and building waste, gaseous emissions and leachate produced during the composting processes and transport of compost from the plan to the final user (Blanco et al., 2010).

The composting plant considered for SB treats 6,000 tonnes of OFMSW along with wood chips as a bulking agent (Cadena et al., 2009). The decomposition process occurs in closed tunnels and forced aerated windrows are used within the curing phase. System boundaries for this option include the composting process and its electricity and fuel requirements, excluding transport of OFMSW, compost and refuse to the wastewater treatment and final destination (Cadena et al., 2009). Finally, the SC has been modelled considering a composting plant that treats 91 tonne of OFMSW per year, using pruning waste as a bulking agent. In this plant decomposition occurs in confined windrows and the curing phase occurs in turned windrows (Cadena et al., 2009). This plant has the same system boundaries previously defined for the closed tunnels plant (Cadena et al., 2009).

The variation in the composting technology leads to an increase in all the impact categories for the scenarios under analysis. This is due to the emissions produced during each composting process. Thus, of all the impacts POCP shows the most significant increase for SC, which is 54 times higher than POCP for BC (2.06 g of ethane eq/fu). This is explained by the VOC emissions produced during the composting process, which account for 96% of the total 114 g ethane eq/fu for SC. As shown in Figure 23 after POCP, AP is the impact category that shows the biggest increase. Acidification potential from the BC (0.023 kg SO2 eq/fu) shows an increase of 16 times and 8 times, to about 0.4 and 0.22 kg SO2 eq/fu for SB and SC respectively. Ammonia emissions to air coming from the composting process represent the cause of this increase, since they vary from 6.4 g for the BC to 0.35 kg for SB and 0.18 kg for SC.

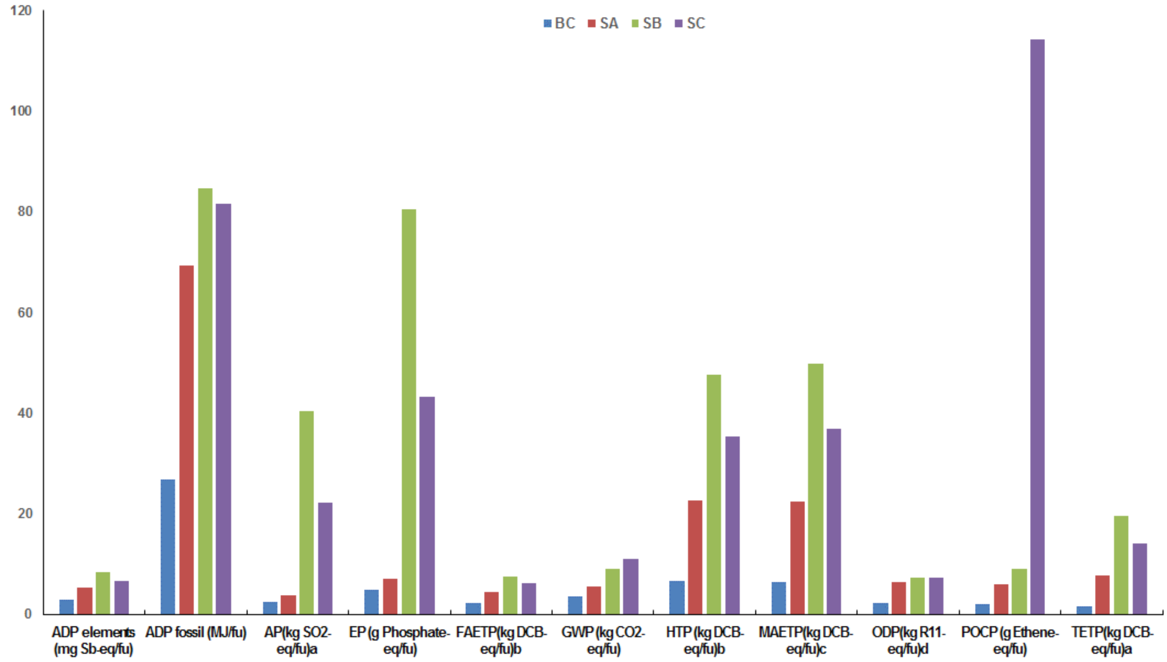

Figure 23 Results of the sensitivity analysis for compost
The values shown have been scaled by a) multiplying by 100; b) multiplying the original value by 10; c) dividing the original value by 100 and d) multiplying the original value by 1 x 10^7.

In addition to POCP and AP, EP is the impact category with the third biggest increase. As seen in Figure 23, eutrophication potential dramatically increases from 4.93 g phosphate eq/fu (BC) to 80.4 g phosphate eq/fu for SB and 43 g phosphate eq/fu for SC, respectively. This variation comes as a result of the increase in the ammonia emissions, which accounts for about 93% of the total EP in both cases. Finally, TETP from SB and SC increases by 12 times and 8 times in relation to BC, which amounts to 1.49 kg DCB eq/fu. This increase is mainly contributed by vanadium emissions to air, which constitute about 60% of TETP from SB (0.19 kg DCB eq/fu) and 55% of TETP from SC (0.14 kg DCB eq/fu). This contribution comes from electricity consumption. The increase for the remainder of the impact categories varies from 0.45 to 4.6 times higher than the BC.

5.1.5 Validation of the results

As already pointed out, there is a lack of studies that analyse the life cycle environmental impacts of the bio-retention unit. There are two previous studies, Kirk (2006) and Flynn and Traver (2012), whose characteristics are shown in Table 29, that have analysed the life cycle environmental impacts of the bio-retention unit. However, neither of these can be used to validate results presented in this study. This is because, as explained in Chapter 3, the drainage area should be uniform in order to make a fair comparison between the impacts obtained by Kirk (2006) and Flynn and Traver (2012). Both studies used different functional units than the one used in this study; Kirk (2006) considered management and treatment of storm water runoff from 0.4 ha of 100% impervious surface, while Flynn and Traver (2012) used management and treatment of runoff coming from an impervious area, without defining the area. As explained before, in this study the functional unit is considered as treatment of 1m^3 of runoff over 30 years, so in order to make a comparison of the results, information about runoff produced and treated by the bio-retention unit presented by Kirk (2006) and Flynn and Traver (2012) would be required to make the corresponding estimation in relation to the functional unit used in the present study.

Table 29 Comparison of the bio-retention unit LCA studies

Study	Functional unit	System boundaries
Kirk, 2006	Management and treatment of stormwater runoff from 0.4 ha of 100% impervious surface	Construction, corrective maintenance, preventive maintenance and decommissioning
Flynn and Traver 2012	Impact per impervious drainage area	Construction, operation and decommissioning

5.1.6 Summary

The results of this study indicate that bio-retention units can help reduce the freshwater aquatic toxicity, but at the same time they increase marine aquatic and human toxicity compared to leaving the runoff untreated for diffuse pollution. In addition, other environmental impacts are generated which would not otherwise have been produced. Therefore, trade-offs need to be made between improving the quality of the local urban environment and the other life cycle impacts generated elsewhere. These are difficult decisions that can only be made by the appropriate stakeholders.

The study also points to the life cycle stage which could be targeted for reducing the environmental impacts from this treatment option. This is the construction stage which is the cause of most impacts, contributing from 43% to ozone layer depletion to 83% to acidification. This is mainly due to the emissions to air generated in this stage. Maintenance is the next largest contributor, while the contribution from the decommissioning and the operation stages is negligible.

The results of sensitivity analysis show that pollutant concentration on the inlet of the unit affects the freshwater, marine and human toxicities, since the main pollutants considered are heavy metals so that the higher the pollution levels the higher the freshwater toxicity potential. For the same inlet concentrations of heavy metals, treatment of the runoff by the bio-retention unit can reduce the freshwater toxicity by about a half. The opposite is the case for marine eco-toxicity, which increases by 50% compared to leaving the runoff untreated. Therefore, the concentration of heavy metals in the runoff is an important parameter which should be considered carefully in decisions related to bio-retention units.

The variation in the rainfall depth also affects the environmental impacts associated with the bio-retention unit. An increase in the treatment volume leads to a decrease in the environmental impacts, ranging from 12% for acidification to 62% for terrestrial eco-toxicity. The variation in the composting technology leads to an increase in all the impact categories, and this increase is generated by the emissions produced during the composting process. Therefore, based on the results, the open windrow composting process is the process with less environmental impact among the technologies considered.

5.2 Economic evaluation

5.2.1 Goal and scope

The goal of the economic evaluation is to quantify the life cycle economic costs of the bio-retention unit and identify the hot spots. The system and system boundaries are the same as in the LCA study (see Figure 15). In the same way, the functional unit is the same as in the LCA study, i.e. “treating 1 m^3 of runoff over 30 years” (see section 5.1.2.1).

5.2.2 Inventory analysis

5.2.2.1 Assumptions

Economic information is usually a key parameter in the selection process of a treatment practice such as the bio-retention unit. In this study the calculation of the life cycle costs is based on the environmental life cycle approach defined by Swarr et al. (2011). This approach has been selected as it is congruent with the LCA approach.

The stages considered in this LCC study are construction, corrective and preventive maintenance and decommissioning. The operation (residual runoff) stage is not taken into account in this analysis, because it only involves passing the runoff through the bio-retention unit, which does not incur any costs. According to Swarr et al. (2011), there is no consensus on the elements that should be considered in the cost estimation within the life cycle of the system. Therefore, in this study the following costs are included:

- capital costs in the construction stage, including material costs, their transport and costs of material handling.
- costs of preventive maintenance, including the costs of inspection, mowing and transport of mowed material.
- costs of corrective maintenance, comprising the costs of inspection, sludge removal, transport and disposal.
- costs of decommissioning, including the costs of machinery used for removing the construction materials and putting in new top soil.

The costs have been estimated by multiplying material requirements previously calculated in the life cycle inventory of the LCA study (see from Table 24 to Table 27) by the cost of each material, which as explained below has been sourced from different information sources. All the costs are in American dollars to facilitate their comparison with related studies. Data sources are discussed below.

5.2.2.2 Data sources

Cost data and their sources are summarised in Table 30 and Table 31. The costs related to the operation of the costs of materials (except compost), machinery and labour have been sourced from a Mexican construction cost database generated by the Mexican Institute of Cost Engineering (Gonzalez, 2012). The cost of compost has been sourced from Rodriguez-Salinas & Cordova-Vazquez (2006). The costs of sludge disposal are based on typical costs of disposing a skip with a capacity of 8 cubic yards. As no Mexican data were available, these data are sourced from the UK (Wills, 2011). Uncertainties derived from either labour or material cost variation within time and between countries have not been estimated.

Table 30 Cost of the construction stage

Construction stage	
Material installation cost	
Activity	Cost (US$)
Trench excavation	884
Soil installation	442
Gravel installation	294
Compost installation	143
Geotextile and PVC installation	14
Material cost	
Material	Cost (US$)
Gravel	654
Sand	786
Top soil	85
Compost	1199
Geotextile membrane	25.74
PVC pipe	45
Transport cost	
Material transport	Cost (US$)
Gravel	162
Sand	195
Compost	11
Top soil	24
Geotextile and PVC pipe	8

Table 31 Cost per life cycle stage

Corrective maintenance stage	
Activity	Cost (US$)
Inspection	645
Sludge disposal	198
Sludge removal	23360
Sludge transport	12857
Preventive maintenance	
Activity	Cost (US$)
Mowing	3967
Inspection	645
Transport of mowed material	1934
Preventive maintenance	
Activity	Cost (US$)
Mowing	3900
Transport	644
Transport due to mowing	1932
Decommissioning	
Activity	Cost (US$)
Material removal	145
Soil installation	145

5.2.3 Life cycle costs

As shown in Figure 24, the total life cycle costs of the bio-retention unit are equal to US$ 17 per 1 m^3 of the treated runoff. Over the life time of the bio-retention unit, this amounts to US$ 49,000. The highest contribution is from the corrective maintenance stage (76%), largely due to the costs of machinery (63%), which are due to the fuel costs and the salaries of the personnel operating the machinery. It should be noted that salaries considered are representative of Mexico City so that these costs may be different in different locations in the country.

Preventive maintenance contributes 13%, mainly because of the cost of mowing (69%) and transport of the mowed material (29%); see Figure 25. Finally, the construction stage contributes 10.5% of the total costs, mainly due to the material costs. The costs of decommissioning are small, adding 0.59% to the total LCC.

These results show that if the economic evaluation of the bio-retention unit was conducted using just the capital costs, this would represent an underestimate of around 90% of the actual LCC. This highlights the importance of considering the life cycle approach in the selection of runoff treatment options.

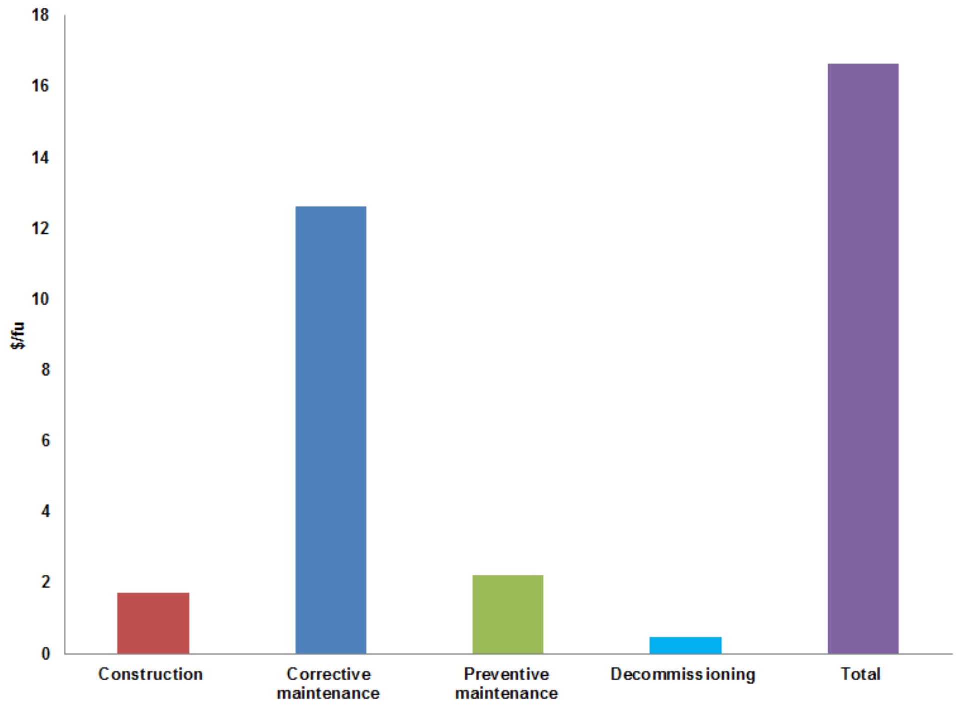

Figure 24 LCC of the bio-retention unit

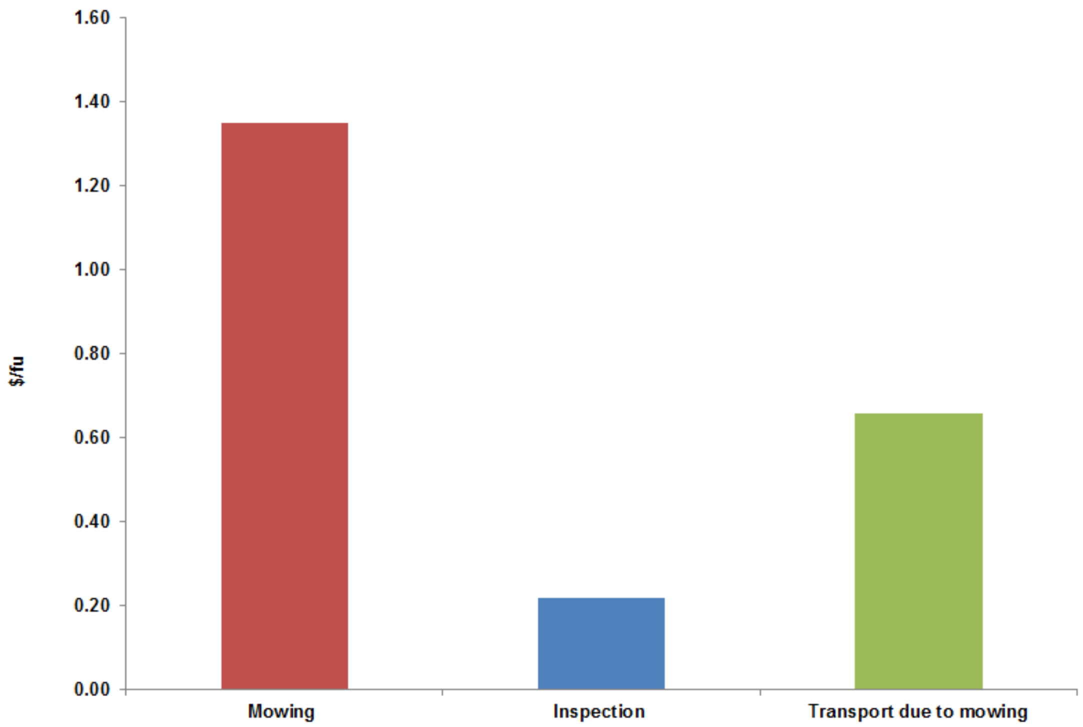

Figure 25 Contribution from preventive maintenance to LCC

5.2.4 Sensitivity analysis

As mentioned in the previous section, the contribution from the sludge removal turns the corrective maintenance into the main LCC hot spot. Based on the pollutant load model used for calculating the sludge accumulation, both concentration and catchment area are fixed parameters; however, the amount of treatment volume could vary in relation to the geographical area. In order to cover the impact of this variation in the LCC, this parameter is increased up to five times, i.e. from 25.4 mm to 127 mm (as in the LCA study). The results from this variation are labelled as scenario SL (large volume).The variation of the EMC concentrations, considered in the sensitivity analysis of the LCA study, is not considered here, since it does not lead to any variation in the LCC.

The results show that increasing the treatment volume by five times compared to the base case, leads to only a modest reduction of the costs: from \$17 to \$15 per functional unit. As shown in Figure 26, corrective maintenance remains the main contributor to the LCC (88%). The contribution from preventive maintenance is 10% and construction 9%; the contribution from decommissioning is negligible.

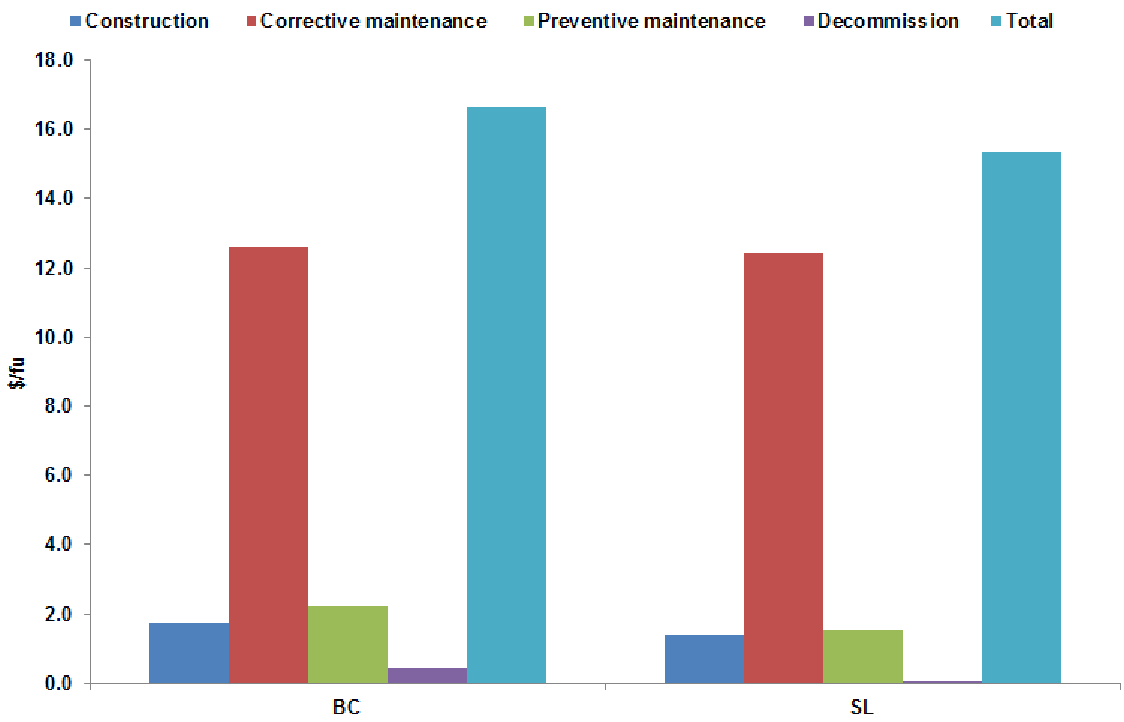

Figure 26 Comparison of LCC for the base case (BC, 25.4 mm) and the high rainfall dept (SL, 127 mm)

This study did not consider net present value (NPV) of the bio-retention unit. To examine how the costs might change if NPV was considered, different discount rates are used to estimate the NPV as part of the sensitivity analysis. A range from 5.5%, a typical value for the U.S. and UK (Lampe et al. 2005) to 12%, a typical rate for Mexico (SHCP, 2009), is considered and the results are shown in Figure 27. As indicated, the higher the discounts rate the lower the LCC, thereby showing that the variation in the discount rate should be considered carefully in the LCC estimations. For example, the NPV considering a 12% discount rate is 48% lower than the NPV calculated with a 5.5% discount rate. In addition, the LCC of the base case (US$ 49,000) is 79% lower than the NPV obtained with a 12% discount rate and 61% lower than the NPV obtained with a 5.5% discount rate.

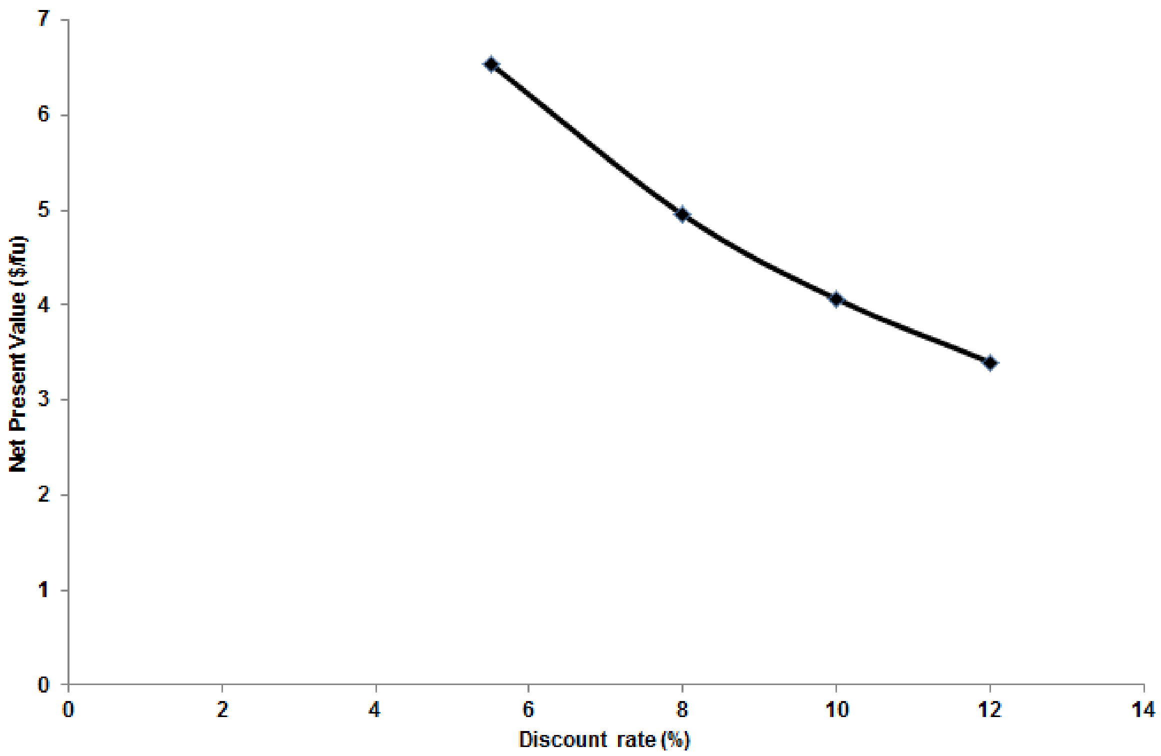

Figure 27 Variation of LCC due to different discount rates

5.2.5 Validation of the results

Life cycle cost (LCC) information for the bio-retention units is scarce, with only two other studies carried out (Lampe et al. 2005 and Lloyd et al. 2002). This is due to two factors. First the bio-retention unit is a relatively new runoff treatment option, therefore cost information is hard to collect (Lampe et al. 2005).The second factor is that this information is dispersed among the different parties in charge of design and maintenance of the bio-retention units (such as local government, local water authorities, community or private owners) (Lloyd et al., 2002, cited in Taylor, 2005).Therefore, it has only been possible to compare the results obtained in this study with the results of annual maintenance and capital costs available in the literature.

As shown in Table 32 the results from this study are in the same order of magnitude as the other published results. The difference in the costs can be explained by different assumptions as well as exchange rates used. For example, the preventive maintenance costs will depend on the frequency of maintenance activities. However, it is not possible to compare the frequencies assumed in the current study and those by Lampe et al. (2005) as they do not state this; their costs are presented as the American and British cost of a "typical" bio-retention unit with no definition of "typical".

As to the comparison between the costs of annual corrective maintenance, the higher value in the current study could be due to the high volume of sediment removed. Even though the amount of sediment collected is not mentioned by Lampe et al. (2005), it is mentioned that the lack of pre-treatment options leads to high sediment volumes and therefore high costs. Since in this study this kind of pre-treatment practice is not considered, sediment accumulated might be the reason for the high cost. As for the comparison between the preventive maintenance results, it is not possible to ensure that sediment is the cause of the high cost because Lampe et al. (2005) do not present data about the amount of sediment removed.

A further comparison is made with the results obtained by Lloyd et al. (2002), who calculate the annual maintenance cost based on the annual amount of 25 m^3 of sediment collected. As shown in Table 32 the difference between the results from this study and Lloyd et al. (2002) is quite high: $1,506 compared to $5,160, respectively. This variation might be attributed to the difference in the design, since in the design considered by Lloyd et al. (2002); there are sediment and litter traps, which are not considered in the present study.

Table 32 Cost comparison for the bio-retention unit

Author	Annual preventive maintenance	Annual corrective maintenance	Total annual maintenance cost	Capital cost
Lampe et al. (2005) US ($)	530	480	1010	NA*
Lampe et al. (2005) UK ($)	1336	636	1972	NA
Lloyd et al. (2002) Australia ($)	NA	NA	5160**	10320**
Present study ($)	269	1237	1506	20389

*NA: Not available
**Calculated considering low maintenance cost relative to volume of trapped material

The calculated capital cost varies based on the design, which determines both the type and quantities of the required materials. There is, therefore, no definitive figure for capital cost of the bio-retention unit. Thus there are different approaches for carrying out this calculation. For example, there is a rule of thumb presented by the low impact development centre (Houdeshel et al., 2011), which points out that the capital cost for a bio-retention unit built in an institutional, commercial or industrial area is $107-430-40 per square meter. In addition, there is an estimator developed by (Brown & Schueler, 1997, cited in Fletcher et al., 2005) and adapted by the EPA (1999a) that establishes a relationship between the capital cost and the treatment volume. In this estimation, the capital cost is calculated as $189/m^3. Finally, Lloyd et al. (2002) show capital cost estimates based on the relationship based on the catchment area. Using the approaches described above, the capital cost has been calculated with the characteristics of the bio-retention unit considered in this study; both data used and the results are shown Table 33.

As indicated in the table, the capital cost calculated using the Houdeshel et al. (2011) and EPA (1999) approaches and the method used in the present study are in the same order of magnitude. The variation depends on the design, since the parameters used (both treatment volume and built area) depend on the design criteria. On the other hand, the difference between results from this study and the cost estimated by Lloyd et al. (2002) depends on the materials considered, since in the current study compost is considered in addition to the materials considered by Lloyd et al. (2002).

Table 33 Data and calculated capital cost

Parameter of the bio-retention unit used for capital cost calculation	Value	Author	Capital cost calculated
Built area*	128 m^2	(Houdeshel et al., 2011)	$13,913-$55,652
Treatment volume**	98 m^3	Adapted from Brown and Schueler in (EPA, 1999)	$18,277
Catchment area	0.4 ha	Lloyd et al. (2002)	$5,000-$10,000
Treatment volume**	98 m^3	Current study (adapted from Swarr et al. (2011)	$19,933

*This area refers to the built area of the bio-retention unit

**This treatment volume is calculated as WQv according to the eq.4.

5.2.6 Summary

The LCC of the bio-retention unit are calculated at $17 per m^3 of runoff treated or US$ 49,000 over the life time of the bio-retention unit. The costs are dominated by the corrective maintenance stage (76%). This is due to the costs of machinery for sludge removal (63%). Increasing the treatment volume by five times reduces the LCC by 20%.

The results are comparable with published studies of annual maintenance cost and capital cost. It is found that the difference between the LCC results in different studies depends on three main factors: frequency of maintenance activities, the amount of collected sediment, and the design considered for the bio-retention unit. Therefore, these factors must be considered carefully when evaluating the LCC costs of bio-retention units. The next chapter evaluates both the environmental and economic costs of the infiltration trench.

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6 Infiltration trench: Environmental and economic evaluation

Infiltration trenches are a treatment practice considered for controlling diffuse water pollution in urban areas. This chapter is dedicated to the analysis of the environmental and economic impacts of an infiltration trench, whose characteristics are described in Chapter 2.

6.1 Environmental evaluation of an infiltration trench

6.1.1 Goal and scope

The goal of this study is to quantify the environmental impacts and identify the hot spots along the life cycle of an infiltration trench. The scope of this analysis is from 'cradle to grave', since the stages considered along the infiltration trench life cycle are: construction, corrective and preventive maintenance, operation (runoff treatment) and decommissioning (see Figure 28). Although infiltration trenches are rarely removed from the site where they are installed, decommissioning has been included as part of the life cycle in order to analyse its contribution to the environmental impacts.

The infiltration trench is a treatment option used for controlling diffuse water pollution (EPA, 1999b). As explained in Chapter 2, an infiltration trench is defined as a stone filled reservoir for runoff storage (Kuo et al., 1989;Lampe et al., 2005). The stone bed acts as a filter where pollutants are caught in the accumulated sludge. After collecting pollutants in the sludge, treated runoff passes through the stone reservoir from where it slowly infiltrates to the subsoil (EPA, 1999).

In order to avoid clogging of the infiltration trench, the accumulated sludge is removed and disposed in a landfill. The environmental impacts of this activity represent a major concern of the use of this treatment practice (Lampe et al., 2005). Hence, that sludge disposal has been included within the system boundaries.

On the other hand, in order to reduce the amount of accumulated sludge, pre-treatment practices are usually installed. However, they are not considered within the system boundaries, because this study is aimed at calculating the environmental impacts of the infiltration trench on an individual basis, so that these impacts can be compared with the ones from other treatment practices on the same basis.

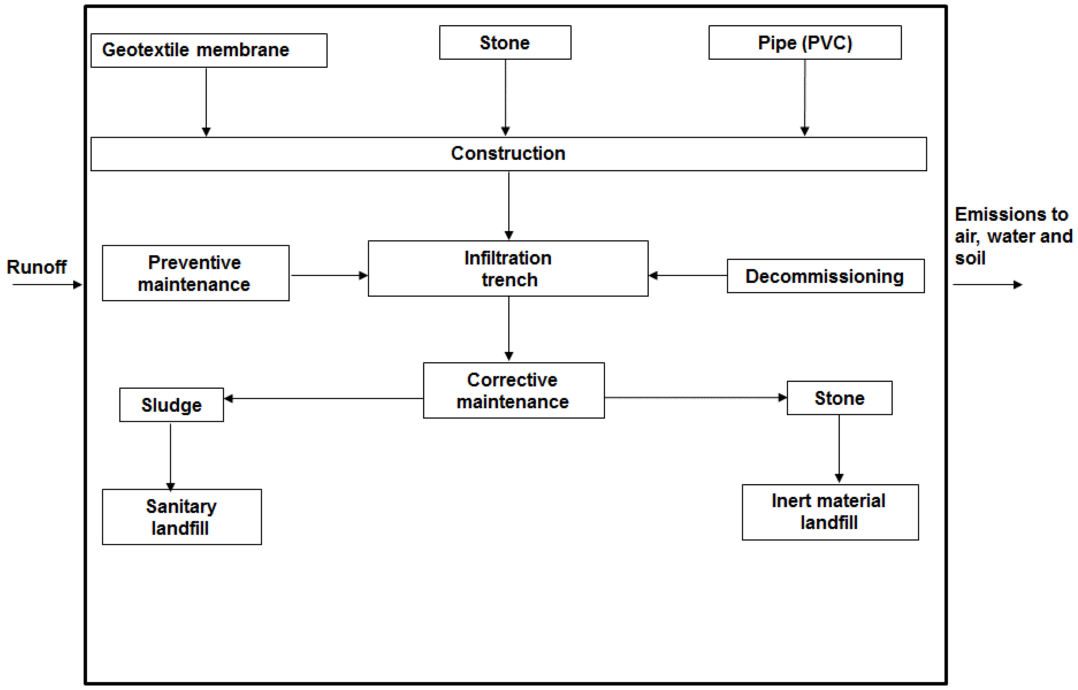


Figure 28 Infiltration trench life cycle

An average life time for the infiltration trench is not well defined. However, based on the information provided by other authors such as Lampe et al. (2005) and Kirk (2006), an average lifetime of 30 years has been considered for the purpose of this study. As to the considered functional unit, see Chapter 4 for its definition.

As explained in Chapter 4 the location of the infiltration trench has been assumed as the Magdalena river catchment in Mexico City.

6.1.2 Inventory analysis

The inventory is divided into the five stages considered along the infiltration trench life cycle (construction, corrective and preventive maintenance, residual runoff and decommissioning). Both data sources and assumptions considered within the inventory for each stage are detailed in the sections below.

6.1.2.1 Data sources

Data for the construction stage is not available for the case of Mexico City, because structural treatment practices have not been built, which is because controlling diffuse water pollution is at a preliminary stage in the country. As explained in section 6.1.2.2, this information is sourced from the Maryland construction guide (MDE, 2000). This guide establishes a set of equations to size the infiltration trench, which are based on the runoff volume to be treated. Thus, materials required during the construction stage are calculated based on the calculated treatment volume. Data sources for carrying out the infiltration trench sizing are shown in Table 34. Furthermore, the life cycle inventory data for the construction stage, which include trench excavation, stone spread, stone and stone transport, have been sourced from the Ecoinvent (Ecoinvent, 2011) and GaBi (PE International, 2010) databases. A description of the background processes considered within the construction stage, as well as the corresponding data source, is provided in Table 35. In order to adapt these requirements to Mexican conditions, the Mexican electricity mix (Santoyo-Castelazo et al., 2011) has been used in all the processes with electricity requirements along the construction stage.

Table 34 Data sources and assumptions for the design of the infiltration trench

(i) Information sources for runoff treatment volume		
Parameter	*Value*	*Data source*
Annual rainfall depth (P)	25.4 mm	(MDE, 2000)
(ii) Information sources for sizing the infiltration trench		
Parameter	*Value*	*Data source*
Infiltration rate	0.0132 m/hour	(EPA, 1999)
Stone porosity	0.4	(EPA, 1999)
Storage time	2 hours	(EPA, 1999)
Infiltration trench depth	1.2 m	(EPA, 1999)

Table 35 Data sources and background processes considered for the construction stage

Material		
Material	*Considered processes*	*Data source*
Stone	Whole manufacturing processes, machinery and transport	(Kellenberger et al., 2007)
Geotextile membrane	Whole manufacturing process of geotextile membrane	(PE International, 2010)
PVC	Whole PVC production process	(PE International, 2010)
Transport		
Vehicle	*Considered processes*	*Data source*
40 tonne truck	Construction, use (including maintenance) and end of life	(Spielmann et al., 2007)
Machinery		
Machinery	*Considered processes*	*Data source*
Hydraulic digger	Materials, transport of the parts to the assembly point and energy and heat requirements during use phase	(Kellenberger et al., 2007)

Pollutant mass entering the infiltration trench is calculated based on the pollutant loads. Computation of this load is based on the simple method presented by Chandler (1994). Information requirements of this method are: pollutant concentration, annual rainfall, percentage of events that produce runoff and catchment area (see Chapter 4). As explained in Chapter 4, from this data pollutant concentration is considered as event mean concentration (EMC), which is sourced from Kayhanian et al. (2007). Information about annual rainfall, percentage of events and catchment area is sourced from the Maryland guide (MDE, 2000).

Both accumulated pollutant mass and pollutant mass leaving the infiltration trench are calculated based on a mass balance. Information required for conducting this mass balance is the pollutant removal efficiency for each parameter, which as shown in Table 36 is sourced from EPA (1999).

Table 36 Infiltration trench removal efficiencies with information from (EPA, 1999)

Parameter	% removal
TSS	90
Cd	90
Cr	90
Cu	90
Ni	90
Pb	90
Zn	90

In the maintenance stages, information requirements encompass the activities to be considered as maintenance and the frequency with which they are carried out. For corrective maintenance, these activities are: stone removal, geotextile, PCV replacement and new stone implementation. As to the activities considered as part of the preventive maintenance, they are: debris removal and activities aimed at inspecting the conditions of the infiltration trench. An explanation of the selection of these activities is given in section 6.1.2.2. In both cases information has been sourced from Lampe et al. (2005), since this guide summarizes common practice for conducting maintenance in installed treatment practices in the UK and U.S.

Life cycle data on the activities considered in the maintenance stages have been sourced from both Ecoinvent (Ecoinvent, 2011) and Gabi (PE International, 2010) databases. A description of these processes considered for each activity of the maintenance stages, as well as the corresponding data source, is provided in Table 37.

As explained in section 6.1.2.2, sediment removed during corrective maintenance stage is landfilled in a sanitary landfill, life cycle information for this has been sourced from Doka (2009).

Table 37 Data sources and considered processes for the corrective and preventive maintenance stages

Corrective maintenance stage		
Machinery	*Included processes*	*Data source*
Hydraulic digger	Materials, transport of the parts to the assembly point and energy and heat requirements during use phase.	(Kellenberger et al., 2007)
Landfill	*Included Processes*	*Data source*
Sanitary landfill	Construction, waste water treatment and municipal waste incineration	(Doka, 2009)
Preventive maintenance stage		
Machinery	Included processes	Data source
Hydraulic digger	Materials, transport of the parts to the assembly point and energy and heat requirements during use phase.	(Kellenberger et al., 2007)
Vehicle	Included processes	Data source
Passenger car	Vehicle manufacturing, operation and disposal	(Spielmann et al., 2007)

6.1.2.2 Assumptions

Although the installation of infiltration trenches might be a potential solution for controlling diffuse water pollution, a drawback of such implementation is the lack of a standard guide for the construction, operation and maintenance of this treatment practice (Kuo et al. 1989). Therefore, in order to conduct this environmental assessment, a series of assumptions has been made. These assumptions are presented by life cycle stage in this section.

The construction stage is preceded by the design stage, where infiltration size is calculated based on the treatment volume. Since no construction guide for infiltration trenches exists for the case of Mexico City, as explained in Chapter 4 the design is based on the construction guide from Maryland Storm water design manual (MDE, 2000). Based on the MDE (2000) guide the design depends on the size of the treatment practice, which in turn is calculated according to the treatment volume. This volume is defined based on the function given by the function by the infiltration trench, which is considered as runoff treatment. The estimation procedure for calculating the treatment volume (or water quality volume WQv) is shown in Chapter 4.

After estimating the WQv, which has been estimated as 98 m^3, the surface area of the infiltration trench is calculated based on this parameter. According to the MDE (2000) guide, this area depends not only on the treatment volume but also on the maximum depth, stone porosity, infiltration rate and maximum allowable pounding time. These parameters are related as shown in Equation 9:

$$Ap = Vw / (d_{max}\, n) + (fT) \tag{9}$$

where:

Ap = surface area (m^2)
Vw = water quality volume (m^3)
n = stone media porosity (percentage (0-1)
dmax = maximum allowable depth of the trench (m)
f = infiltration rate (m/hour)
T = maximum allowable pounding time (hour)

Maximum depth of the infiltration trench is calculated based on a test of the geological characteristics of the installation area (MDE, 2000). However, since this data is not available for the installation area, a typical depth of 1.2 m is assumed (EPA, 1999). The stone porosity has been assumed as 0.4, which is the typical value reported for the infiltration trench in the EPA (1999) data sheet.

In the absence of local data for the infiltration rate, this value has been assumed as the infiltration rate corresponding to moderate infiltration rate soils, amounting to 0.013 m/hour (MDE, 2000). This value has been considered as appropriate since this is the prevailing kind of soil in the Magdalena river catchment (GDF, 2012). Finally, the storage time has been assumed as 2 hours, based on the guidance provided in the EPA (1999) infiltration trench data sheet. Considering these data, surface area is calculated according to Equation 9 as follows:

$Ap = 98\ m^3/((1.2\ m \times 0.4)+(2\ hours \times 0.013\ m/hour))$

$Ap = 193\ m^2$

The set of assumptions for the residual runoff stage are related to the calculation of the pollutant load. As mentioned above, pollutant load is calculated based on the simple method (Chandler, 1994). See Chapter 4 for the estimation procedure. To know the pollutant mass that leaves the infiltration trench, a mass balance is calculated. This mass balance is carried out under the assumption that neither chemical nor physical reactions occur during the time that runoff is stored.

Assumptions for the corrective and preventive maintenance stages encompass the kind and frequencies of the activities considered for each kind of maintenance. Although these activities might vary from place to place based on the local characteristics, they and their frequencies have been defined assuming typical activities defined by Lampe et al. (2005) (see Table 38).

Table 38 Information source for maintenance activities

Corrective maintenance stage	
Activity	*Frequency*
Sediment removal	Once every five years
Stone removal	Once every five years
Geotextile, PVC pipe replacement	Once every five years
Clean stone implementation	Once every five years
Preventive maintenance stage	
Activity	*Frequency*
Debris and trash removal	Once per year
Inspection activities	Once per year

As mentioned above, pollutant accumulated in the sludge is a major concern in regard to the corrective maintenance. Since it has been assumed that the composition of the influent is made up of heavy metals, these are the pollutants accumulated in the sludge. The accumulated sludge is assumed to be disposed in a sanitary landfill, since it has been considered that metals contained in the sludge do not pass the threshold (see Table 39) established by the Hazardous Waste Directive as cited in Lampe et al. (2005). This guideline has been used in the absence of a Mexican regulation for the classification of hazardous waste.

In addition to sludge disposal, stone disposal is also a key activity considered within the corrective maintenance. Due to the lack of life cycle data for stone, this information has been assumed as life cycle data for gravel (Kellenberger et al., 2007). Since it is assumed that pollutants are accumulated in the sludge, stone is treated as rubble which is disposed in an inert material landfill (Doka, 2009).

Table 39 Maximum concentration values in sediment from infiltration trench taken from (Lampe et al. 2005)

	Maximum concentration (mg/kg)					
Treatment practice	*Cd*	*Cr*	*Cu*	*Ni*	*Pb*	*Zn*
Infiltration trench	2.34	2.88	11.2	3.68	15.7	62.6
EU threshold value for hazardous waste	30,000	2,500	N/A	2,500	N/A	1,000

Finally, for the decommissioning stage, it has been assumed that the infiltration trench is removed only once after completing its life span. The restoration process is considered as filling the infiltration trench's space with top soil. Due to the lack of information, all transport distances have been assumed as 100 km.

A summary of the parameters and inventory considered within each life cycle stage of the infiltration trench is given from Table 40 to Table 45. The results of the impact assessment, as well as the results of the sensitivity analysis, which has been conducted to analyse some of the uncertainties, are shown in section 6.1.3.

Table 40 Summary of the assumptions for the design of the infiltration trench

(iii) WQv		(iv) SA	
Parameter	*Value*	*Parameter*	*Value*
Annual rainfall depth	0.0254 m	WQv	98 m^3
Catchment area	4047 m^2	Infiltration trench depth	1.2 m
Runoff coefficient	0.95	Stone porosity	0.4
		Infiltration rate	0.013 m/hour
		Storage time	2 hours

Table 41 Inventory for the construction stage

Material requirements		
Parameters	*value*	*unit*
Stone	231	m^3
Geotextile membrane	701	m^2
PVC pipe + PVC cape	1	m
Transport activities		
Parameters	*mass (ton)*	*ton*km*
Stone transport	371	37100
PVC pipe + PVC cape	0.0063	0.63
Geotextile membrane + PVC	0.721	72.1

Table 42 Inventory for the preventive maintenance stage

Preventive maintenance stage		
Activity	Frequency	Quantity
Inspection activities	once per year	3000 pkm[a]
Transport for debris and trash removal	once per year	3000 pkm[a]

The value shown has been calculated as: a) 100km*1person*30trips

Table 43 Inventory for the corrective maintenance stage

Corrective maintenance stage		
Activity	Frequency	Quantity
Sediment removal (stone removal)	once every five years	371 tonne
Sediment transport (stone removal)	once every five years	18534 ton*km
Stone	once every five years	371 tonne
Geotextile membrane	once every five years	0.72 tonne
PVC pipe + PVC cape	once every five years	0.0063 tonne
Clean stone implementation	once every five years	231 m^3
dirty stone removal	once every five years	231 m^3
Sediment composition		
Parameters	kg accumulated over 5 years	kg pollutant /kg sediment (TSS)[a]
TSS	1179	
Cd	2.76E-03	2.34E-06
Cr	3.40E-03	2.88E-06
Cu	1.32E-02	1.12E-05
Ni	4.34E-03	3.68E-06
Pb	1.86E-02	1.57E-05
Zn	7.38E-02	6.26E-05

[a] These values are shown per kg of sediment as guidance to estimate pollutant mass when sediment mass removed from the infiltration trench is known

Table 44 Inventory for the runoff treatment stage

Pollutant load in[a]	Mass (kg/year)
TSS	262
Cd	2.63E-03
Cr	8.25E-03
Cu	2.37E-02
Ni	1.14E-02
Pb	2.28E-01
Zn	1.47E-01
Pollutant load out	**Mass (kg/year)**
TSS	26
Cd	2.63E-04
Cr	8.25E-04
Cu	2.37E-03
Ni	1.14E-03
Pb	2.28E-02
Zn	1.47E-02
Pollutant load accumulated	**Pollutant accumulated**
TSS	236
Cd	2.37E-03
Cr	7.42E-03
Cu	2.13E-02
Ni	1.03E-02
Pb	2.05E-01
Zn	1.33E-01

[a] The pollutant loads on the inlet of the infiltration trench have been calculated based on the average EMC values from Kayhanian et al. (2007)

Table 45 Inventory considered for the decommissioning stage

Material removal		
Material	*Value*	*Unit*
Stone	231	m^3
Geotextile membrane	701	m^2
PVC pipe	1.2	M
Material transport		
Material	tonne	tonne *km
Stone	371	37100
Geotextile membrane	0.721	72.1
PVC pipe +cape	0.0063	0.63
Top soil	277	27700
Material implementation		
Material	*Value*	*Unit*
Top soil	277	Tonne

6.1.3 Impact assessment

The LCA methodology used in this work is according to the ISO standard guide 14040-14044 (ISO, 2006a; ISO, 2006b). The tool used to conduct the calculation of the environmental impacts is GaBi software version 4.4 (PE, 2010) considering the CML 2001 impacts characterisation method (Guinee et al., 2001). An analysis of the causes of these hot spots is given in this section.

6.1.3.1 Abiotic depletion potential (elements)

Total ADP (elements) amounts to about 13 mg Sb eq./fu, of which 85% comes from the corrective maintenance stage (Figure 29). Around half of this is due to the stone production (47%), while another half (49%) comes from the stone disposal (see Figure 30).

The remaining 15% of the contribution to the total ADP of elements comes from the construction stage. Again, most of this contribution (87%) is due to the stone processes, since, as seen in Figure 31, about 40% of this is produced by the stone extraction.

6.1.3.2 Abiotic depletion potential (fossil)

Most of the contribution to the total ADP of 340 MJ/fu comes from the corrective maintenance stage (86%) (see Figure 30). From this, the activities that contribute the most are landfilling of stone (46%) and geotextile membrane production (38%). About 80% of the stone disposal is due to building the landfill facility.

As seen in Figure 31, the contribution from the construction stage accounts for the remaining 14% of the ADP. Both geotextile membrane (53%) and stone (18%) are the main sources of this.

6.1.3.3 *Acidification potential*

The majority of the 0.10 kg of emissions of SO_2 eq./fu is due to the activities conducted in the corrective maintenance stage (85%) (Figure 30). Stone extraction (30%) and its corresponding disposal (38%) are the major causes of this contribution. About 62% of the stone extraction is due to the electricity generation. Major contributors from the electricity generation are emissions to air of sulphur dioxide (82%) and nitrogen oxides (13%).

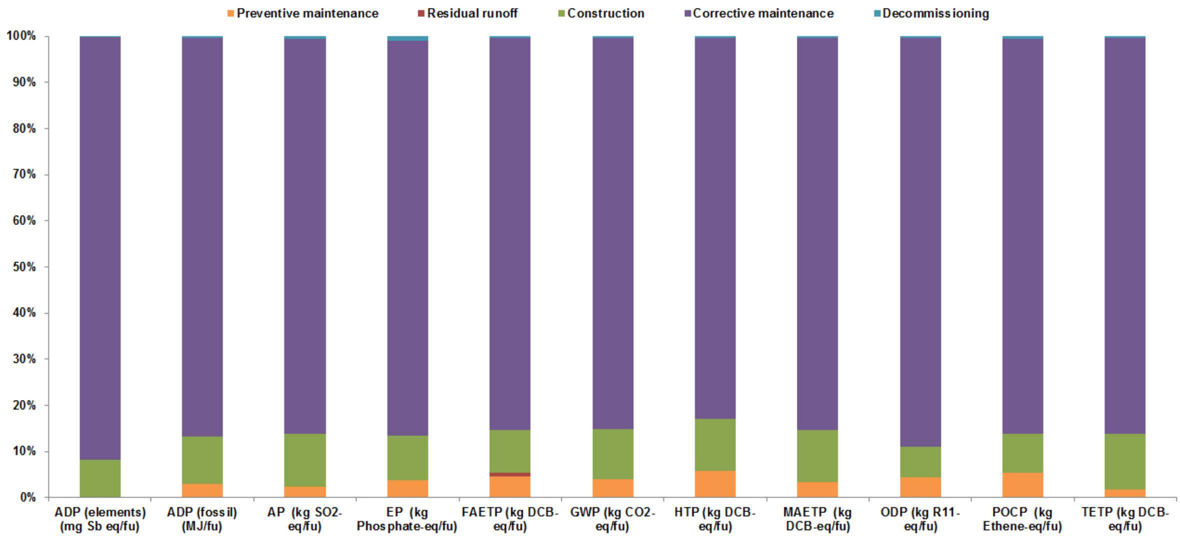


Figure 29 Contribution analysis for the infiltration trench

In the case of the stone disposal processes, the largest source of SO_2 eq./fu emissions comes from building the landfill facility. This accounts for 54% of the contribution of the stone disposal process. Most of this comes from the emissions to air of nitrogen oxides (70%) and sulphur dioxide (29%).

As can be seen in Figure 31, the contribution from the construction stage accounts for 11% of the AP, most of which is due to stone extraction (38%). Sixty percent of this comes from the process of electricity generation, whose major emissions to air are sulphur dioxide (85%) and nitrogen oxides (14%).

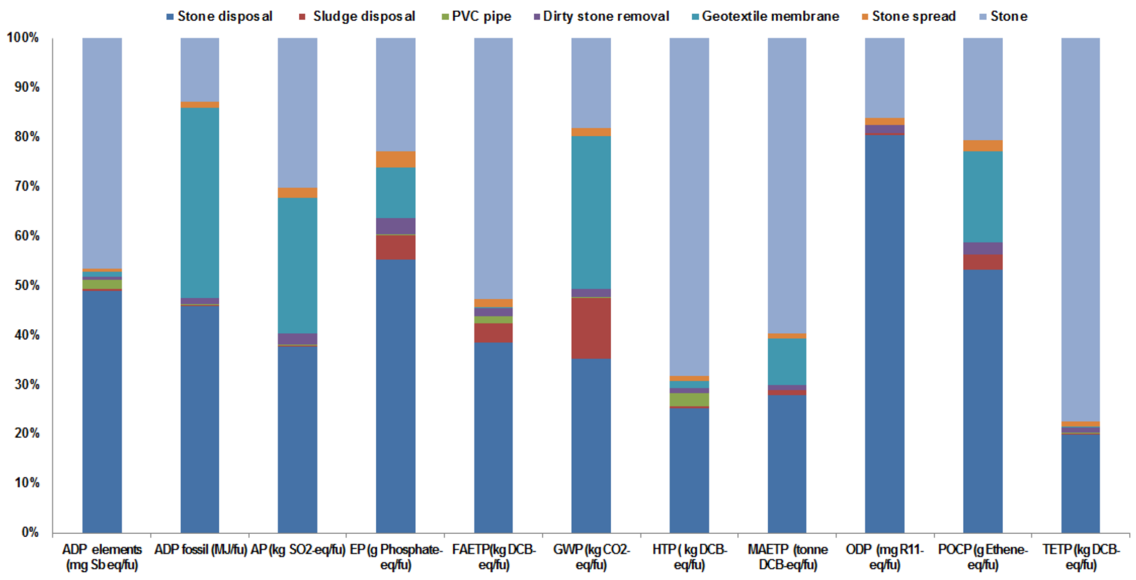


Figure 30 Contribution analysis for the corrective maintenance stage

6.1.3.4 Eutrophication potential

The corrective maintenance stage is the largest contributor (86%) to the EP of 19 g phosphate eq/fu (see Figure 30). Stone disposal (55%) is mainly responsible for this. Most of the stone disposal is due to the construction of the landfill facility (56%), which is dominated by the emissions of nitrogen oxides (60%) to air and emissions of phosphate (60%) to freshwater. The second largest contribution to the corrective maintenance stage is the stone extraction (23%). The main cause of this is the use of construction machinery (56%), whose major emissions are emissions of nitrogen oxides to air (73%).

The rest of the EP is dominated by the construction stage, accounting for 10% (see Figure 31). This is due to the stone transport (41%) and stone extraction (34%). Major emissions from the stone transport are emissions to air (48%), while the largest source of emissions in stone extraction is the machinery used during this process (56%), which mainly emits nitrogen oxides (73%) to air.

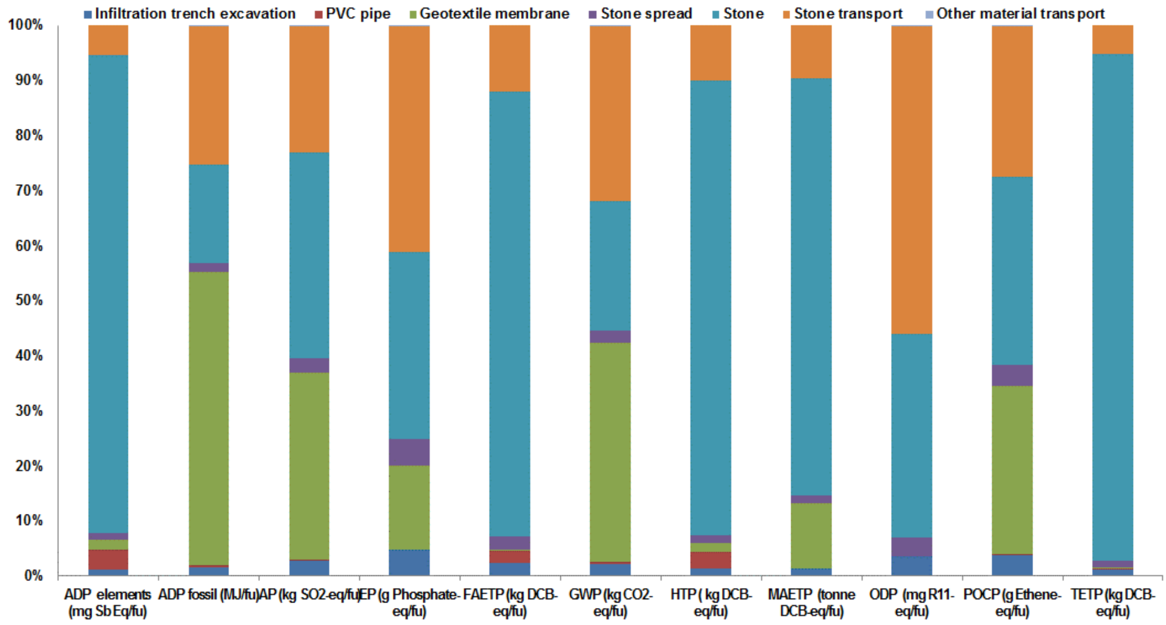


Figure 31 Contribution analysis of the construction stage

6.1.3.5 *Freshwater aquatic ecotoxicity potential*

The major source of emissions to FAETP of 1.19 kg DCB eq/fu comes from the corrective maintenance stage (85%) (see Figure 30). Both stone (53%) and stone disposal (39%) are the main causes of this. In the stone processes electricity generation (33%), which is required during the stone extraction processes, is the biggest source of emissions of DCB eq/fu . This is due to nickel, which is sent to freshwater (50%). The construction of the landfill facility (76%) is the major cause of the stone disposal contribution, which is mainly created by the vanadium emissions to freshwater.

The construction stage accounts for another 9% of the total FAETP (Figure 31). The majority of this is due to the stone extraction (81%), most of which is due to the electricity generation (33%). The contribution from the electricity generation is dominated by the emissions of the heavy metals to air.

6.1.3.6 *Global warming potential*

In this impact category the corrective maintenance stage accounts for 85% of the total 18 kg emissions of CO_2 eq/fu. The main factors responsible for this are stone disposal (35%) and sludge disposal (12%) (see Figure 30). Most of the stone disposal contribution comes as a consequence of the construction of the landfill facility (64%). Carbon dioxide emissions to air (90%) are mainly emitted during these construction activities. The contribution from sludge disposal is dominated by the sludge disposed (98%), this is mainly due to the methane emissions to air.

About 11% of the GWP comes from the construction stage (see Figure 31). In this stage the main source of emissions of CO_2 eq/fu is the geotextile membrane (40%), mainly because of the emissions to air of carbon dioxide (85%) and methane (14%).

6.1.3.7 Human toxicity potential

Human toxicity potential is largely affected by the contribution coming from the corrective maintenance stage (Figure 30), which accounts for 83% of the HTP of 3.8 kg DCB eq/fu. Within this maintenance stage, the largest source of emissions of DCB eq/fu comes from the stone (68%) and stone disposal (25%). Most of the contribution from the stone comes from the electricity generation (60%) due to its emissions to air, specifically of nickel (43%), vanadium (28%) and arsenic (12%).For the stone disposal, the main contributor is the construction of the landfill facility (73%). Most of this is originated by the emissions of chromium and arsenic to air.

Within the construction stage (11%), geotextile membrane accounts for 40% of its contribution (see Figure 31); this is mainly due to the emissions of carbon dioxide (85%) and methane (14%) to air.

6.1.3.8 Marine aquatic ecotoxicity potential

Total MAETP reaches about 4298 kg DCB eq/fu, of which the largest contribution comes from the corrective maintenance stage (85%) (Figure 30). This is mainly due to two processes: stone (60%) and stone disposal (28%). From the stone contribution, electricity generation (66%) is the main factor responsible; this is due to emissions to air of vanadium (48%) and hydrogen fluoride (37%). In the stone disposal process, the largest source of emissions DCB eq/fu is the construction of the landfill facility (52%). Of this, emissions of beryllium to freshwater and of hydrogen fluoride to air are the main factors responsible.

As shown in Figure 31, the construction stage (11%) is the second largest contributor to this impact category. From this, stone (76%) and geotextile membrane production (12%) are the main source of emissions of DCB eq/fu. Most of the stone contribution is due to emissions to air of vanadium (48%) and hydrogen fluoride (37%), while the emissions from the geotextile membrane are entirely dominated by hydrogen fluoride to air (99%).

6.1.3.9 Ozone layer depletion potential

The largest source to ODP is due to the corrective maintenance stage, which accounts for about 88% of the total 2.26 mg R-11 eq/fu. Eighty percent of this is originated from the stone disposal (see Figure 30); this is mainly due to the construction of the landfill facility (84%). Within the landfill facility contribution, emissions of Halon 1301 (98%) are the main factor responsible. Emissions of the stone (16%) are mainly from the machinery (40%). This is produced from the emissions of Halon 1301 to air.

As shown in Figure 29, due to its 6% contribution, the construction stage is the second largest source of emissions to ODP.Of this, stone transport (56%) and stone (37%) are the main factors responsible due to the emissions of Halon 1301 (99%). Electricity generation (45%) is the main cause of the stone contribution; this is also as a result of the emissions of Halon 1301 to air.

6.1.3.10 Photochemical ozone creation

Total Photochemical ozone creation (POC) amounts to 0.014 kg ethane eq/fu. Most of this comes from the corrective maintenance stage (85%) (see

Figure 30). Stone disposal (53%) and stone (21%) are the main factors responsible in this maintenance stage. The stone disposal contribution is dominated by emissions coming from the landfill facility (67%). Emissions of NMVOC (65%) and methane (2%) to air are the main source of this. The electricity generation accounts for about 41% of the stone contribution. This is due to the emissions of sulphur dioxide and nitrogen oxides to air.

Six percent of the remaining contribution comes from the construction stage, mainly due to the stone (34%) and geotextile membrane (31%). Electricity generation (41%) is the major source of emissions of R-11 eq/fu of the stone. This is because of the emissions of sulphur dioxide (63%) and nitrogen oxides (12%) to air. The contribution from the geotextile membrane is dominated by the emissions of sulphur dioxide and hydrocarbons to air.

6.1.3.11 Terrestrial ecotoxicity potential

Most of the 0.11 kg emissions of DCB eq/fu come from the corrective maintenance stage (85%). As shown in Figure 30, the largest contribution to corrective maintenance comes from stone (77%) and stone disposal (20%). Most of the emissions of DCB eq/fu from the stone are caused by the electricity generation (69%), this is due to the emissions to air of vanadium (82%) and chromium (5%). As to the emissions from the stone disposal, they are mainly produced in the construction of the landfill facility (72%). This is due to emissions of chromium and mercury to air.

The construction stage contribution accounts for 12% of the TETP, most of which is due to the stone contribution (92%). This is due to the electricity generation (69%).

6.1.4 Sensitivity analysis

6.1.4.1 Analysis of the variation in the stone removal frequency

As shown in the previous results, the corrective maintenance stage is the main hot spot amongst the life cycle stages, mainly due to stone production and stone disposal. Since the stone is the basis of the infiltration trench design, neither lesser quantity nor replacement by another material is considered. Therefore, the stone disposal is selected as an area of improvement, which is analysed to carry out the sensitivity analysis.

Within the stone disposal processes, the parameter that can be modified is the removal frequency. Stone removal frequency is considered as five years in the base case scenario, since it is assumed that longer removal periods lead to a reduction in the treatment efficiency of the infiltration trench (Lampe et al. 2005). This reduction in efficiency, in turn increases the sludge and pollutant accumulation. Since there is no information about the reduction in the removal efficiency due to longer removal periods, it has been assumed that per every period of five years the efficiency is decreased. Three scenarios have been defined based on this assumption. The removal frequencies as well as the assumed pollutant removal efficiencies are defined for each scenario (Table 46). The variation in the impact categories between these three scenarios is shown in Figure 32. The results from this analysis are summarized and analysed further below.

Table 46 Scenarios defined for the stone and sediment removal frequencies

Stone and sediment removal	Pollutant removal efficiency	Scenario label
5 years	90% for all the parameters	Base case
10 years	45% for all the parameters	SG
15 years	22% for all the parameters	SH
30 years or no removal	11% for all the parameters	SI

Considering as a basis the results from the SI and comparing them with the results from the base case, the impact category with the smallest reduction is freshwater aquatic ecotoxicity potential, with a reduction of 68% from the SI in relation to the base case (1.2 kg DCB eq/fu). Furthermore, human toxicity potential from the SI is reduced by 68% when compared with the base case (3.8 kg DCB eq/fu). The rest of the impact categories have been grouped, since they decrease 70% on average in relation to the value from the base case (see Figure 32)

The increase in the stone removal frequency produces a reduction of impact in all categories (see Figure 32). However, in spite of this reduction, the contribution from the corrective maintenance stage remains the largest contributor.

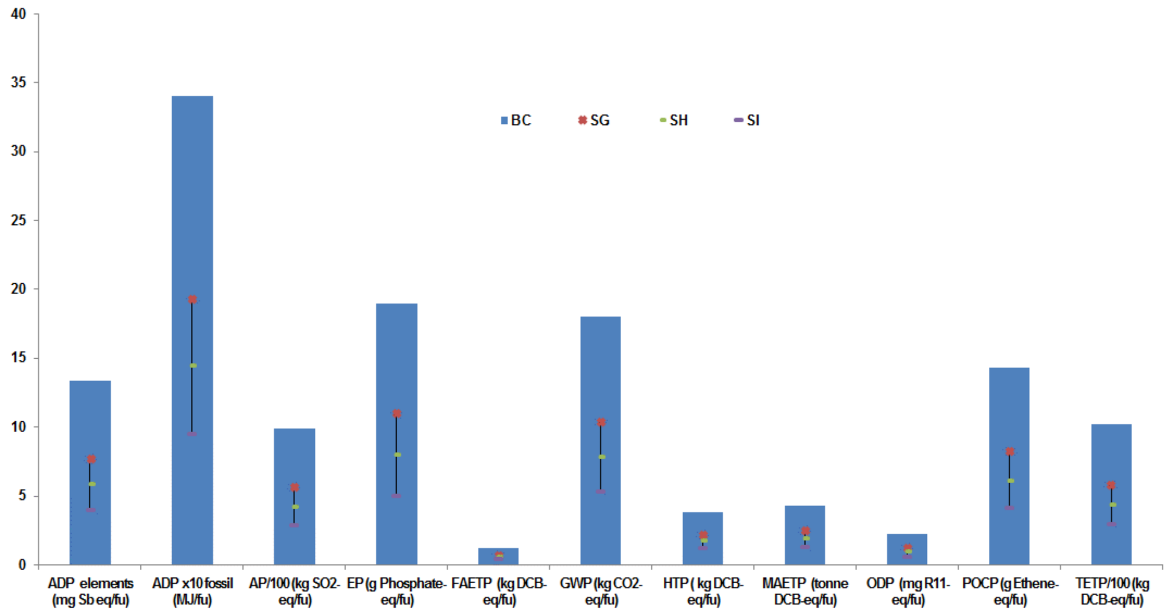


Figure 32 Life cycle contribution analysis due to stone removal frequency

6.1.4.2 Analysis of the variation in the pollutant concentration

As mentioned in section 6.1.2.2, the EMC information is sourced from Kayhanian et al. (2007). In this data set EMC is presented as a range for each parameter (see Chapter 4). The average EMC of this data set has been used for modelling the base case scenario, which is labelled as BC. However, in order to analyse the effect of the EMC variation a sensitivity analysis has been carried out. In this analysis both maximum and minimum EMC have been considered, labelling the maximum as SA the minimum

EMC as SB. Additionally, another scenario (scenario C) has been included. The SC shows the effect on the environmental impacts, if EMC were six times higher than the maximum EMC considered. This scenario might illustrate the prevailing situation in urban areas in developing countries, where a higher EMC might be expected (Jamwal et al., 2008). Finally, in order to analyse the environmental impacts coming due to untreated runoff the Scenario W (SW) is defined. The results from this scenario are compared with the results from the SA, to show the results of the comparison with the worst case in which EMC is the highest.

Due to the assumed composition of the runoff influent, which is made of heavy metals, only FAETP and MAETP are affected by the EMC variation. Therefore, the results described in this section are focused on these two impact categories.

6.1.4.2.1 Freshwater aquatic ecotoxicity potential

The variation of the FAETP is small, ranging from 1.15 (SB) kg DCB eq/fu to 1.61 (SA) kg DCB eq/fu. This change is directly related to the contribution from the corrective and residual runoff stages as shown in Figure 33.

The variation observed in Figure 33 in the contribution from the corrective maintenance stage, comes due to the sludge disposal. This is explained by the large difference in the EMC from the SA, BC and SB. Hence, that the higher the EMC, the higher the contribution from this stage.

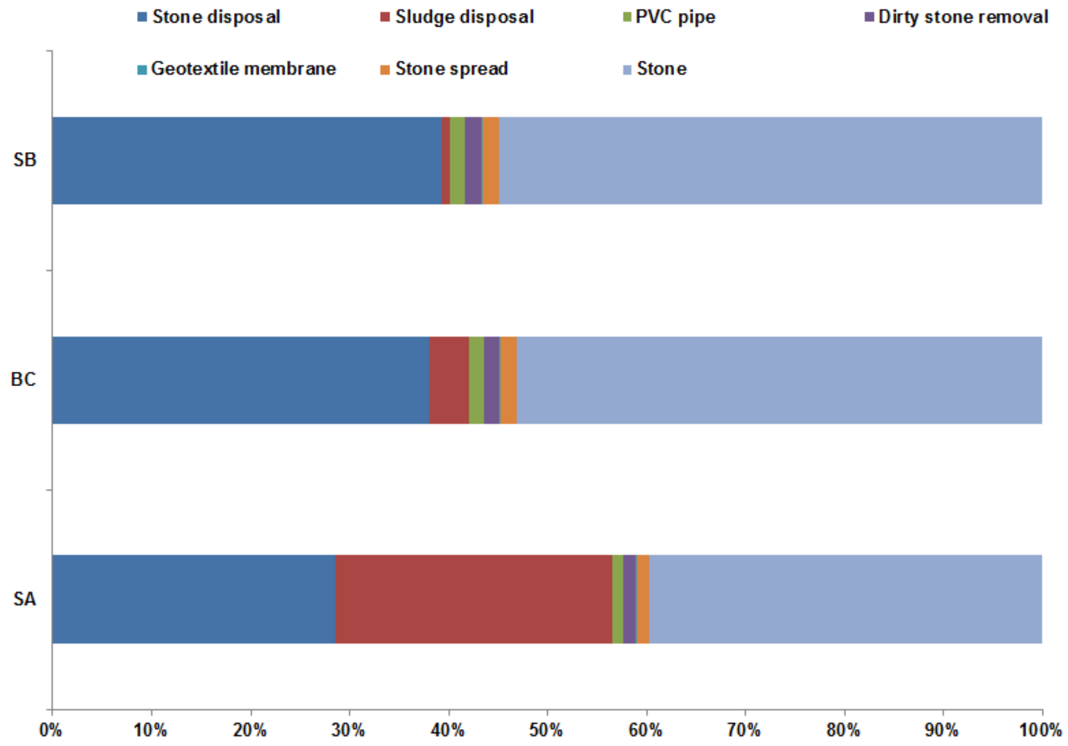
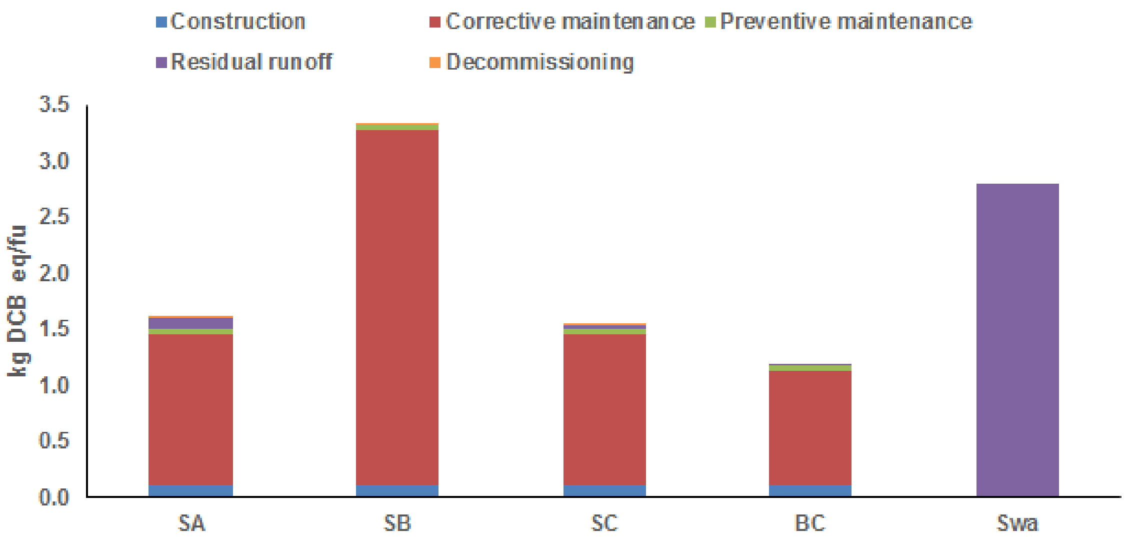


Figure 33 Contribution analysis for SA, BC,SB for FAETP

In contrast, since the difference between the EMC from the SA and the SC is small, their corrective maintenances' contributions remain without change. Hence, as seen in Figure 34, the contribution from the residual runoff stage is the main difference in the FAETP from SA and SC.

In the analysis of the FAETP, it is observed that treating this runoff actually leads to an increase in the emissions of DCB eq/fu. As seen in Figure 34, this is caused by the contribution of the corrective maintenance stage, which would not be noticed if the analysis were carried out considering only the residual runoff stage.



a)The values shown have been scaled by multiplying by 100;

Figure 34 FAETP for different scenarios

6.1.4.2.2 Marine aquatic ecotoxicity potential

MAETP varies from 4276 to 4550 kg DCB eq/fu, the highest value observed in the SA. As in the results from FAETP, this is explained by the fact that the EMC only affects the contribution from the residual and corrective maintenance stages. Hence, that the higher the EMC, the higher the contribution from these stages.

When the MAETP from the SC is compared against the MAETP from the SA, an increase of 3% is found. This increase is derived from the contribution of the residual runoff stages, as shown in Figure 35.

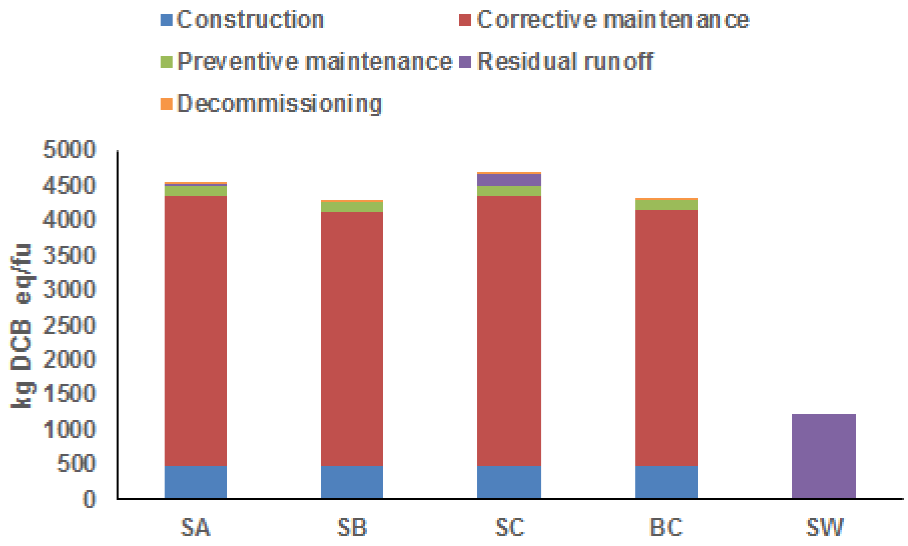


Figure 35 MAETP for different scenarios

As explained in the case of FAETP, that the difference between MAETP from the SC and SA is small is due to the fact that the EMC between these scenarios is also small.

Finally, in the comparison of the MAEPT with the runoff treatment (SA) and the scenario without treatment (SW), it is observed that the SA is the highest. As shown in Figure 35, this is mainly due to the emissions of DCB eq/fu from the construction and the corrective maintenance stages.

6.1.4.3 Analysis of the volume treatment variation

A parameter that affects the runoff volume to be treated is the rainfall depth of the design storm. Since this information is not available for the case of the area of the Magdalena river catchment, the variation of this parameter is included as part of the sensitivity analysis. In the absence of data, the rainfall depth of 24.5 mm considered for the base case scenario is increased five times.

In order to analyse this variation of the volume treatment within the EMC range, three scenarios have been defined. The BC2 corresponds to the average EMC, while the SE and the SF correspond to the maximum and minimum EMC respectively.

The results are presented in two groups; in the first group the impact categories not affected by the EMC variation are shown. These impact categories are ADP (elements), ADP (fossil), AP, EP, GWP, HTP, ODP, POCP and TETP. In the second group FAETP and MAETP, which are affected due to the EMC are described. As seen in Figure 36, the variation in the categories analysed in the first group amounts on average to 70%. The main difference comes from the preventive maintenance; this is because unlike the other life cycle stages, preventive maintenance requirements remain without change in spite of the treatment volume increase.

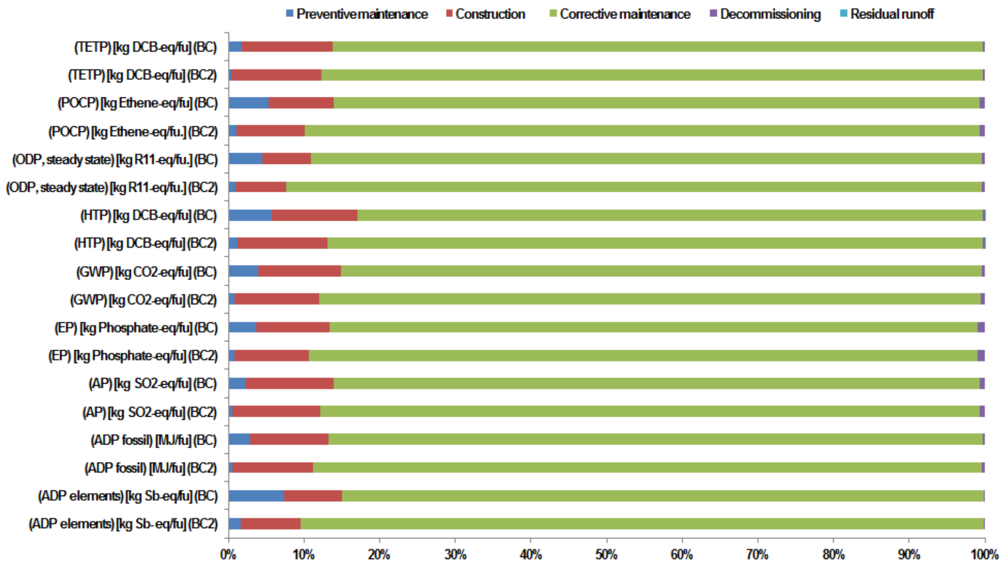


Figure 36 Sensitivity analysis results due to rainfall depth variation

6.1.4.3.1 <u>Freshwater aquatic ecotoxicity potential</u>

The FAETP range that corresponds to the increase of the treatment volume is shown in Table 47. These values are smaller than their counterparts with the same EMC and smaller treatment volume. In the case of the case of the SD, it is 3% smaller than the FAETP from the SA, while in the case of BC2 and SE, they are 4% and 4.5% smaller than the BC and the SB.

As shown in Figure 36, these reductions are mainly due to the decrease in the contribution from the preventive maintenance stage.

Table 47 FAETP and MAETP variation due to the volume treatment

Scenario	MAETP (kg DCB eq/fu)	FAETP (kg DCB eq/ fu)
SD	4456	1.56
BC2	4204	1.14
SE	4182	1.10

The contribution from the preventive maintenance stage is reduced from 3.37% (SA) to 0.70% (SD), from 4.54% (BC) to 0.95% (BC2) and from 4.7% (SB) to 0.98% (SE). The contributions from the rest of the life cycle remain without change.

6.1.4.3.2 Marine aquatic ecotoxicity potential

The MAETP variation due to the increase in the treatment volume is shown in Table 47. These values are overall 2% smaller than the corresponding values from the scenarios with the same EMC. This reduction is due to the difference from the contribution from the preventive maintenance stage.

In the contribution analysis for the BC2, MAETP is dominated by the contribution from the corrective maintenance stage. This contribution accounts for 87.5% of the 4202 kg DCB eq/fu which comes from this maintenance stage. From this the largest contribution comes from the stone (60%), this is due to the emissions of hydrogen fluoride to air (33%). The increase in the treatment volume leads to a reduction in FAETP and MAETP.

In a nutshell, the increase in the treatment volume leads to a reduction for all the impact categories. This reduction is due to an inverse relationship created by the increase in the treatment volume.

6.1.5 Validation of the results

Since the installation of infiltration trenches is a practice in its infancy, information for conducting LCA for this practice is scarce. Hence, no LCA study has been previously conducted for this treatment practice, so that there is no basis for comparing the results of the current study.

6.1.6 Summary

A summary of the major outcomes found in this analysis is given in this section, which is divided into two main sets, the first one for the results for the base case and the second one for the results of the different sensitivity analysis. For the base case the following results are observed:

- Corrective maintenance is the stage that contributes most to all the impact categories, accounting for on average 85% of all the environmental impacts; this is mainly due to the emissions to air caused by stone and stone disposal processes.

- Construction stage is the second largest contributor for all the impact categories; it is also the stage where due to the substitution of machinery by hand labour (when this is possible), a reduction in the environmental impacts might be achieved.

- The contributions from decommissioning, preventive maintenance and residual runoff are not representative for the considered impact categories.

- Although the contribution from the sludge disposal was expected to be one of the highest for the considered impact categories, results show that it is not a hot spot for any impact category. This is because, assuming that sludge is removed every five years, the accumulated heavy metals are not sufficient to be a representative contribution.

In addition to the results of the base case scenario, the results from the sensitivity analysis are described in this section. Although runoff is not identified as a hot spot, the variation of the pollutant concentration is defined as a parameter to be varied, since there is no unique value for this parameter. Therefore, the effect on the impact categories is described further below. Furthermore, the results of increasing the pollutant concentration (SC) six times, which might illustrate the EMC for an urban area in the developing country, are also described in this section.

- Based on the parameters considered as part of the residual runoff stage, pollutant concentration affects two impact categories: MAETP and FAETP. Within the pollutant range considered in Chapter 4, FAETP ranges from 1.19 to 1.61 kg DCB eq./fu, and MAETP ranges from 4278-4550 kg DCB eq./fu.

- FAETP and MAETP from the SC increase by 3% and 18% in comparison with the FAETP and MAETP from the SA;

The final comparison related to the variation in the EMC refers to the case where the effect of untreated runoff is analysed, in order to know if runoff treatment is helping to reduce the environmental impacts.

- The infiltration trench installation leads to a dramatic increase in both FAETP and MAETP. FAETP from the SA is 53 times higher than the FAETP SW, while the MAETP increases by about 400 times in relation to SW. These increases are both a consequence of the clean stone installation during the corrective maintenance stage. This is considered as an inevitable variation because stone removal is a determinant for ensuring proper treatment.

Finally, the results of the sensitivity analysis of the treatment volume, which affects the contribution from the construction stage, are described below. Even though this stage is not identified as a hot spot, it is analysed due to the high variability that might occur in the rainfall depth that affects the treatment volume.

- The increase in the treatment volume leads to a decrease in all the impact categories; this is because the variation in the contribution from the life cycle stages is smaller than the variation in the treatment volume.

- The variation in the stone removal frequency that is required during the corrective maintenance leads to a reduction of 50% in all the environmental impacts from these scenarios in comparison with the results from the BC.

As shown in the results from the base case, the only hot spot identified is the corrective maintenance stage. However, as explained above, both the construction and the runoff residual stages are also selected as part of the sensitivity analysis.

6.2 Economic evaluation of an infiltration trench

6.2.1 Goal and scope

The aim of this study is to identify and quantify the economic “hot spots” along the life cycle stages considered in the LCA study. As shown in Figure 28, these stages are construction, corrective maintenance, preventive maintenance, residual runoff and decommissioning. Both the functional unit, which is defined as “treatment of 1 m^3 runoff over the 30 years” and the system boundaries are also defined based on the LCA study.

6.2.2 Inventory analysis

6.2.2.1 Assumptions

There is no standardized way to calculate the cost of this kind of practice. On the one hand, storm water managers have used capital cost as a basis for economic comparisons. This cost is highly variable, since it depends on different parameters such as the characteristics of the land, the cost of the land, the design, the characteristics of the drainage area and the elements considered as part of the capital cost estimation (Young et al., 1996;EPA, 1999;Taylor, 2005). Furthermore, when this estimation is considered, other costs such as maintenance cost and decommissioning are underestimated in an initial budget. Hence there is a need to introduce a life cycle approach that enables the calculation of these costs over the life span of the infiltration trench. The use of life cycle cost (LCC) is a recent innovation in the economic analysis of this treatment practice; therefore there is a shortage of models for conducting this calculation. In this regard, Lampe et al. (2005) and (Taylor & Fletcher, 2007) have developed a framework for conducting the life cycle cost analysis. In addition to these studies, the current study aims to contribute to the life cycle costing analysis by calculating the life cycle cost using the approach developed by Swarr et al. (2011), which has been selected since it is consistent with the LCA approach.

As stated before, the life cycle stages considered for the life cycle cost estimation are: construction, corrective and preventive maintenance and decommissioning. The elements considered for the estimation of the cost per life cycle stage are:

- Construction stage: capital cost, including material cost, material transport material and handling;
- Corrective maintenance stage: inspection activities, sludge removal, sludge transport and sludge disposal as well as the cost of stone transport;
- Preventive maintenance stage corresponds to the cost of the inspection activities, mowing and transport of mowed material;
- Decommissioning stage encompasses the cost of removing material and top soil installation.

The cost of the residual runoff stage has not been considered, since within this stage runoff passes through the infiltration trench and this does not lead to any cost. Information for the cost estimation has been taken from the life cycle inventory of the LCA study. Costing information for each activity considered is shown below from Table 52 to Table 55, while data sources are described further below.

6.2.2.2 Data sources

Since there is no data available for the cost of sludge and stone disposal, these data have been sourced from the typical cost for disposing an 8 cubic yard skip in the UK from (Wills, 2011). The rest of the costing information, which encompasses the cost of machinery operation, hand labour and materials, has been sourced from a Mexican construction cost database generated by the Mexican Institute of Cost Engineering (Gonzalez, 2012).

The cost of each stage considered in the life cycle has been calculated by multiplying the flows from each life cycle stage of the LCA evaluation by the corresponding cost. The quantities of the flows are presented in section 6.1.2.2, while the costs of each activity are presented from Table 48 to Table 51.

Table 48 Data sources for the construction stage

Material installation cost			
Activity	*Cost*	*Quantity*	*Data source*
Trench excavation	1151	\$/hour	(Gonzalez, 2012)
Stone spread	1151	\$/hour	(Gonzalez, 2012)
Material cost			
Material	*Cost*	*Quantity*	*Data source*
Stone	213	$\$/m^3$	(Gonzalez, 2012)
Geotextile membrane	306	\$/roll	(Gonzalez, 2012)
PVC pipe	46	\$/m	(Gonzalez, 2012)
Material transport cost			
Transported material	Cost (US\$)	Quantity	Data source
Stone	53	$\$/m^3$	(Gonzalez, 2012)
Geotextile membrane and PVC	53	$\$/m^3$	(Gonzalez, 2012)

Table 49 Data sources for the preventive maintenance stage

Activity cost	*Cost*	*Quantity*	*Data source*
Inspection	512	\$/hour	(Gonzalez, 2012)
Debris and trash removal	512	\$/hour	(Gonzalez, 2012)

Table 50 Data sources for the corrective maintenance stage

Material removal cost			
Material	*Cost*	*Quantity*	*Data source*
Dirty stone removal	1151	$\$/m^3$	(Gonzalez, 2012)
Clean stone installation	1151	$\$/m^3$	(Gonzalez, 2012)
Sludge removal	1151	$\$/m^3$	(Gonzalez, 2012)
Material cost			
Material	*Cost*	*Quantity*	*Data source*
Stone	213	$\$/m^3$	(Gonzalez, 2012)
Geotextile membrane	306	\$/roll	(Gonzalez, 2012)
PVC pipe	46	\$/m	(Gonzalez, 2012)

Cost of material disposal			
Activity	*Cost*	*Quantity*	*Data source*
Stone disposal	573	$/m^3$	(Wills, 2011)
Sludge disposal	0.82	$/m^3$	(Wills, 2011)
Cost of material transport			
Material	*Cost*	*Quantity*	*Data source*
Stone	53	$/m^3$	(Gonzalez, 2012)

Table 51 Data sources for the decommissioning stage

Cost of material removal/implementation			
Material	*Cost*	*Quantity*	*Data source*
Stone removal	1151	$/hour	(Gonzalez, 2012)
Soil implementation	1151	$/hour	(Gonzalez, 2012)
Material cost			
Material	*Cost*	*Quantity*	*Data source*
Top soil	310	$/m^3$	(Gonzalez, 2012)

Estimated costs following the procedure above described are shown from Table 52 to Table 55.

Table 52 Costs of the construction stage

Construction stage	
Material installation cost	
Activity	Cost (US$)
Trench excavation	1751
Stone spread	1751
Geotextile membrane and PVC installation	56
Material cost	
Material	Cost (US$)
Stone	3891
Geotextile membrane	169
PVC pipe	5
Material transport	
Material transport	Cost (US$)
Stone	964
Geotextile and PVC pipe	22

Table 53 Costs of the preventive maintenance stage

Preventive maintenance	
Activity	Cost (US$)
Inspection	2,000
Debris and trash removal	2,000

Table 54 Costs of the corrective maintenance stage

Corrective maintenance stage	
Material installation using machinery	
Activity	Cost (US$)
Dirty stone removal	10501
Clean stone installation	10501
Sludge removal	36000
Material installation (hand labour)	
Activity	Cost (US$)
Geotextile membrane pvc pipe removal	674
Material	
Material	Cost (US$)
Stone	23,300
Geotextile membrane	1000
PVC pipe	26
Disposal	
Activity	Cost (US$)
Stone disposal	62000[a]
Sludge disposal	302[b]
Transport cost	
Activity	Cost (US$)
Geotextile membrane and PVC pipe transport	6
Sludge	19000
Stone transport	5800

a and b: The difference between these values is due to the costs for each activity, which are presented in Table 50.

Table 55 Costs of the decommissioning stage

Decommissioning	
Material handling	
Activity	Cost (US$)
Stone removal	820
Soil installation	820
Material	
Material	Cost (US$)
Top soil	5,600

After completing the LCC estimation following the assumptions described above, the activities that mainly contribute to this cost are identified. These results are described in the section below, they are presented in American dollars for consistency purposes with the available information for the infiltration trench cost.

6.2.3 Results and discussion base case

Over the infiltration trench life cycle, its LCC amounts to US$ 191,100, when calculated per functional unit, this LCC amounts for $65/fu. About 89% of this cost is due to the corrective maintenance (see Figure 37). Most of the maintenance contribution comes from the cost of machinery which is used for sludge disposal (37%), and removal and installation of clean stone (33%).

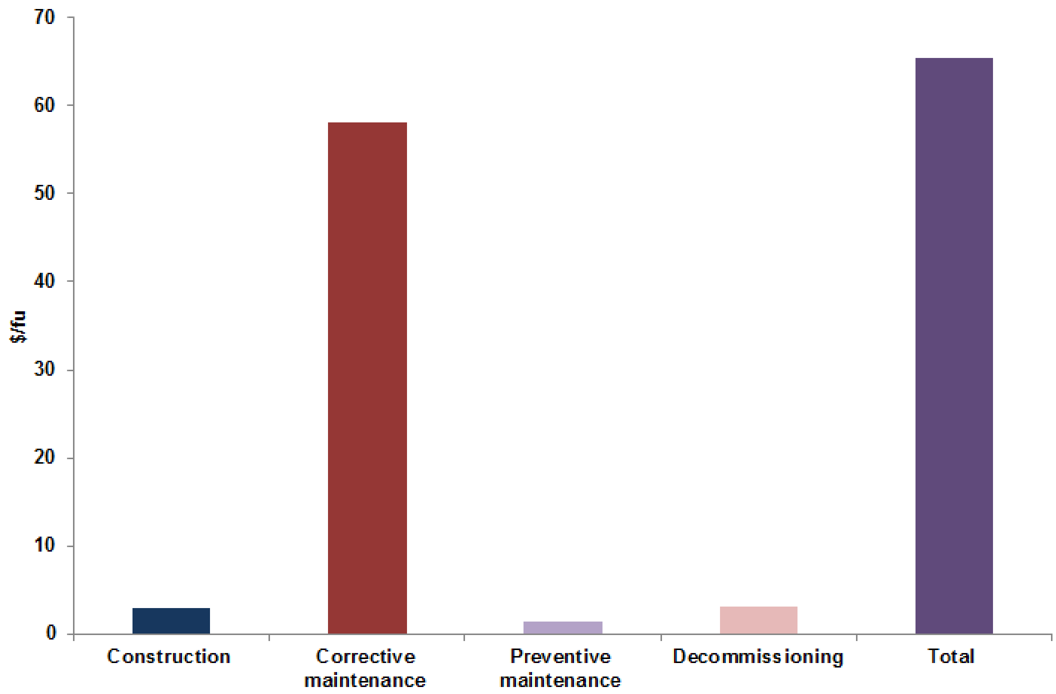


Figure 37 LCC results for the infiltration trench

The remaining 11% comes from the construction stage (4%), decommissioning (4%) and preventive maintenance (3%). The material cost represents most of the cost of the construction stage (53%) and the decommissioning stage (62%), while the cost of debris and trash removal accounts for 51% of the preventive maintenance cost.

Even when the cost of the corrective maintenance stage is identified as a hot spot, this cost cannot be analysed under a sensitivity analysis, since it mainly comes due to the machinery use which has a fixed cost. However, a sensitivity analysis has been carried out, in order to analyse the variation in the LCC due to the increase in the volume treated in the infiltration trench. This parameter is selected for consistency with the sensitivity analysis of the LCA study, which also included the analysis of the pollutant concentration, however since this variation does not affect the calculated cost, it has not been included as part of the sensitivity analysis of the LCC.

Another aspect considered in the sensitivity analysis is the fluctuation of the net present value (NPV) of the LCC due to different discount rates. As explained further below, the NPV is calculated according to Swarr et al. (2011) principle that the time-value of money should be analysed if the life time of the asset is more than two years. The results of the sensitivity analysis are described in the section below.

6.2.4 Results and discussion: sensitivity analysis

6.2.4.1 Analysis of the treatment volume

The treatment volume is the parameter that defines the size of the infiltration trench (MDE, 2002). As mentioned in section 6.1.2.2, according to the LCA study the treatment volume has been calculated based on a design storm with a rainfall depth of 24.5 mm (1 in). This rainfall depth can vary, affecting the calculated treatment volume and therefore the material requirements. The impact of the size increasing on the cost is analysed in this section.

Total cost amounts to $64.71/fu, which is 0.45% less than the cost of the infiltration trench with five times less treatment volume. The cost from the corrective maintenance is the highest, accounting for 91% of the LCC, followed by the cost of decommissioning (4.5%), construction (4%) and preventive maintenance (0.5%). Therefore, the results show that the five times increase of the rainfall depth, has no impact in terms of cost. That the cost is reduced is explained since the increase in the rainfall depth leads to an increase in the treatment volume, which has been defined as the functional unit.

According to Swarr et al. (2011), if the life span of the asset under analysis is more than two years, then a discount rate should be considered in order to calculate the money variation in relation to time. In order to analyse this variation, net present value is used as a discounting method, since it is the recommended method by Lampe et al. (2005) for the economic analysis of treatment practices.

The discount rate used to calculate the net present value is analysed, in this case a discount rate of 12% (SHCP, 2009), which corresponds to the case of Mexico, has been used. As part of this sensitivity analysis, this discount rate has been varied to analyse its effect in the NPV estimation (Figure 38). The discount rate ranges from 5.5% to 12% in order to make a comparison of the typical values for discounting future cost to the present in the US and the UK, which according to Lampe et al. (2005) can be considered as 5.5%.

The results of the LCC calculated with different discount rates show that the higher the discount rate the lower the LCC. Thus the LCC with 5.5% is 55% lower than the LCC from the BC (US$64.71/fu), while the LCC with 12% is 76% lower than the LCC of the BC.

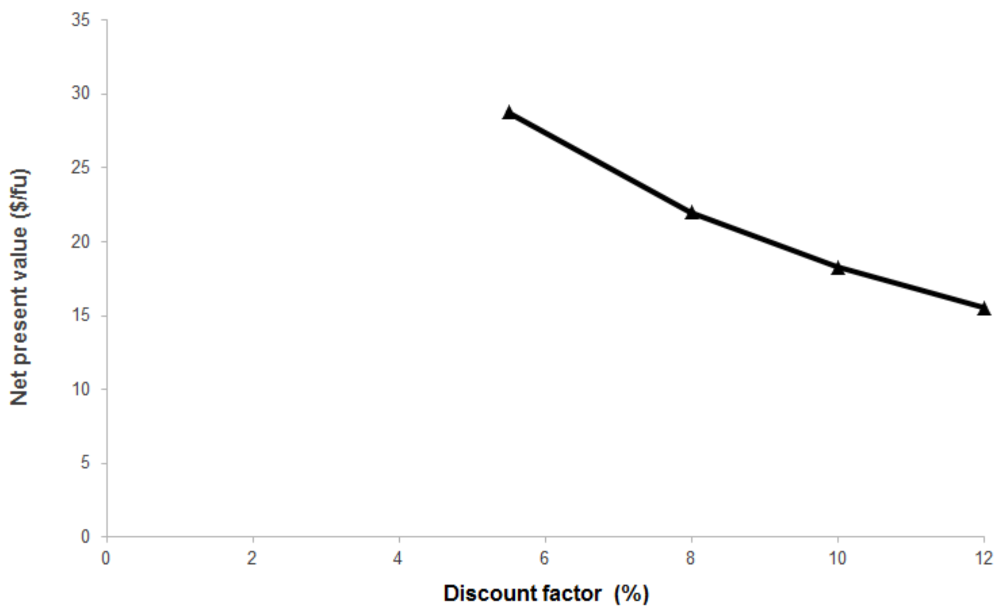


Figure 38 Variation of LCC using different discount rates

6.2.5 Validation of the results

There is little information for the LCC comparison of the infiltration trench. Therefore, based on the available information the validation of the results from this study is carried out in two senses: quantitative and qualitative.

The results of the quantitative analysis are presented in absolute values and not per functional unit, since information for calculating the treatment volume is not presented in Lampe et al. (2005). For carrying out this quantitative analysis, information about the drainage area, imperviousness, land used for the design of the infiltration trench used in the LCA study (see Table 57) is input in the spread sheet used for conducting the LCC according to Lampe et al. (2005). Based on this approach the LCC amounts to 35,640 US dollars, which is about five times less than the value obtained in this study (US$ 191,000). This is due to the difference in the activities considered in the corrective maintenance, which for this study represents the main contribution, accounting for 89% of the total LCC. This is since it has been assumed that the stone is removed every five years in order to avoid improper operation due to clogging. Although there is no definitive frequency for conducting this activity, based on the results from the infiltration trenches in the field, they do not last more than five years without clogging (Lampe et al. 2005). As shown in Table 56, the contribution from the corrective maintenance is not the main one according to the Lampe et al. (2005) approach, since it accounts for 28% of the calculated LCC. This cost includes as part of the regular maintenance activities only cleaning and jetting, which explains the reduction in the cost of this maintenance stage in comparison with the calculated cost of the corrective maintenance in this study.

In regard to the cost of preventive maintenance, in this study it is 4 times higher than the cost of preventive maintenance calculated by Lampe et al. (2005). On the one hand, as shown in Table 57, the difference comes from the kind of activities considered as preventive maintenance. On the other hand, the cost of these activities is mainly due to hand labour which is widely different between the U.S. and the UK and Mexico, where the cost of hand labour is considerably lower. Finally, the capital cost calculated by Lampe et al. (2005) is about 1.2 times higher than the one calculated in this study. As shown in Table 57 this difference might come from the cost of land, design and planning which have not been included in this study. These costs have not been included because there is no information available, since controlling diffuse water pollution in Mexico is at a preliminary stage (see Chapter 4). For this reason, although different sources were consulted, none could provide information related to either the cost of the design and planning or land to build an infiltration trench.

Table 56 Comparison of the contribution per cost element

Cost element	Lampe et al. 2005 ($)	This study ($)
Capital cost	9,750 (27.4%)	7,576 (4%)
Preventive maintenance	15,810 (44.4%)	4,031 (2%)
Corrective maintenance	10,080 (28.3%)	169,419 (89%)
Decommissioning	NA*	9,160 (5%)
LCC	35,640	190,185

*NA: not available

As shown in Table 57, the contribution of each of the cost elements greatly varies in relation to the activities considered. For example, in regard to the corrective maintenance it is hard to define a standard for the activities, because they will depend on the failures that occur in every infiltration trench in the field, these will depend in turn on the frequency and actual implementation of the preventive maintenance (EPA, 1999). This explains the variation in the calculation of the costs coming from the maintenance.

The site is another factor which might increase or decrease the frequency of carrying out infiltration trench maintenance. Hence the costs from both corrective and preventive maintenance are considered as a key factor in the variability of the calculated LCC. Finally, since the decommissioning of the infiltration is rarely performed, its cost is hard to estimate, thus it might either be considered or not in the models developed for the LCC estimation.

In a nutshell, the results obtained in the current study show that, as is explained in the qualitative analysis, there is a high variation in the LCC based on the activities and their frequencies for being implemented.

Table 57 Comparison of cost elements for different LCC approaches

Element cost	**Lampe et al. (2005)**	(Taylor & Fletcher, 2007)	**Present study**
Capital cost	Planning and site investigation cost; Design and project management site supervision; Clearance and preparation; Material; Construction (labour and equipment); Planning and post construction; Cost of land	Conceptual design; Preliminary design; Construction/ purchase; Overhead cost (eg. Project contract management cost);	Material Labour Machinery Material transport
Preventive maintenance	Clearing debris; Grass cutting; Vegetation management; Litter removal; Jetting of permeable surfaces	Typical annual maintenance cost	Inspection (labour and personal transport); Trash and debris removal (labour and personal transport)
Corrective maintenance	Clearing and jetting; Major rehabilitation	Renewal and adaptation cost	Stone removal (material, equipment and labour); Sediment removal (equipment and labour)
Decommissioning		Optional fully reinstalling the site	Material removal (equipment and labour); Top soil installation (equipment and labour)

6.2.6 Summary

The calculated LCC over the life cycle of the infiltration trench amounts to US$ 191,000 or US$65/fu, most of which comes due to the contribution from the corrective maintenance stage (89%). This is due to the inclusion of the stone removal as part of the corrective maintenance. As to the contribution of the preventive maintenance, it amounts to 2% of the LCC.

The results show that the difference from the LCC from Lampe et al. (2005) comes due to three factors: kind and frequency of activities considered as maintenance, as well as the cost of hand labour. In addition, it is observed that the inclusion of the land cost, which is not considered in the current study, might represent a source of variability in the estimation of the LCC.

After analysing bio-retention and infiltration trench economic and environmental impacts, both h environmental and economic impacts of the porous pavement are described in the next chapter.

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7 Porous pavement: Environmental and economic evaluation

Porous pavements are becoming an attractive option to combat the constant increase of impervious surfaces in urban areas. This is due to the fact that this kind of pavement provides improvements in controlling both runoff quality and quantity, which could help to transform polluted runoff into a valuable resource. The environmental and economic impacts of porous pavement are analysed and described in this chapter.

7.1 Environmental evaluation

7.1.1 Goal and scope definition

This study is focused on the treatment of polluted runoff by porous pavement. Thus, the goal of this study is the identification and quantification of the environmental impacts due to this treatment along the life cycle of the porous pavement. The scope of the analysis is from “cradle to grave” for the stages considered, which as shown in Figure 39 are: construction, operation (residual runoff) and maintenance. In this case, decommissioning has not been considered, since usually, if well designed, porous pavement remains installed without requiring removal (Cahill, 2003).

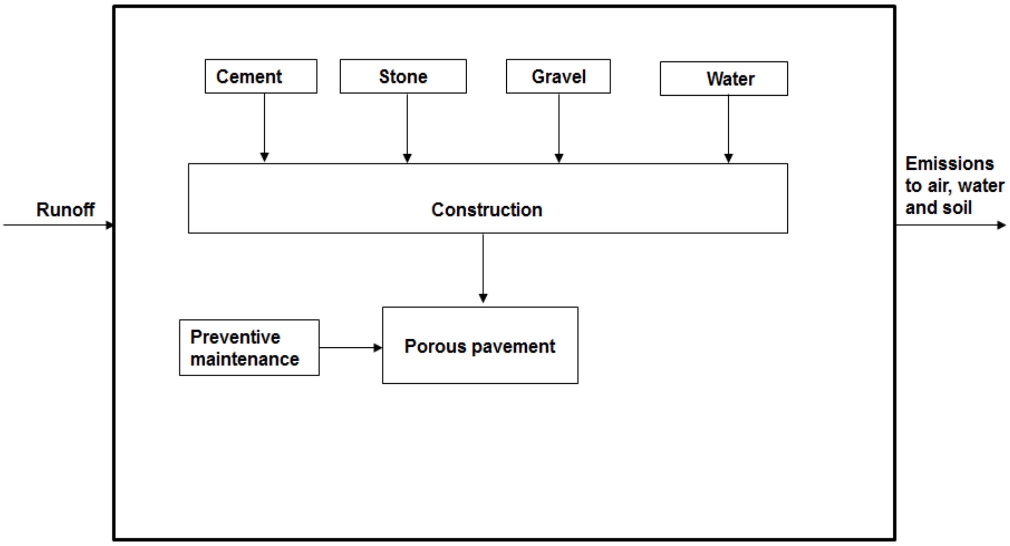

Figure 39 Life cycle of porous pavement

In this work, maintenance has been considered as preventive, in the sense that it is conducted in order to avoid porous pavement clogging (EPA, 1999). This maintenance refers to washing the porous pavement surface, thus preventing the porous pavement from working inadequately as a treatment option. In regard to the operation stage, in this work it is referred to as residual runoff stage, which denotes polluted runoff passing through the porous pavement structure. Lastly, in regard to the construction stage, both processes and assumptions considered are also described in section 7.1.2.

The life span of the porous pavement has been considered as 30 years. Since the implementation of porous pavement is a relatively young practice, there is no established standard life span. Therefore, based on the information provided by Lampe et al. (2005), the period of 30 years has been considered for the research purposes of this work. As to the functional unit refer to Chapter 4 to see its definition.

7.1.2 Inventory analysis

This section details the assumptions and data sources used in this study to describe the three life cycle stages of porous pavements: construction, preventive maintenance and residual runoff.

7.1.2.1 Assumptions

In order to analyse the environmental impacts from controlling diffuse water pollution in an urban area in the developing world, as explained in Chapter 4 it has been assumed that the porous pavement is installed in Mexico City, specifically in the Magdalena River Catchment (MRC). Therefore, a series of assumptions for each stage considered within the porous pavement life cycle has been elaborated based on this fact. A description of the criteria used to define assumptions per life cycle stage is given in this section.

Porous pavement systems are categorized into porous asphalt and porous concrete (EPA, 1999). This work is based on porous concrete, which has been selected based on data availability for this kind of pavement. Porous concrete is already built and installed in motorways and parking lots in Mexico City and it is being built in the MRC area. However, it is not aimed at controlling diffuse water pollution, but at the control of peak flows during the wet season. Material requirements correspond to the calculated treatment volume. There is no guideline that establishes the procedure to estimate this treatment volume in the Mexican context. Hence, guidelines have been taken from the Maryland storm water design manual (MDE, 2000). The estimation of the treatment volume following the guidelines provided in this design manual is shown in Chapter 4. This volume amounts to 50 m^3, since it corresponds to 0.4 ha of impermeable area, which according to the company[6] information corresponds to 1053 m^2 of porous pavement, which actually acts as the drainage area.

[6] Due to a confidentiality agreement the name of the company cannot be shown

Based on the calculated treatment volume, materials required during the construction stage are calculated. The system defined by porous pavement is composed of three layers: porous material, gravel and ballast; and their depths are calculated based on the number of vehicles that pass over the porous surface. Nevertheless, since in this work the functional unit has been established as treatment of 1 m^3 runoff over 30 years, the criteria for material calculation has been changed. In order to consider volume as a basis for material computation, two of the three layers' depths of the porous pavement system have been fixed. The depth of the porous material has been fixed as 8 cm, while the gravel depth has been fixed as 5 cm, based on the guidelines provided by the company. The depth of the ballast layer has been calculated based on the water volume to be stored and treated. A summary of the calculated materials is shown in Table 58.

Table 58 Inventory for the construction stage

Construction stage	
Material requirements	
Ecocreto layer	
Parameters	**Quantity**
Ecocreto layer depth	8 cm
cement	21 tonne
aggregate (gravel)	77 m^3
aggregate (gravel)	117 tonne
additive (□olymers)	608 lt
additive (□olymers)	608 kg
water	7040 lt
Gravel layer	
Parameters	**Quantity**
gravel layer depth	5 cm
gravel	40 m^3
gravel	61 tonne
Ballast layer (small stone 4"-8")	
Parameters	**Quantity**
Ballast layer depth	0.083 m
ballast	66 m^3
ballast	105 tonne
Total amount of gravel	
Parameters	**Quantity**
gravel	117 m^3
gravel	178 tonne

In regard to the residual runoff stage, pollutant mass treated in porous pavement is assumed as pollutant load. Pollutant load has been calculated based on the simple method (Chandler 1994), according to Equation 6. Both the procedure and data sources for conducting this estimation are shown in Chapter 4. A summary of the calculated pollutant loads is shown in Table 59.

Table 59 Inventory for the residual runoff stage

Residual runoff	
Pollutant loud in [a]	**Mass (kg/year)**
TSS	524
Cd	1.23E-03
Cr	1.51E-03
Cu	5.88E-03
Ni	1.93E-03
Pb	8.25E-03
Zn	3.28E-02
Pollutant load out	**Mass (kg/year)**
TSS	47.18
Cd	8.23E-04
Cr	1.51E-04
Cu	3.41E-03
Ni	1.93E-04
Pb	2.14E-03
Zn	6.23E-03
Pollutant load accumulated	**Mass (kg/year)**
TSS	477.07
Cd	4.05E-04
Cr	1.36E-03
Cu	2.47E-03
Ni	1.74E-03
Pb	6.10E-03
Zn	2.66E-02

[a] The pollutant loads on the inlet of the porous pavement have been calculated based on the average EMC values from Kayhanian et al. (2007).

Preventive maintenance activities have been assumed as hosing the porous pavement surface twice per year, (see Table 60) based on the recommendations from the installation company.

Table 60 Assumptions for preventive maintenance stage

Preventive maintenance		
Activity	Frequency	Quantity
Personal transport to wash surface	Twice per year	200 pkm[a]

a: the shown value has been estimated as: 1 person x 100kmx 2 trips

7.1.2.2 Data sources

The volume of material required to construct porous pavement will vary with the volume of water it is designed to treat. As explained above, in this study the treatment volume has been calculated based on the WQv, and material requirements with regard to the kind of materials have been sourced on a confidential basis from a Mexican company that produces and installs porous pavement throughout Mexico City. Additionally to material requirements, life cycle inventory data for the activities considered within the life cycle stage have been sourced from Ecoinvent (Ecoinvent, 2011) and GaBi (PE International, 2010). The background and foreground process considered in each life cycle stage are described in Table 61 and Table 62.

Table 61 Considered processes and data sources for the construction stage

Materials		
Material	*Considered processes*	*Data source*
Gravel	Whole manufacturing process for digging, transport and machinery used for operation	(Kellenberger et al., 2007)
Ballast	Whole manufacturing processes, machinery and transport	(Kellenberger et al., 2007)
Cement	Cement production	(Kellenberger et al., 2007)
Water	Water extraction, supply, transport and treatment	(Ecoinvent, 2011)
Transport		
Vehicle	*Considered processes*	*Data source*
40 tonne truck	Construction, use (including maintenance) and end of life	(Spielmann et al., 2007)
Machinery		
Machinery	*Considered processes*	*Data source*
Hydraulic digger	Materials, transport of the parts to the assembly point and energy and heat requirements during use phase.	(Kellenberger et al., 2007)

Table 62 Processes and data sources for the preventive maintenance

Preventive maintenance		
Vehicle	Included processes	Data source
Passenger car	Vehicle manufacturing, operation and disposal	(Spielmann et al., 2007)

The processes with electricity requirements, such as the material production have been modified in order to adapt them to Mexican conditions. The adaptation is the use of the Mexican electricity mix (Santoyo et al. 2011) for the electricity requirements of the processes mentioned.

Data requirements for the residual runoff stage encompass data required for the pollutant loads calculation, which has been considered as a basis for calculating pollutant mass to be treated and calculated according to Equation 6. Thus, information about pollutant concentration has been sourced from Kayhanian et al. (2007), while data sources for other parameters considered are summarized in Table 63.

Table 63 Data sources for the residual runoff stage

Parameter	Value	Information source
Drainage area	0.4 ha	MDE,2000
Annual rainfall depth	50 mm	MDE,2000
Fraction of rainfall events that produces runoff	90%	Chandler,1994
Imperviousness	100%	MDE,2000

Having calculated the pollutant load and the volume to be treated, pollutant mass treated in porous pavement is calculated based on the pollutant removal efficiencies. These removal efficiencies have been sourced from Cahill (2003) and are shown in Table 64.

Table 64 Pollutant removal efficiencies considered for mass balance (Cahill et al., 2003)

Parameter	% removal
TSS	91
Cd	33
Cr	90
Cu	42
Ni	90
Pb	74
Zn	81

7.1.3 Impact assessment

The impact assessment has been done based on the CML impact characterisation method (Guinee, 2001), while the evaluation is conducted using version 4.4 of the GaBi software (PE International, 2010). Since this is the second LCA study for porous pavement, all the following impact categories are selected: abiotic depletion potential (ADP elements), abiotic depletion potential (ADP fossil), acidification potential (AP), freshwater aquatic ecotoxicity potential (FAETP), global warming potential (GWP), human toxicity potential (HTP), marine aquatic ecotoxicity potential (MAETP), ozone layer depletion potential (ODP), photochemical ozone creation potential (POCP) and terrestrial ecotoxicity potential (TETP). All of these have been included in order to extract as much information as possible to better understand life cycle impacts of porous pavement. The interpretation of these results is also provided in this section.

7.1.3.1 Abiotic depletion potential (elements)

As shown in Figure 40, the construction stage is the stage that mainly contributes to the total of 0.02 g Sb eq./fu (75%). Of this percentage, the major contribution (50%) occurs due to depletion of non-renewable elements during production of additive.

The 25% remaining contribution to abiotic depletion potential of elements is produced by the preventive maintenance stage, whose contribution comes from the personnel transport for doing maintenance.

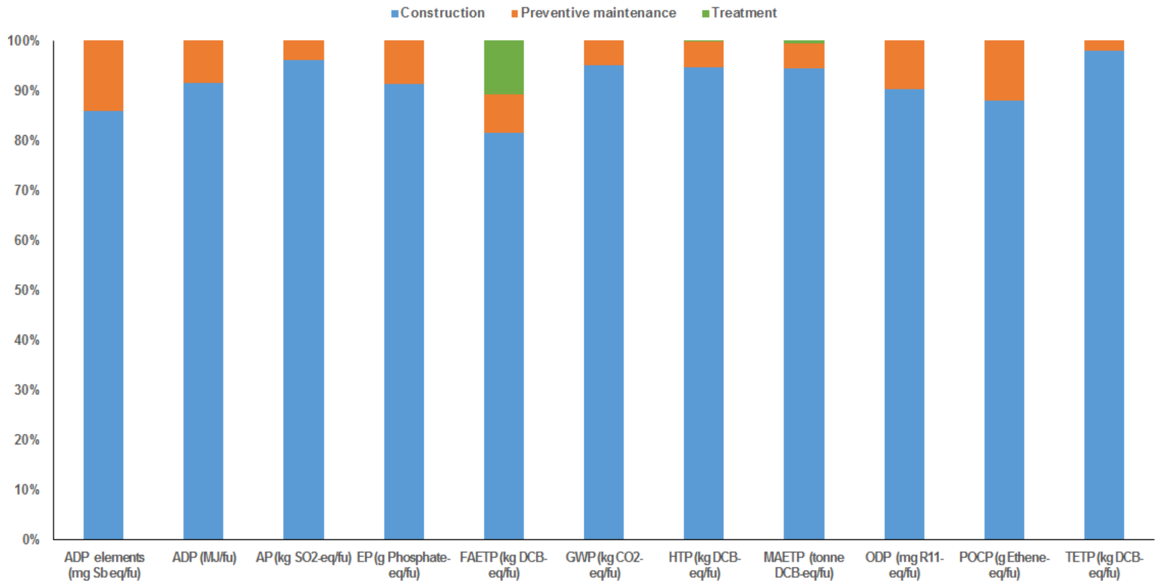

Figure 40 Contribution analysis for porous pavement

7.1.3.2 Abiotic depletion potential (fossil)

As seen in Figure 40, of the total 150 MJ/fu ADP of fossil resources consumed among the porous pavement life cycle, the largest consumption occurs during the construction stage (85%). Materials with the highest fossil resource consumption are cement and additive, accounting for 29% and 24% respectively of the construction stage contribution (see Figure 41). Of the materials required for cement production, gravel is the one used in major quantities. For this reason, the contribution of cement is mainly due to consumption of crude oil and hard coal for the electricity generation for gravel manufacture.

The transport of personnel to conduct washing of the porous pavement surface accounts for 15% of the abiotic depletion of fossil resources. Major amounts of crude oil and natural gas are consumed during this transport activity.

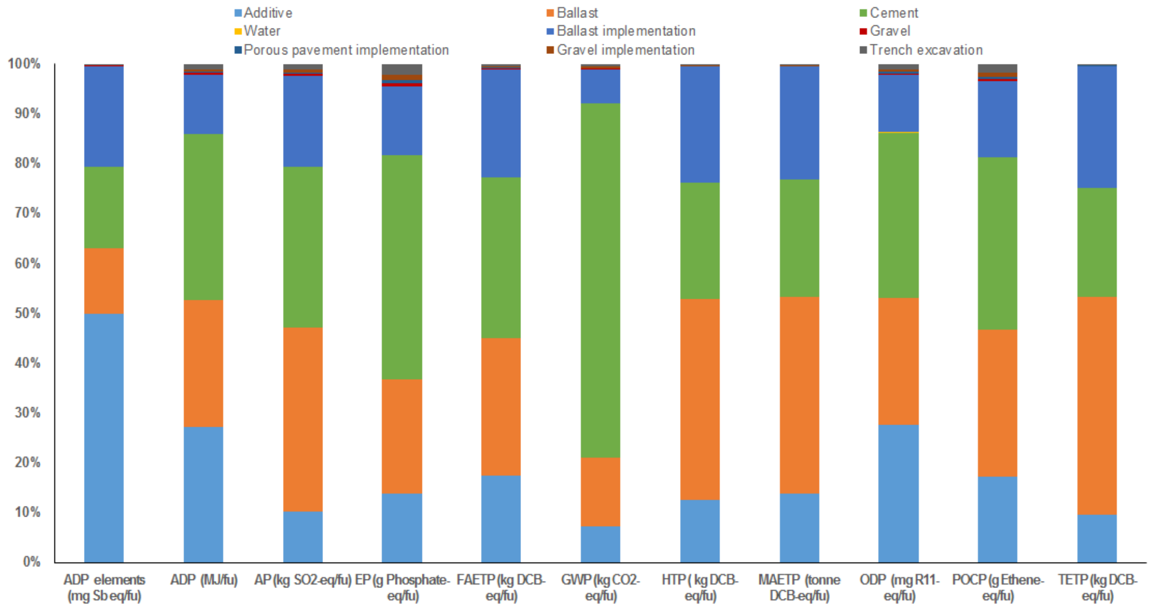

Figure 41 Contribution analysis for the construction stage

7.1.3.3 Acidification potential

The construction stage is the main factor responsible for the AP (93%) (see Figure 40), which accounts for 0.067 kg SO_2 eq./fu. As can be seen in Figure 41, the major cause of this contribution is the production of ballast (34%) and cement (29%), both required for the porous pavement manufacture. Sulphur dioxide and nitrogen oxides, which are emitted to air during the generation of electricity required for the manufacture of ballast and cement, are the main cause of their contributions to acidification potential.

Of the 7% contribution of preventive maintenance to AP, emissions of sulphur dioxide and nitrogen oxides emitted to air are the main factors responsible.

7.1.3.4 Eutrophication potential

The main contributors to total EP of 11 g phosphate eq/fu are the activities conducted in the construction stage (see Figure 40), putting in 93% of this value. This is due to cement and ballast, which represent 36% and 18% respectively of the construction stage. The contribution of cement is mainly caused by emissions to air of nitrogen oxides and by the emissions of phosphate to water generated during the production process of cement. The contribution of preventive maintenance represents about 11% of EP. Of this, the emissions responsible are mainly of phosphate added to freshwater and nitrogen oxides emitted to air.

7.1.3.5 Freshwater aquatic ecotoxicity potential

Total FAETP amounts to 1.29 kg DCB eq./fu, most of which comes due to the construction stage (76%). This is due to the cement (32%) and ballast (26%) manufacturing; these contributions are mainly caused by the emissions to air coming from the electricity generation process. The preventive maintenance stage is the second culprit of the emissions of DCB eq/fu, accounting for 14% of the total FAETP. Lastly, the remaining 10% of the FAETP is contributed by the residual runoff stage, mainly due to the emissions of copper and zinc to freshwater.

7.1.3.6 Global warming potential

Most of the total 17 kg of the CO_2 equivalent emissions/fu comes from the construction stage (91%) (see Figure 40). This contribution is caused by the cement manufacture process which contributes 66% of GWP in this stage. Within the cement manufacture process, emissions of carbon dioxide to air are the main factor responsible.

The contribution from the preventive maintenance stage represents the remaining 9% of the GWP. As in the cement manufacture it is also caused by the carbon dioxide emissions to air, which are produced during the transport of personnel for doing maintenance.

7.1.3.7 Human toxicity potential

The biggest contribution to the total HTP, which accounts for 4.56 kg DCB eq./fu, comes from the production of ballast (40%), gravel (23%) and cement (23%) required within the construction stage (see Figure 40), which accounts for 90% of the HTP. Generating emissions of vanadium and nickel to air, electricity generation accounts for 85% and 70% respectively of the ballast and gravel's contribution within the construction stage. In the case of cement, its 23% is caused by the cement production, accounting for 80%.The major emissions produced during the cement production are emissions to air of arsenic and chromium.

The remaining 10% contribution to HTP comes from the preventive maintenance stage, this is due to the emissions of chromium and arsenic to air.

7.1.3.8 Marine aquatic ecotoxicity potential

The largest contribution to the total 5004 kg DCB eq/fu MAETP comes from the construction stage (90%). As in the HTP, this is mainly caused by the production of ballast (39%), gravel (22%) and cement (23%). Most of the cement contribution, comes from the cement production (70%).Within the cement production, major emissions are the emissions of beryllium to freshwater and hydrogen fluoride to air. The electricity generation required for the manufacture of gravel and ballast accounts for 90% of the ballast contribution and 74% of the gravel contribution to the construction stage. This is due to the emissions of vanadium to air and hydrogen fluoride to water, produced during the electricity generation process.

Although small, preventive maintenance is the second largest contributor to MAETP (9%). Of this, major emissions are emissions of beryllium to freshwater and hydrogen fluoride to air.

7.1.3.9 Ozone layer depletion potential

As occurs in MAETP and HTP, cement (29%) and ballast (22%) turn the construction stage into the largest contributor (85%) to the total 1.3 mg R-11 eq/fu. (see Figure 40). A third material contributing to ODP is additive, which represents 24% of the contribution from the construction stage.

The cement contribution is due to the clinker production during cement manufacture (41%), while the contributions from ballast and additive are created by the electricity generation (43%) and carbon black production at plant (71%). The emissions to air of halon 1301 are the main factor responsible.

Since residual runoff does not contribute to the ODP, the remaining 15% of the contribution to ozone depletion potential comes from the preventive maintenance stage. As with the contribution from the construction stage, emissions of halon 1301 are causing the contribution from transport.

7.1.3.10 Photochemical ozone creation potential

The POCP is mainly due to the construction stage, which accounts for about 80%. Of this percentage, cement (31%), ballast (26%) and additive (15%) cause the largest contribution. In the case of cement, 77% of its contribution is caused by the clinker manufacture. The generation of electricity represents around 70% of the ballast contribution to this stage.

Finally, the major cause of the contribution from additive comes from the production of oil and polyethylene, accounting for 31% and 25% respectively. The second largest contributor is caused by the transport of personnel required for the preventive maintenance; which represents 20% of the POCP. This mainly originates from the emissions of carbon monoxide and non-methane volatile organic compounds (NMVOC) to air.

7.1.3.11 Terrestrial ecotoxicity potential

About fifty percent of the contribution of the construction stage (97%) of the total 0.14 kg DCB eq/fu TETP comes due to ballast. Ballast contribution is caused by the electricity generation (93%), which is required for ballast manufacture. In addition to the ballast, gravel (24%) and cement (21%) are also major contributors to the construction stage.

Electricity generation (77%) is also the major cause of the contribution of gravel to the construction stage, while clinker production (55%) required for cement is the major cause of its contribution to the construction stage. In regard to the emissions, in the case of the electricity generation major emissions are of vanadium to air, while in the case of the clinker production major emissions are emissions of chromium and mercury to air.Emissions of cobalt and mercury are the major factors responsible for the 4% contribution from preventive maintenance to TETP.

7.1.4 Sensitivity analysis

7.1.4.1 Impact of the pollutant concentration change

Even though residual runoff stage is not identified as a hot spot within the life cycle environmental impacts of the porous pavement, it has been selected for a sensitivity analysis. This selection is based on the fact that the pollutant concentration (EMC) used for the life cycle environmental assessment of the base case scenario corresponds to the mean value of the range considered for the EMC, which has been sourced from Kayhanian et al. (2007). Therefore, with the aim of analysing the effect on the environmental impacts due to the variation in the EMC, both maximum and minimum values of the range considered for the EMC are considered for the pollutant load calculation. Both drainage area and rainfall depth considered for the pollutant load calculation remain as the values presented in the section 7.1.2.2. Thus, for the purpose of this analysis, three scenarios have been defined based on the EMC value (see Table 65); which have been labelled as scenario A (SA, maximum EMC), base case (BC, mean EMC) and scenario B (SB, minimum EMC).

In addition to the variation in the EMC within the range considered by Kayhanian et al. (2007), another EMC value has been considered in this sensitivity analysis. As shown in Table 65 this value is six times higher than the maximum EMC considered. It has been included, in order to analyse what would be the effect on the environmental impacts if higher values than the one reported in the literature were used. This scenario is labelled as scenario C (SC). Lastly, a scenario without porous pavement treatment, which has been labelled as scenario W (SW), corresponds to the effect on the environmental impacts if polluted water did not receive treatment.

Table 65 EMC values considered for each scenario

Parameter (mg/l)	(SA)	(BC)	(SB)	(SC)	(SW)
Cd	0.03	0.007	0.002	0.18	0.03
Cr	0.09	0.009	0.001	0.54	0.09
Cu	0.27	0.034	0.001	1.62	0.27
Ni	0.13	0.011	0.001	0.78	0.13
Pb	2.60	0.047	0.001	15.6	2.60
Zn	1.68	0.187	0.005	10.1	1.68

The results show that EMC variation in the pollutant load calculation affects only two impact categories, which are freshwater aquatic ecotoxicity potential (FAETP) and marine aquatic ecotoxicity potential (MAETP). Results for these impact categories for the four scenarios previously defined are described in this section.

7.1.4.1.1 Freshwater aquatic ecotoxicity potential

As is shown in Figure 42, comparing the emissions of DCB eq/fu from the SA and the SW, a reduction of 25% is observed. Thus, although during the life cycle stages considered for the porous pavement emissions of DCB eq/fu are produced, they are less than those produced if runoff did not receive any treatment.

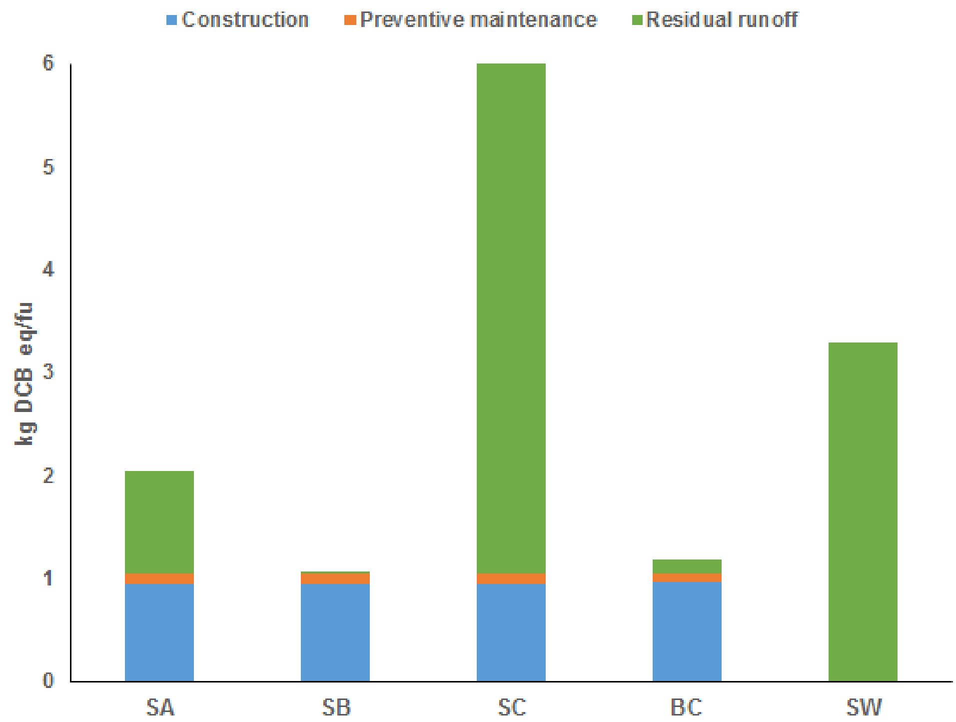

Figure 42 FAETP for different scenarios

The results show that FAETP from SA (2.05 kg DCB eq/fu) is about 70% higher than FAETP from BC (1.18 kg DCB eq/fu) (see Figure 42), while the variation between BC is 26% less than the BC.

Since the contributions from the construction and preventive maintenance stages are not affected by the EMC variation in the pollutant load calculation, the cause of this result comes only from the contribution from the residual runoff stage. Hence, it is found that the greater the EMC the greater the contribution from the residual runoff stage.

7.1.4.1.2 Marine aquatic ecotoxicity potential

As shown in Figure 43, MAETP varies within the range from 4663 kg DCB eq/fu (SB) to 4920 kg DCB eq/fu (SA). The difference within these results comes from the contribution from the residual runoff stage. On the other hand, when the SA and the SC are compared an increase of 21% in the MAEPT is found.

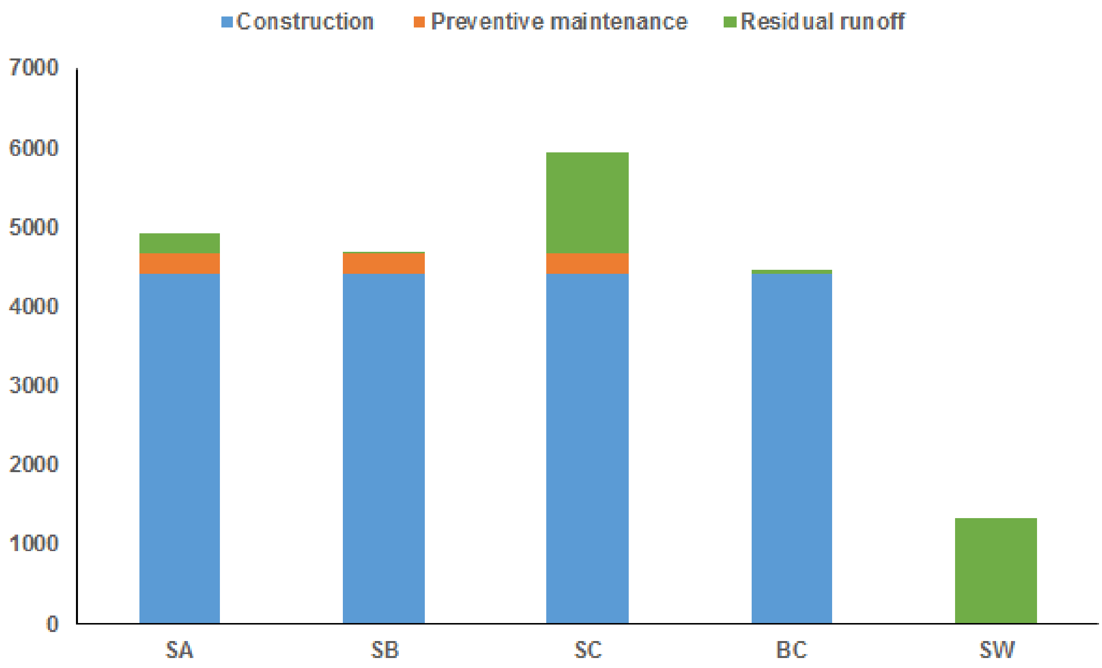

Figure 43 MAETP for different scenarios

This difference is due to the contribution from the residual runoff stage, which as shown in Figure 43 is the only one affected due to the increase in the EMC. Finally, the results from the SW and SA show that the runoff treatment leads to an increase in the MAETP. This is explained because, as shown in Figure 43, the contribution from the construction stage creates more emissions of DCB eq/fu than are contributed from the untreated runoff.

As described in this section, the effect of the variation of EMC in the pollutant load calculation affects two impact categories: FAETP and MAETP. This is due to the contribution from the residual runoff stage, which is increased as far as the EMC increases.

In the comparison between the SW and SA, it is observed that on the one hand, the FAETP is reduced due to the installation of porous pavement, however on the other hand there is an increase in MAETP. For this reason, porous pavement installation should be cautiously considered as a treatment option for controlling diffuse pollution.

7.1.4.2 Impact from the variation in the treatment volume

As mentioned before, the design of the porous pavement is based on the water quality volume (MDE, 2000), which is calculated according to Equation 4 based on the rainfall depth of a specific design storm. The design storm recommended by the Maryland manual is 10-year return period[7], 24 hours duration design storm. Since this parameter is geographically variable, it might lead to a variation in the environmental impacts. For this reason, rainfall depth has been analysed as part of the sensitivity analysis in this work. The variation that has been considered ranges from a rainfall depth of 50 mm to the 250 mm of the design storm, which due to the lack of data for the MRC has been assumed on an arbitrary basis.

The first part of the analysis is done considering the average EMC and 250 mm rainfall depth (referred to as BC2 in this work) thus it can be compared with its counterpart BC, which is calculated with the rainfall depth of 50 mm for the same design storm. The second part of this sensitivity analysis is carried out considering 250 mm rainfall depth and maximum (SD) and minimum EMC (SE) values. Thus, a comparison is made among these scenarios and the scenarios with the same EMC (SA and SB) and a treatment volume calculated with 50 mm rainfall depth of the previously defined design storm.

Except for MAETP and FAETP the results from the life cycle impacts and stage contributions are the same for scenarios (A-B) and (D-E). Therefore, results for the rest of the impact categories are grouped and shown in the first section, while MAETP and FAETP are described for scenarios (A-B) and (D-E).

A reduction in the overall life cycle impacts generated due to the rainfall depth variation is observed. This pattern is observed in all the impact categories analysed in this work. The reason for this is that the larger the

[7]10 year-24 hour refers to 10 year return period, 24-hour duration storm - a storm lasting 24 hours and depositing this amount of water takes place on average once every 10 years in the specified location (in this case, Maryland).

treatment volume, the smaller the impact. Despite the inverse relationship between the volume treated and the effect on the impact categories, there is an increase in the contribution from the construction stage, and a corresponding reduction in the contribution from the preventive maintenance stage is also observed.

7.1.4.2.1 Abiotic depletion potential (elements)

The total abiotic depletion of elements due to the increase in rainfall depth accounts for 5.47 mg Sb eq/fu. This represents a decrease of about 63% in comparison with the ADP of elements from BC. This reduction occurs as a result of the increase in the treatment volume, as already explained.

In the contribution analysis from the BC, the construction stage contributes about 75% of the total ADP of 15 mg Sb eq/fu. By contrast, the contribution from the same stage contributes about 86% of the total ADP from the BC2. This increase generates a decrease in the contribution from the preventive maintenance stage; this contribution accounts for 25% of the BC, while it is reduced to 14% in the BC2.

7.1.4.2.2 Abiotic depletion potential (fossil)

The abiotic depletion of fossil resources for the BC is 150 MJ/fu, 42% less than the corresponding value of fossil ADP from BC2. This reduction is due to the increase in the treatment volume.

From BC, as already explained, the construction stage contributes 85%, while in the BC2 this stage contributes 95% of its ADP value. The contribution from the preventive maintenance stage is also affected, since for the BC this accounts for 15% of the ADP value, while for the BC2 it accounts for 5%.

7.1.4.2.3 Acidification potential

The acidification potential is slightly affected due to the variation in the treatment volume, the AP from the BC of 0.07 kg SO_2 eq/fu is reduced by 23% for the BC2. In the contribution analysis, the contribution from the construction stage is 93% for the BC; and due to the increase in the treatment volume this contribution increases to 98% of the AP in the BC2.

The preventive maintenance contribution is therefore reduced, since for the BC it represents 7% and for the BC2 it represents only 2% of the AP.

7.1.4.2.4 Eutrophication potential

The eutrophication potential accounts for 11 g phosphate eq/fu for BC; due to the increase in the treatment volume it is reduced by 43% in the BC2. However, despite this reduction in the eutrophication potential life cycle impact, the contribution analysis shows that the increase in material requirements leads to an increase in the contribution from the construction stage, which for BC2 represents 95%. This contribution is 84% for the case of the BC.

As is shown the effect from increasing the treatment volume through the increase in the rainfall depth does not have a strong impact on the contribution from the construction stage. Therefore, the pattern in the contribution from the preventive maintenance is similar, since its contribution from BC is 14% while its contribution from BC2 is 5%.

7.1.4.2.5 Global warming potential

The global warming potential of the BC2 corresponds to 56% less than the global warming potential from the BC of 17.5 kg CO_2 eq/fu. Of this GWP for the BC, the contribution from the construction stage represents 91% and due to the increase in the treatment volume it increases to 96% in the BC2. Also, due to the increase in the treatment volume, the contribution from the preventive maintenance stage for the BC is reduced from 9% to 4% for BC2.

7.1.4.2.6 Human toxicity potential

That the treatment volume is increased decreases the HTP from the SD in 22% in comparison with the HTP from the SA. While 90% of the HTP from SA corresponds to the construction stage, this contribution is 97% in the case of the SD. The remaining 10% of the contribution to HTP of the SA comes from the preventive maintenance stage, since the contribution from the residual runoff stage is negligible.

Due to the increase in the contribution from the construction stage in the BC2, the contribution from the preventive maintenance stage is reduced from the given 10% to 3%.

7.1.4.2.7 Ozone layer depletion potential

The increase in the treatment volume leads to 43% reduction of the ODP from the BC of 1.3 mg R-11 eq. /fu. This value corresponds to BC2 and its major contributor comes from the construction stage (95%) while the second contributor is the preventive maintenance stage accounting for the remaining 5% of the ODP. In the contribution analysis of the BC, it can be seen that the contribution from the construction stage is 10% lower than in the case of the BC2, representing 85% of the total ODP. Since the residual runoff does not contribute to the ODP the remaining 15% comes from the preventive maintenance stage.

7.1.4.2.8 Photochemical ozone creation

POCP of 10 g ethane eq./fu from the BC is reduced by 40% in the case of the BC2. Eighty-one percent of the POCP from the BC comes from the construction stage, this contribution is increased to 94% in the BC2. Due to this increase there is a decrease in the contribution from the preventive maintenance stage from the BC, which accounts for 10% of the POCP; this contribution decreases to 6% of the POCP from the BC2.

7.1.4.2.9 Terrestrial ecotoxicity potential

Lastly, the BC2 represents a reduction of 11% in the TETP of 0.13 kg DCB eq/fu of the BC. As in the rest of the impact categories described in this section, the construction stage is the largest contributor to BC, accounting for 96% of its TETP. In this impact category, the increase from the contribution of the construction stage in the BCS is almost negligible, accounting for 3% more of the contribution in the BC. Consequently, the reduction in the contribution of the preventive maintenance stage in the BC is also reduced by 3% in relation to its contribution to BC2.

7.1.4.2.10 Freshwater aquatic ecotoxicity and marine aquatic ecotoxicity potential

The effect of increasing treatment volume leads to a decrease in FAETP and MAETP; however these two impact categories have been grouped because their variations are affected by the EMC variation.

In regards to FAETP, the reduction corresponding to the comparison among the SA and the SD is 67% (see Figure 44). This reduction occurs due to the increase in the treatment volume. By contrast, this variation in the treatment volume is responsible for the increase in the contribution from the construction stage. In the case of the SB and SE, where minimum pollutant concentration is considered, construction stage contribution tends to

dominate the contribution analysis. This is because as the pollutant concentration decreases, the contribution from the residual runoff increases.
A pattern is found from this result, that when the contribution from the residual runoff decreases, the contribution from the construction stage dominates the contribution analysis. The cause of the increase in the contribution from the construction stage is due to the increase in the requirements of construction materials, which comes as a consequence of the increase in the treatment volume.

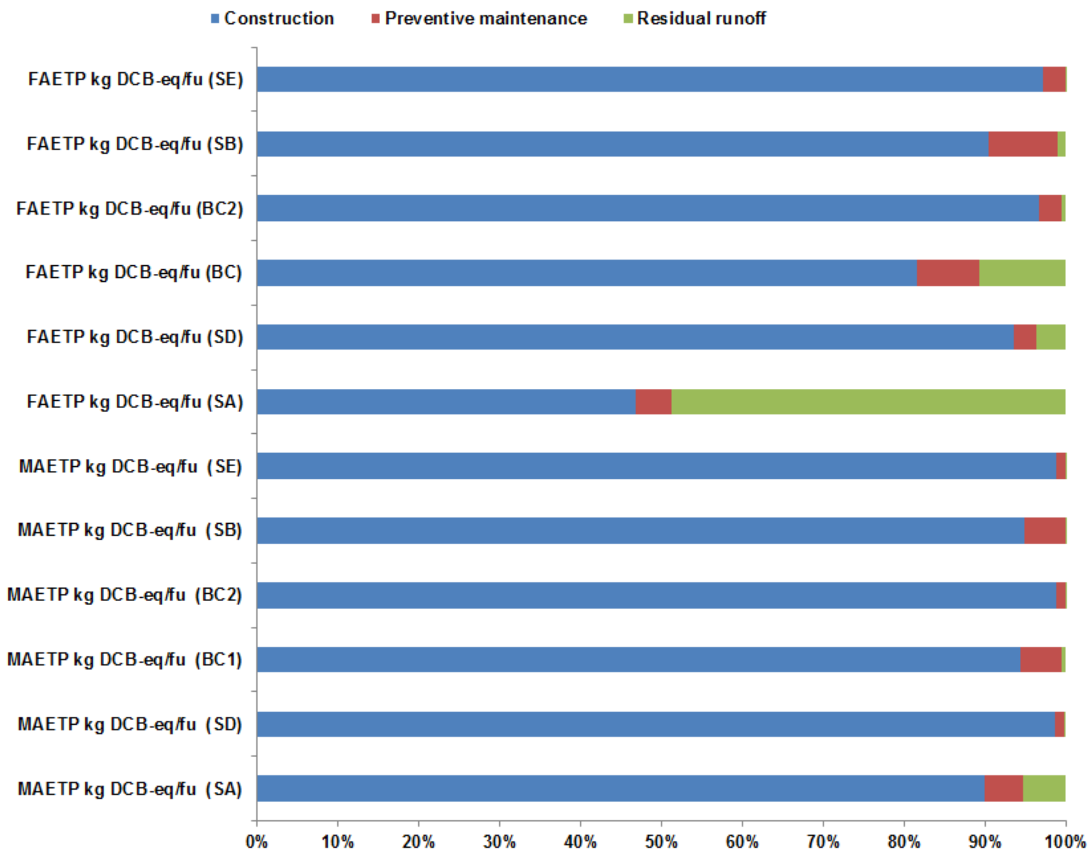

Figure 44 Contribution analysis for MAETP and FAETP

Lastly, in the case of the pollutant concentration, variation with the same treatment volume leads to a small FAETP variation. In this case, FAETP ranges from 0.71 (SD) to 0.68 (SE) kg DCB eq/fu within the pollutant concentration range considered. This is explained because the contribution from the residual runoff stage is reduced due to the increase in the treatment volume (see Figure 44). Therefore, it can be said that FAETP tends to remain constant.

The MAETP is also reduced when the SA and SD are compared, this reduction amounts to 25%. In both cases, the contribution from the residual runoff stage is reduced as long as the pollutant concentration is reduced (Figure 44). In the SD, the highest contribution of the 3904 kg DCB eq./fu comes from the construction stage (98%), while the contribution from the residual runoff only accounts for 0.17% of the MAETP.

In the evaluation of the effect of the pollutant concentration variation, the variation in MAEPT is small. In fact, as shown in Figure 44, MAETP remains constant between the BC and the BC2. This is because the contribution from the residual runoff stage is decreased when the pollutant concentration decreases.

As a conclusion of this section, based on the results, the variation of MAETP and FAETP depends on the variation of the residual runoff stage. This stage varies in relation to the pollutant concentration, therefore when this parameter is reduced a reduction in the contribution from the residual runoff stage is expected. The increase in the treatment volume leads to an increase in the contribution from the construction stage and the reduction in the FAETP and MAETP life cycle impacts.

7.1.5 Validation of the results

There is only one study (Critten, 2009) that has analysed the life cycle environmental impacts of porous pavement. Although it has not analysed porous pavement as a treatment option, its results are used as a reference for the validation of the results obtained in this study. An analysis of the aim of this study as well as its system boundaries, functional unit, life span and assumptions is given in order to explain the discrepancies with the results obtained in the present work.

The study presented by Critten et al. (2009) presents the GHG impact originated by the porous pavement. It does not present other impact categories, since the aim of the work is to calculate the reduction in the GHG emissions from the waste water treatment plant due to the flow reduction produced by porous pavement installation. Thus, GHG impact accounts for 105 tonne CO_2 eq/ha of permeable area over 25 years life span. In order to compare this result with the one obtained in this study, the latter has been converted considering the area of permeable pavement calculated in the construction stage. Thus, this value is smaller than the GWP obtained in this work, which accounts for 225 tonne CO_2 eq/ ha permeable area over a 25 year life span. It has to be mentioned that in order to obtain this value the calculation is done assuming a 25 years life span.
The difference between these values might result from the embodied burdens of the materials used during the construction stage. In this work, the entire life cycle of the material has been considered (see Table 61), while in the work conducted by Critten et al. (2009), it is not specified until what point in the life cycle the embodied burdens are calculated. Quantities of materials used during the construction stage are obscured, since they are not presented in the work; therefore it is not possible to compare the contribution per material in order to identify which material might create the difference in the GWP; which might be another reason for the difference in the GWP values.

In regard to the results obtained from the treatment volume variation that comes from the rainfall depth variation, a pattern is found. In the comparison between the results from the increase in the treatment and base case scenario, a reduction for all the impact categories resulted. Although it could be thought that higher treatment volumes could lead to higher environmental impacts due to the high corresponding size of the porous pavement area, it is exactly the opposite. Thus, it is established that the higher the volume treated the smaller the environmental impact. Hence, there is an offset between the volume treated and the generated environmental impact.

The highest reductions occur in the GWP and ADP elements, they account for 56% and 43% respectively in comparison with the SA. Additionally, in the case of the pollutant concentration variation, the highest reduction occurs for MAEPT, which is reduced by 25% for the SD when it is compared to their corresponding counterpart in SA.

It is worth mentioning that even though this is the first time treatment has been considered as the main function provided by the porous pavement, the pattern found in this work is not dependent on the selected function. For example in the case of Critten et al. (2009), where permeable area is considered as the functional unit, results for GWP would be smaller to the extent that the permeable area is increased.

The decision whether or not to install porous pavement should depend on taking into consideration not only environmental but also social and economic aspects of such installation. However, based on the environmental impacts analysed in this work, the installation of porous pavement is not recommended, since it might pose risks to human health due to the emissions coming from the construction stage. Finally, it has to be said that the results are only valid for the assumptions considered, the discrete rainfall events and concentration ranges considered in this work.

7.1.6 Summary

Based on the assumptions and the system boundaries considered in this work, the conclusions reached during this analysis are described in this section. This description is divided into three sections; the first part is aimed at the conclusions of the results from the base case scenario. The second and third parts are dedicated to the analysis of the variation of the pollutant concentration (EMC) and treatment volume considered in the sensitivity analysis.

In regard to the base case scenario the following conclusions are reached:

Construction stage is the major contributor, amounting on average to 92% for all the impact categories under analysis. Within the construction stage, cement production is the largest contributor to ADP fossil, EP, FAETP, GWP, ODP, POCP and TETP; in the case of AP, HTP and MAETP the largest contributor in the construction stage is the ballast production, finally for ADP element, the largest contribution comes from the additive production.

The pollutant concentration used as part of the analysis might vary in relation to the activities conducted in the catchment area. As explained in Chapter 4, this information is not available for the case of Magdalena river catchment. Therefore, this information is supplied from an alternative source which does not provide a current value for the pollutant concentration. For this reason, in order to analyse the effect of the variation of this parameter, the range of pollutant concentrations is considered and compared against the average pollutant concentration value. The result from this analysis show that FAETP varies from 1.18 to 2.16 kg DCB eq/fu, and MAETP varies from 4690 to 4920 kg DCB eq./fu. Although there is a variation, the effect in HTP is considered negligible, since this impact category varies from 4.57 to 4.61 kg DCB eq/fu. Finally, the increase of up to six times (SC) the highest pollutant concentration value given in literature, leads to an increase in FAETP and MAETP from the SA. This increase is from 2.16 to 6.14 kg DCB eq for FAETP for SA and SC respectively. Moreover, in regard to MAETP this increase leads to an increase of 120% from the SA of 5232 kg DCB eq/fu corresponding to SC. Therefore, it can be concluded that in the hypothetical case of increasing the maximum pollutant concentration, both FAETP and MAETP results would be affected.

In the comparison between the cases with and without the installation of porous pavement, it is not possible to reach a direct conclusion. This is because the installation of porous pavement leads to a reduction in FAETP; but it also increases MAETP and HTP. In the case of FAETP, there is a reduction of 38% due to the installation of the porous pavement system. On the other hand, there is an increase of 291% for MAETP and 1492% for HTP due to the installation of the porous pavement.

This increase is caused by the emissions produced during the construction stage, while the reduction of the FAETP is generated by the reduction of the emissions from the residual runoff stage. For this reason, it is not possible to reach a clear conclusion on the environmental advantages of the porous pavement selection as a treatment option.

7.2 Economic evaluation

The estimation of the cost of porous pavement has been analysed in more detail than the estimation of the environmental impacts derived from this treatment practice. This may be because it entails management of money, which is usually an important parameter when considering different treatment options. Thus, porous pavements cost estimation has been based on individual estimations of the construction and maintenance of the porous pavement (Taylor, 2005; SWRPC, 1991). However, this cost estimation might underestimate other costs along the life cycle of the porous pavement. Therefore, life cycle costing has been analysed as an option that would allow the estimation of all the relevant costs that take place along the life cycle of the porous pavement (Taylor & Fletcher, 2007). The current work is aimed at increasing the knowledge about the use of life cycle costing for the estimation of the cost of porous pavement, since it has been calculated based on the environmental approach for life cycle costing established by the Society of Environmental Toxicology and Chemistry (Swarr et al., 2011). The results of this estimation are shown in this section.

7.2.1 Goal and scope

The goal of this study is to identify and quantify the economic “hot spots” along the life cycle of the porous pavement. The stages considered in this life cycle are construction, preventive maintenance and operation and they have been defined from the parallel LCA study above described (see Figure 39). Moreover, the scope has been also defined based on the LCA study, thereby considering them from cradle to grave. Finally, as established by Swarr et al. (2011) the LCC is calculated per functional unit, which according to the LCA has been defined as “treatment of 1 m^3 runoff over 30 years”.

7.2.2 Inventory analysis

7.2.2.1 Assumptions

Due to its consistency with the life cycle approach, life cycle cost (LCC) has been estimated based on the environmental approach described by Swarr et al. (2011). The calculation includes the cost of each of the life cycle stages previously defined in the LCA study, which are described in Figure 39. These stages are: construction, preventive maintenance and residual runoff, however residual runoff is not considered, since within this stage runoff crosses the porous pavement structure without creating any cost. In regard to both construction and preventive maintenance stage, calculated cost includes:

- Construction stage: Capital cost, cost of material, material handling considering both machinery and labour costs and material transport;
- Preventive maintenance: Cost of washing the porous pavement surface.

7.2.2.2 Data sources

Quantities used for the cost calculation have been taken from the life cycle inventory of the LCA study (see Table 66 and Table 68), while data sources for labour, machinery and materials have been sourced from a Mexican construction cost database generated by the Mexican Institute of Cost Engineering (Gonzalez, 2012).

The LCC results are shown in the following section and they are presented in American dollars in order to facilitate comparison with related studies.

Table 66 Cost of the construction stage

Construction stage	
Material installation cost	
Activity	Cost (US$)
Trench excavation	733
Ecocreto installation	276
Gravel installation	172
Ballast installation	285
Material cost	
Material	Cost (US$)
Cement	4540
Gravel	1171
Ballast	1591
Additive	1316
Water	1
Transport cost	
Material transport	Cost (US$)
Cement	58
Gravel	492
Ballast	286
Additive	3

Table 67 Cost of the preventive maintenance stage

Preventive maintenance stage	
Activity	Cost (US$)
Porous pavement washing	3866

Table 68 Activities per stage considered for the LCC

Construction stage		
Material installation	*Material transport*	*Material*
Trench excavation	Cement	Cement
Porous pavement installation	Gravel	Gravel
Gravel installation	Ballast	Ballast
Ballast installation	Additive	Additive
		Water
Preventive maintenance		
Porous pavement washing		

7.2.3 Results and discussion

Total LCC amounts to \$8/f.u, this is mainly due to the construction stage, which accounts for about 74% of this cost. Most of the cost from the construction stage comes from material costs (78%), while another 13% comes from the machinery use and the remaining 9% results from the material transport. The remaining 26% of the LCC is entirely due to the preventive maintenance stage.

These results show that due to material cost, the construction stage is the hot spot of the LCC; however, since the materials are fixed in the porous pavement “recipe”, it is not possible to conduct a sensitivity analysis in this sense. However, because this LCC estimation is carried out in parallel with a LCA study, the sensitivity analysis has been carried out in order to analyse the effect in the LCC due to the increase in the treatment volume as in the LCA study. It has to be mentioned that although the LCA study also considers the sensitivity analysis due to the variation in the pollutant concentration, this case has not been considered in the LCC estimation since it does not incur in any variation in the cost.

In addition to the variation in the treatment volume, another parameter is also analysed in the sensitivity analysis. This parameter is the variation of the discount rate in the estimation of the net present value. The net present value is calculated, since Swarr et al. (2011) point out that if the life span of the porous pavement is longer than two years, then the variation of the money in time should be considered. Thus, the results of these sensitivity analyses are described further below.

7.2.4 Sensitivity analysis

7.2.4.1 Analysis of the treatment volume

The LCC amounts for $2.3/f.u which is mainly due to the cost of the construction stage (88%) while the remaining 12% comes as a consequence of the preventive maintenance cost. When this cost is compared with the one calculated in the base case, a reduction of about 4 times is observed (see Figure 45). This reduction is explained due to the fact that the increase in the rainfall depth leads to an increase in the treatment volume, which is why the higher the treatment volume, the lower the cost.

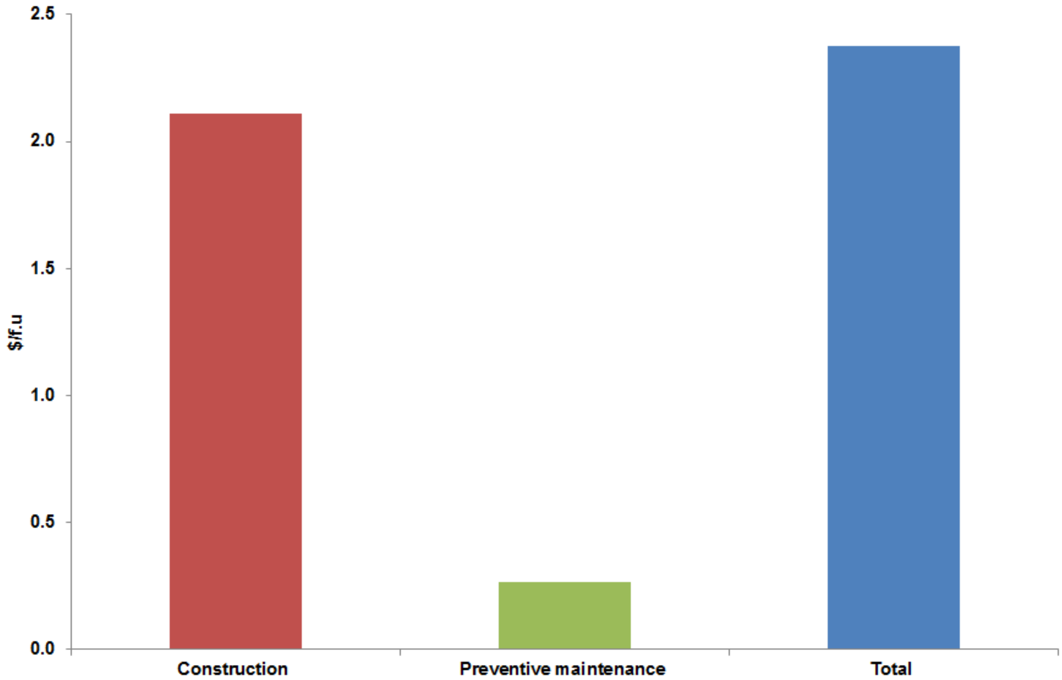

Figure 45 Effect in LCC due to the variation in the treatment volume

7.2.4.2 Analysis of the discount rate

The NPV is initially calculated with a discount rate of 12%, which is the one corresponding to Mexico (SHCP, 2009). However, in order to analyse what would happen if different discount rates were considered, this parameter has been varied in order to compare the calculated discounting future cost to the present in the UK and the U.S. Thus, the discount rate has been varied from 5.5% to 12%, since according to Lampe et al. (2005) it has been considered as 5.5%. As shown in Figure 46, the results from this sensitivity analysis have proven that the discount rate of 5.5% leads to a reduction of 37% in relation to the calculated LCC of the base case (US$8), while the discount rate of 12% decreases the LCC of the base case by 61%.

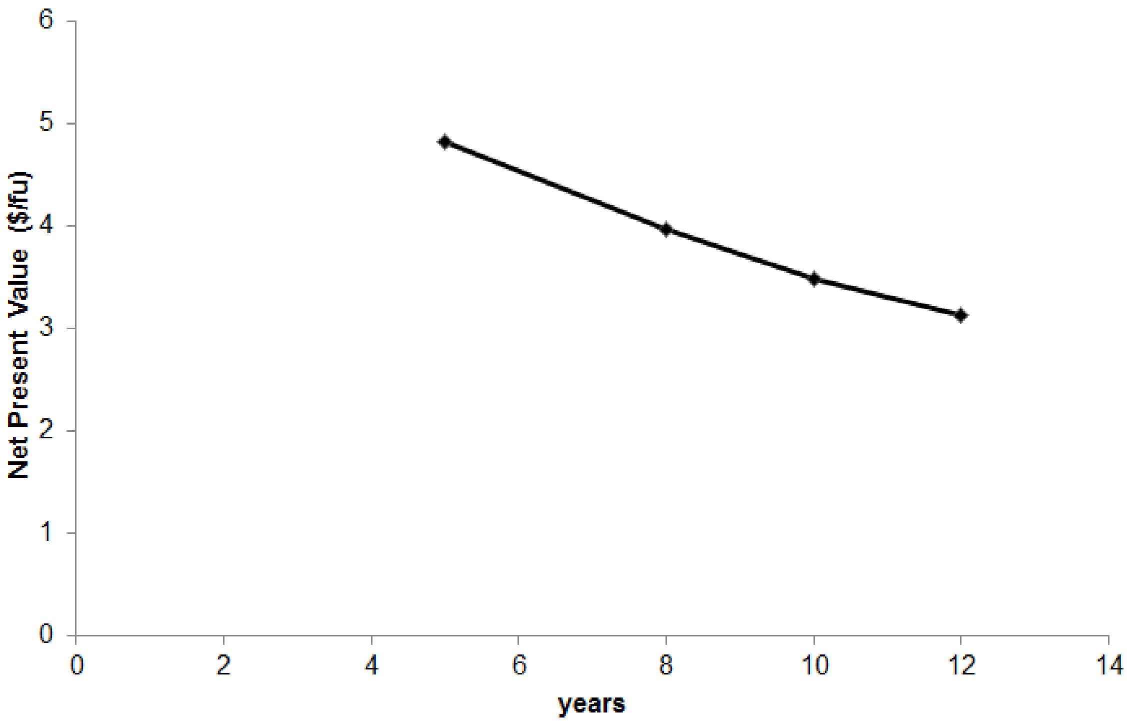

Figure 46 Effect in LCC due to variation in discount rate

7.2.5 Validation of the results

The results of the LCC estimated in the current study are compared with the results obtained using the spread sheet developed by Lampe et al. (2005). This estimation is presented as an aggregated value and not per functional unit, therefore the comparison is carried out considering absolute values. As shown in Table 69, the LCC calculated in this study is about half the LCC estimated by Lampe et al. (2005).

Table 69 Comparison per cost element

Element cost	Lampe et al. (2005)		This study	
	Absolute	%	Absolute	%
Capital cost	$22,930	75	$10,926	74
Preventive maintenance	$7,410	25	$3,865	26
Total	$30,340	100	$14,791	100

As shown in Table 69, this difference in the results is because the capital costs calculated in the current study do not include planning, design and land costs due to a lack of data. Although an attempt was made to obtain a general estimation of the cost of the design of porous pavements, this information was not disclosed by the manufacturer due to confidentiality. As to the difference in the cost of the preventive maintenance stage, this comes mainly from the salaries of the workers, which in the case of Lampe et al. (2005) amounted to $40/hour and for the present study to $3.8/hour.

Furthermore, in the present study the only activity considered is porous pavement housing, while in Lampe et al.'s (2005) estimation both inspection and litter removal are considered, which represents a higher requirement of hand labour and therefore higher cost for preventive maintenance than the one estimated in the present study.

Table 70 Comparison per cost element per different LCC approaches

Element cost	Lampe et al. (2005)	Present study
Capital cost	Planning and site investigation Design and project/site supervision cost Clearance and land preparation Material cost Construction cost Planting and post construction Cost of land	Material cost Material transport Material installation
Preventive maintenance	Inspection, reporting and information management Litter and minor debris removal	Porous pavement hosing
Corrective maintenance	Remove existing pavement and aggregate Wash and/or replace and reinstall	Not considered

7.2.6 Summary

The estimated LCC in the current study, which amounts to \$8 /f.u., is due to the cost of the cost of the construction stage (74%) and preventive maintenance stage (26%). However, these results need not be taken cautiously, since they are valid within the assumptions considered in the current study. As proven in the validation of these results, the estimated LCC can vary in relation to four factors: life cycle stages considered in the estimation of the LCC, activities considered in each life cycle stage, the frequency with which these activities are carried out and the salaries established in the area where porous pavement is being installed.

In addition, during the sensitivity analysis it is found that the increase in the treatment volume leads to a reduction in the estimated LCC, thus due to an increase of five times the treatment volume, LCC is reduced from \$8/f.u to \$2/fu. This is explained since the treatment volume is related inversely to the LCC, thus, these results show that the higher the treatment volume, the lower the LCC.

Finally, the estimation of the net present value and the analysis of different discount rates illustrate that this factor is a key factor if the variations of future discounting costs are going to be considered in the LCC estimation. Thus, the results of this analysis show that the higher the discount rate the lower the LCC. Thus if a discount rate of 12% is considered, the NPV amounts to \$2/fu, while if a discount rate of 5.5% is considered, the result is \$3/fu. Therefore, special attention should be given to this parameter in the LCC estimation.

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8 Overall sustainability evaluation of the treatment practices to control diffuse water pollution

This chapter compares the three structural options for the treatment of diffuse water pollution considered in this work: bio-retention unit, infiltration trench and porous pavements. They are first compared on the environmental sustainability based on the findings of the LCA studies, and then on their economic sustainability, using the results of the LCC analysis. This is followed by a discussion of their social sustainability. Finally, the three options are ranked on their environmental, economic and social performance in an attempt to identify the most sustainable practice for treating diffuse water pollution.

8.1 Environmental sustainability

The life cycle impacts of the three options are compared in Figure 47. As can be seen, the bio-retention unit has the lowest and the infiltration trench the highest impacts for most impact categories. The reason for this is that the bio-retention unit uses compost as the main construction material, while the infiltration trench requires stones which also need to be replaced periodically, which pushes up its environmental impacts. Porous pavement requires concrete thus causing higher impacts than compost, making it the second worst option after the infiltration trench. However, it should be borne in mind that pavements are inherent to urban spaces, so that environmental impacts are being generated every time a pavement is installed. As explained in Chapter 7 (see section 7.1.2.1), in this study all the environmental impacts are attributed to water treatment provision, with no impacts allocated to the pavements, since porous pavement design has been conducted on this basis. If, however, porous pavements were installed as a replacement of conventional pavements, the impacts would change (reduce), since the function provided would change from runoff treatment to pedestrian mobility.

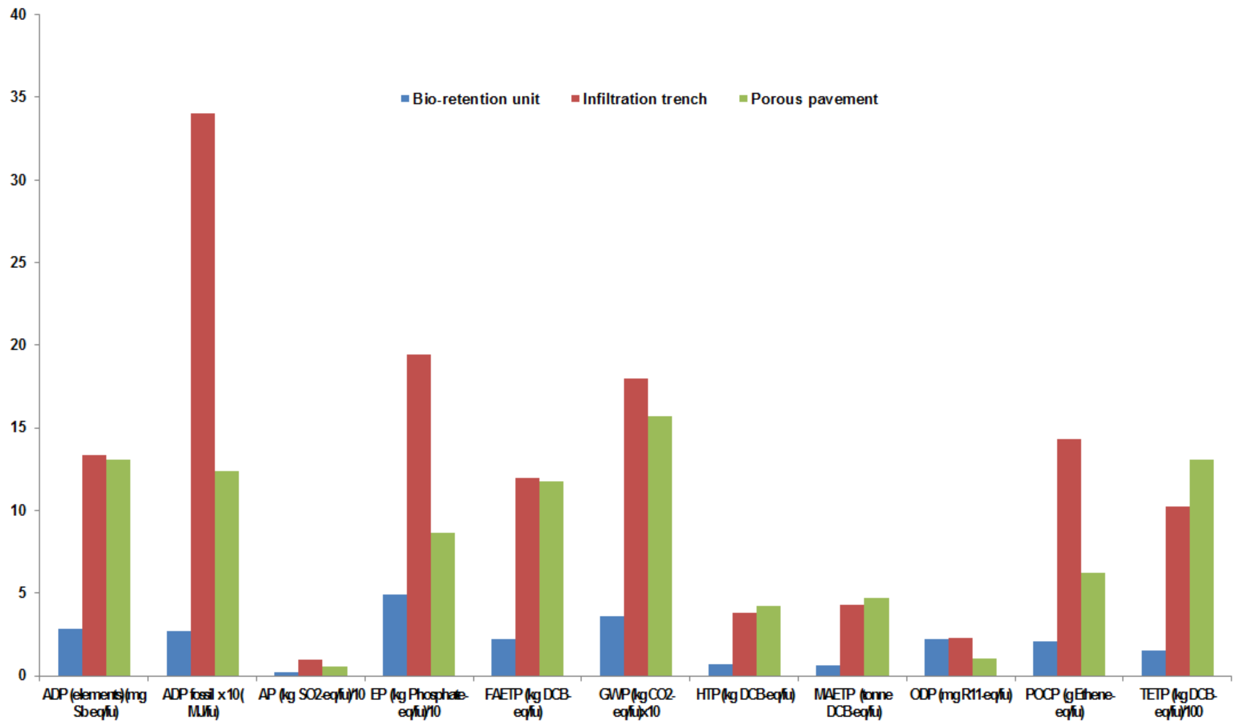

Figure 47 Comparison of the life cycle environmental impacts of the three treatment practices

8.2 Economic sustainability

The economic sustainability of the options is compared in Figure 48, based on their life cycle costs. As indicated, in contrast with the environmental impacts, the porous pavement is the cheapest option with the LCC of $8/m^3. The infiltration trench remains the worst option for the economic costs with $65/m^3 and the bio-retention unit is the second best option at $17/m^3.

The porous pavement is the cheapest option, mainly due to low maintenance requirements compared to the other two options. As explained in Chapter 7, the maintenance activities are aimed at preventing clogging, and in the current study water hosing is assumed every six months for these purposes. However, other preventative activities such as vacuuming and salt application during winter (Cahill, 2003) could increase not only the costs but also the environmental impacts. Furthermore, this activity could be noisy, thereby disturbing people around the cleaning area.

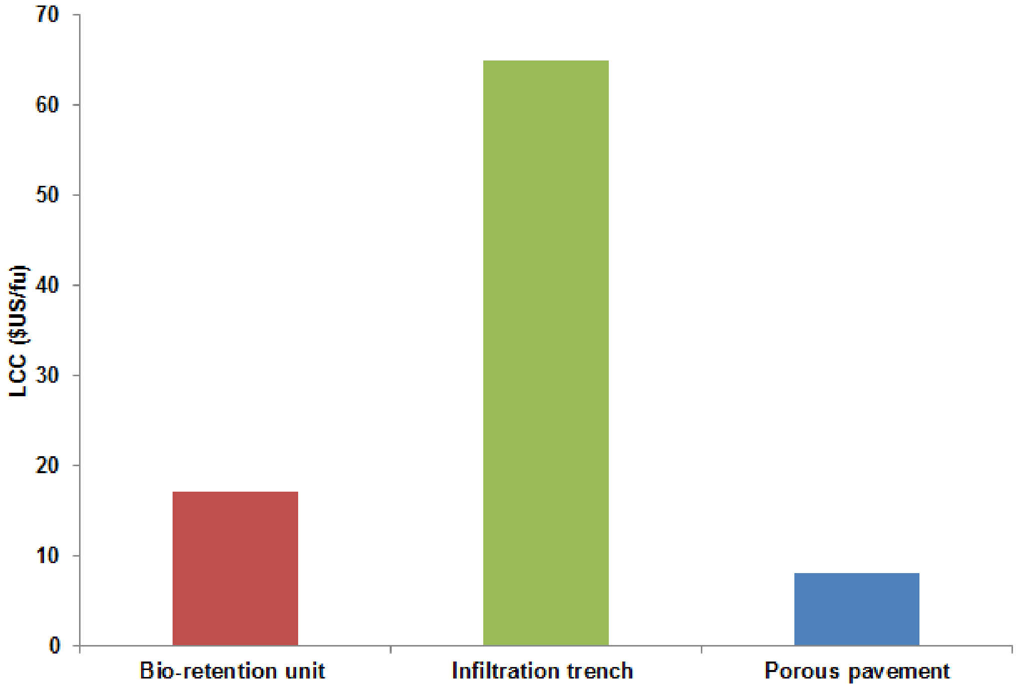

Figure 48 Comparison of the life cycle economic costs of the three treatment practices

On the other hand, the infiltration trench has high maintenance requirements associated with the stone replacement and disposal. Therefore, although the infiltration trench is a simple option, comprising a trench lined with geotextile membrane and filled with stone, it is precisely the stone handling in the life cycle of this practice that makes it the most expensive option. If manual labour were to be used instead of the machinery for stone handling, both the economic and environmental impacts would be reduced. Although the possibility of using manual labour depends on the local construction regulations, this is a potential measure to take advantage of the simplicity in the design of the infiltration trench.

The bio retention unit, like the infiltration trench, needs sludge removal in order to avoid clogging of the unit. Therefore the cost due to this activity could not be reduced unless sludge is handled manually. Thus, under the assumptions of this study, this might be one option to reduce the LCC of the bio-retention unit. Another option for reducing the costs would be by reducing the cost of the mowing, which could be carried out using a manual mower instead of the gas powered mower, which has been considered in this study. In both cases, the reduction in the costs of the infiltration trench and the bio-retention unit would increase social costs, replacing machinery with manual labourers who would probably be earning minimum wages. However, as discussed below, in areas with low income this might represent employment opportunities for the local communities.

8.3 Social sustainability

As explained in Chapter 3, the analysis for the social impacts of the analysed treatment practices has been conducted on a qualitative basis. The social impacts relevant to these three practices are discussed below (see section 3.5.3 for the criteria considered in the selection of the social impacts).

8.3.1 Employment and training

Installation and maintenance of the considered treatment practices can provide employment opportunities in the local community. This is assuming a low-income community such as the Magdalena Catchment Area (MRC) in Mexico City considered in this work, as these are low-paid jobs. However, they may lead to future skilled employment following the training and skills gained during the construction and maintenance process. The number of workers required might vary depending on the local regulation for the construction activities, the size of the practice and whether or not machinery is used for material handling. If the treatment practice is not built in a low-income area, the employment opportunities are still provided, except that the labour will probably not be sourced from the community where the treatment practice is being installed.

8.3.2 Human toxicity potential

Human toxicity potential (HTP), which is calculated within the LCA, is considered as a social impact in this study. This is because it represents the potential to damage human health due to the emissions produced in the life span of the treatment options. The LCA results discussed in Chapters 5-7 show that the emissions of heavy metals to air in the life cycle of the treatment options are the main cause of the HTP. The porous pavement has highest HTP (4.56 kg DCB eq. /fu) and the bio-retention unit the lowest (0.67 kg DCB eq/fu). This impact would be much lower (0.27 kg DCB eq/fu for porous pavement and 0.07 kg DCB eq/fu for the bio-retention unit) if the treatment practices were not installed. Therefore, arguably, from the social point of view with respect to the human toxicity potential, diffuse water pollution should not be treated in the urban environment. However, there are trade-offs which must be considered carefully: leaving the runoff untreated affects freshwater eco-toxicity in the urban environment and therefore the ecosystem services provided by freshwater, while HTP represents a cumulative impact occurring elsewhere along the life cycle of the treatment practices. This is discussed further below in the section on preservation of ecosystem services.

8.3.3 Aesthetics

Treatment practices that incorporate water and green areas as part of their design, such a ponds, grass swales, bio-retention units or vegetated filter strips, may improve the aesthetics of the area where they are installed. For example, according to Bastien et al. (2012) people in the UK consider it as an advantage to live near an area with a pond, as it provides a home for wildlife. In contrast, ponds can also be perceived as a potential hazard due to the risk they pose for children falling into the water (Bastien et al. 2012).

In the case of the treatment options considered in this study, child risk is not an issue since there are no deep-water pools in their design. However, from the three options analysed, only the bio-retention unit might enhance the aesthetics of the area. This is because this treatment practice can act as a garden due to the variety of plants that can grow on it and so increase the amenity of the surrounding area (MDE, 2000).

8.3.4 Flood control

Although flood control is beyond the scope of this study, some treatment practices can be designed to meet this requirement. In the specific case of the three practices analysed in this study, porous pavement is the only one that can be designed for flood control (EPA, 2009). However, this requires a different design, as the volume of the runoff that can be controlled varies in relation to the local parameters such as soil conditions, weather (Brattebo & Booth , 2003) and the kind of permeable pavement and the corresponding pavement fill (Collins et al., 2006).

8.3.5 Water supply

Due to their design characteristics, the treated runoff can be stored and used for water supply from only two of the treatment practices under analysis: bio-retention unit and porous pavement. In this way, they turn the polluted runoff which would have been waste into a valuable source for the local communities. The treated runoff can also be utilised through infiltration into the aquifer. Both the infiltration trench and porous pavement can be used for these purposes. However, before that the treated runoff should be analysed to ensure that it meets the water quality and pollutant levels defined by local regulations (EPA, 1999).

Additionally, the reduction in the runoff volume that would have gone to the drainage system may help to reduce the pollution coming from the waste water treatment plants and related environmental impacts (Critten, 2009).

8.3.6 Heat island effect

Pavements in urban areas cover most of the land in these urban areas, thereby creating a space for heat accumulation. Due to this, temperatures are higher in urban areas than in adjoining rural areas, which is an effect known as urban heat island (UHI) effect (Invisible structure, 2011). In the U.S., conventional pavement in some cities can reach peak temperatures of 48-67 ^{o}C (EPA, 2005). This heat accumulation leads to an increase in the local temperatures as well as thermal water pollution, as storm water carries heat as it passes over the paved areas (EPA, 2005). Different strategies are under consideration to control the heat island effect and its consequences. These include an increase in both trees and green areas within the urban space, green roofs and installation of so called 'cool pavements' (Invisible structure, 2011).

In the case of the treatment options considered in this work, two treatment practices can be considered to attenuate the UHI: bio-retention and porous pavement. This is because the bio-retention unit can help to increase the green space and porous pavement is considered a 'cool pavement' option (EPA, 2005).

8.3.7 Mosquito breeding

Although the runoff should flow and drain constantly if the treatment practice is well design and maintained, for various reasons it may start accumulating in the treatment practice. This may create favourable conditions for mosquito breeding, thereby leading to widespread West Nile Virus (Lampe et al., 2005). Although there are no recorded cases of mosquito breeding for any of the three treatment practices analysed in this study, the bio-retention unit is more prone to serving as a mosquito habitat, due to the combination of the wet environment and vegetation.

8.3.8 Preservation of ecosystem services

As shown in Chapters 5-7, the runoff treatment leads to the generation of environmental impacts such as global warming, acidification, marine and freshwater toxicity as well as social impacts such as human toxicity. Figure 49 illustrates this for the latter three impacts, comparing treated and untreated runoff. Thus, based on these results, it could be argued that the runoff should be left untreated. However, this could potentially jeopardise the ecosystem services provided by the Magdalena river catchment area considered in this study as discussed below.

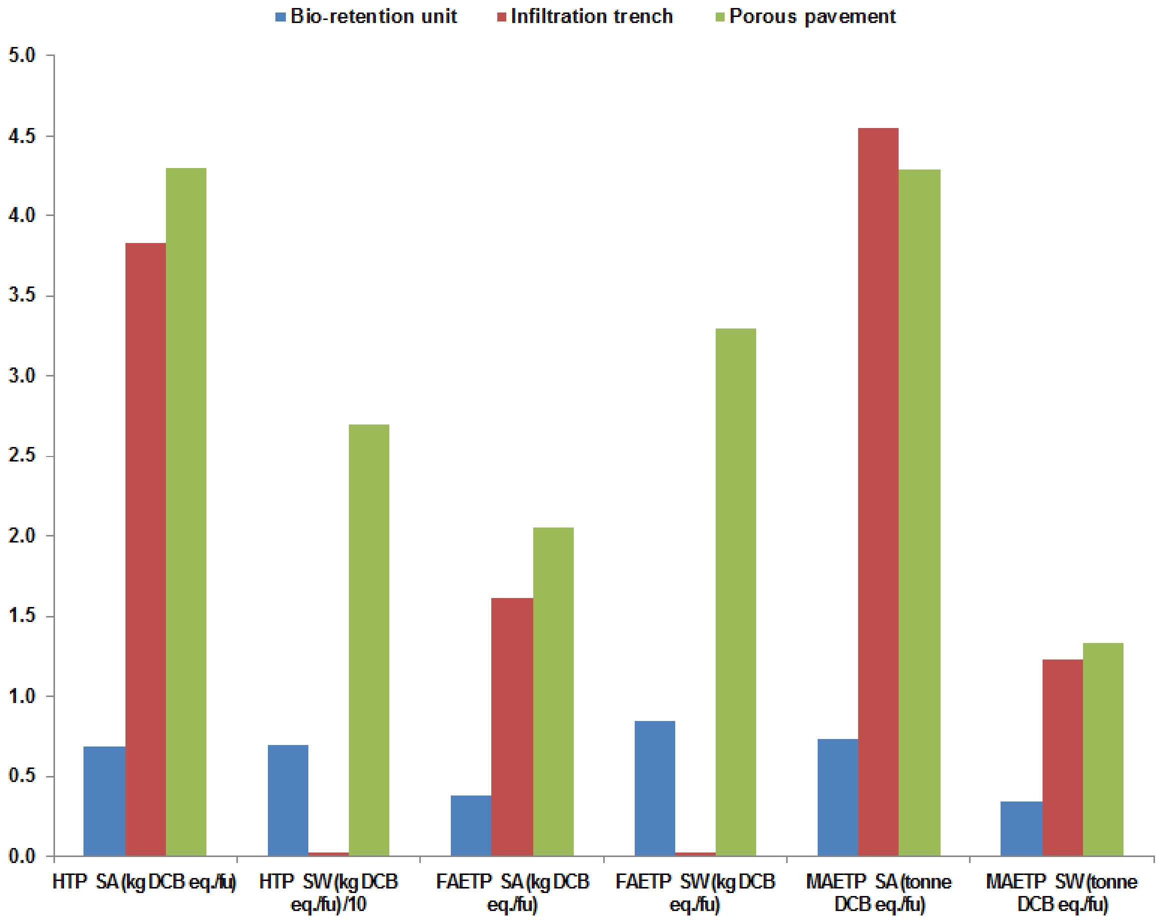

Figure 49 Human and eco-toxicity with and without treatment practices
[SA: Scenario A; SW: Scenario without treatment]

The ecosystem services provided by the MRC include support, provision, regulation and cultural services (see Table 71). For example, it has been estimated that the MRC provides 20 million m^3 of water per year, of which a third is treated to be consumed by the MRC community and the rest is used by the merchants who work by the riverside (Jujnovsky et al., 2010). The MRC forest also captures 102 t C/ha and it maintains the biodiversity, with 487 flora, 113 algae, 132 fungus and 158 vertebrates species (UNAM, 2008, cited in GDF, 2008). Another important service provided by the MRC is the provision of an area for rainfall infiltration, which helps to recharge the aquifer. Although there is no quantitative information for the remainder of the ecosystem services, these examples illustrate that the MRC is an important resource for an urban area such as Mexico City, which could be diminished or even lost with increasing diffuse water pollution. Thus the trade-offs between the human and eco-toxicity potentials with ecosystem services provided by the MRC should be considered carefully before any decisions on treatment or otherwise of diffuse water pollution are made.

Table 71 Ecosystem services provided by the MRC (UNAM (2008), cited in GDF (2008))

Support services	Provision services	Regulation services	Cultural services
Soil formation and retention	Food	Pest and disease control	Amenity
Nutrient cycling	Fresh water	Flood and sludge control	Recreation and ecotourism
	Wood, fiber and non-wood products	Water quality	Cultural heritage
	Medicinal plants	Pollination	
	Genetic resources	Air purification	
		Seed dispersal	

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The following sections conclude this sustainability assessment of the three treatment practices by ranking them on different sustainability aspects considered in this work.

8.4 Sustainability ranking of the treatment options

Identifying the “most sustainable” treatment option is a difficult task, especially without information on stakeholders’ preferences. In this work, as explained in Chapter 3, it has been assumed that all the sustainability aspects are of equal importance and a simple ranking approach has been used to identify the best option using a scale from 1 to 3, with 1 being the best and 3 the worst option; for the methodology see Chapter 3. Whilst the scores for the environmental and economic sustainability are based on the quantitative information from the base case scenario obtained through the LCA and LCC studies, the scores for the social impacts have been derived based on the qualitative discussion in Section 8.3 (with the exception of HTP which is quantitative as it is calculated as part of LCA).

Therefore, as shown in Table 72 for the base case results, the most sustainable option for treating diffuse water pollution is porous pavement, with the overall score of 4, followed closely by the bio-retention unit with the score of 5. The infiltration trench scores 9 is the least sustainable option of the three considered. The bio-retention unit is the best with respect to the environmental impacts, scoring 10 points. The other two options come close with 24 for the porous pavement and 26 for the infiltration trench. In contrast, porous pavement is the best option for the life cycle costs, followed by the bio-retention unit, while the infiltration trench represents the worst option. Finally, the porous pavement is the best option for the social aspects, scoring in total 10 points, followed closely by the bio-retention unit with a score of 13, while the infiltration trench scores 18 points.

The above results correspond to the base case for each treatment practice. However, the result of the sensitivity analysis could change the ranking. Therefore, as an illustration, the results for different compost technologies for the bio-retention unit are considered below as an example (note that owing to the many sensitivity analysis carried out in this work, showing all the variations here is deemed impractical). As seen in Table 73, the ranking of the options remains unchanged compared to the base case results. Therefore, the overall results are not sensitive to composting practices.

Table 72 Ranking of treatment options (base case results)

Treatment option / **Impact category**	**Bio-retention unit**		**Infiltration trench**		**Porous pavement**	
	Value	Ranking	Value	Ranking	Value	Ranking
Environmental sustainability						
ADP elements (mg Sb eq./fu)	2.86	1	13	2	20	3
ADP fossil (MJ/fu)	27	1	340	3	150	2
AP (g SO_2 eq./fu)	23	1	100	3	67	2
EP (g phosphate eq./fu)	4.9	1	19	3	11	2
FAETP (kg DCB eq./fu)	0.23	1	11.9	2	12.9	3
GWP (kg CO_2 eq./fu)	3.58	1	18	3	17	2
MAETP (tonne DCB eq./fu)	0.65	1	4.3	2	5	3
ODP (mg R-11 eq./fu)	0.22	1	2.26	3	1.3	2
POCP (kg ethane eq./fu)	0.21	1	14,000	3	6230	2
TETP (kg DCB eq./fu)	0.015	1	0.11	2	0.14	3
Total score for environmental sustainability		*10*		*26*		*24*
Total ranking for environmental sustainability		***1***		***3***		***2***
Economic sustainability						
Life cycle costs (US$/fu)	17	2	65	3	8	*1*
Total score and ranking for economic sustainability		***2***		***3***		***1***
Social sustainability						
Employment and training	-	2	-	3	-	1
Human toxicity potential	-	1	-	2	-	3
Aesthetic	-	1	-	3	-	2
Flood control	-	2	-	2	-	1
Water supply	-	2	-	3	-	1
Heat island effect	-	2	-	3	-	1
Mosquitoes	-	3	-	2	-	1
Total score for social sustainability		*13*		*18*		*10*
Total ranking for social sustainability		*2*		*3*		*1*
OVERALL SCORE		***5***		***9***		***4***

*NA: Not applicable to this treatment practice.

Table 73 Ranking options considering the results of the compost sensitivity analysis

Treatment option / **Impact category**	**Bio-retention unit**		**Infiltration trench**		**Porous pavement**	
	Value	Ranking	Value	Ranking	Value	Ranking
Environmental sustainability						
ADP elements (mg Sb eq./fu)	8.26	1	13	2	20	3
ADP fossil (MJ/fu)	85	1	340	3	150	2
AP (g SO_2 eq./fu)	41	1	100	3	67	2
EP (g phosphate eq./fu)	80	3	19	2	11	1
FAETP (kg DCB eq./fu)	0.075	1	1.19	2	1.29	3
GWP (kg CO_2 eq./fu)	8.9	1	18	3	17	2
MAETP (tonne DCB eq./fu)	4.9	2	4.3	1	5	3
ODP (mg R-11 eq./fu)	0.74	1	2.26	3	1.3	2
POCP (kg ethane eq./fu)	0.009	1	14000	3	6230	2
TETP (kg DCB eq./fu)	0.19	2	0.11	1	0.14	3
Total score for environmental sustainability		*14*		*23*		*23*
Total ranking for environmental sustainability		***1***		***2***		***2***
Economic sustainability						
Life cycle costs (US$/fu)	17	2	65	3	8	*1*
Total score and ranking for economic sustainability		***2***		***3***		***1***
Social sustainability						
Employment and training	-	2	-	3	-	1
Human toxicity potential	-	1	-	2	-	3
Aesthetic	-	1	-	3	-	2
Flood control	-	2	-	2	-	1
Water supply	-	2	-	3	-	1
Heat island effect	-	2	-	3	-	1
Mosquitoes	-	3	-	2	-	1
Total score for social sustainability		*13*		*18*		*10*
Total ranking for social sustainability		*2*		*3*		*1*
OVERALL SCORE		***5***		***8***		***4***

*NA: Not applicable to this treatment practice.

8.5 Summary

As shown in this analysis, assuming that all the sustainability impacts are of equal importance, porous pavements, followed closely by the bio-retention unit, are the most sustainable option for diffuse water pollution for the Magdalena river catchment area. The porous pavement is best for the economic and social sustainability while the bio-retention unit is best for the environmental impacts. The infiltration trench is the least sustainable on all dimensions of sustainability, ranking overall third.

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9 Conclusions

The aim of this research was to evaluate the sustainability of structural practices for controlling diffuse water pollution in urban areas, considering a life cycle approach. The environmental evaluation has been conducted using LCA as a tool, while the economic evaluation has been conducted using environmental life cycle costing as a tool. Due to the lack of information the social evaluation has been conducted on a qualitative basis, analysing social impacts in different literature sources. Finally, the results are ranked, in order to identify the most sustainable option. Three case studies have been analysed: bio-retention unit, infiltration trench and porous pavement. It was assumed that all three options are situated in the Magdalena river catchment in Mexico City. This city has been selected as an example of the conditions prevailing in urban areas of developing countries, where study of diffuse water pollution is still in the early stages. This chapter summarises the conclusions of the sustainability evaluation of each treatment practice and gives recommendations for future work.

9.1 Bio-retention unit

9.1.1 Environmental sustainability

- Bio-retention units can help reduce freshwater aquatic toxicity; but at the same time they increase marine aquatic and human toxicity compared to leaving the runoff untreated for diffuse pollution. In addition, a number of other environmental impacts are generated which would not otherwise have been produced. Therefore, trade-offs need to be made between improving the quality of the local urban environment and the other life cycle impacts generated elsewhere. These are difficult decisions that can only be made by the appropriate stakeholders.

- The study also points to the life cycle stage which could be targeted for reducing the environmental impacts from this treatment option. This is the construction stage, which is the cause of most impacts, contributing from 43% to ozone layer depletion to 83% to acidification. This is mainly due to the emissions to air generated in this stage. Maintenance is the next largest contributor, while the contribution from the decommissioning and the operation stages is negligible.

- The results of sensitivity analysis show that pollutant concentration on the inlet of the unit affects the freshwater, marine and human toxicities, since the main pollutants considered are heavy metals, so that the higher the pollution levels the higher the freshwater toxicity potential. For the same inlet concentrations of heavy metals, treatment of the runoff by the bio-retention unit can reduce the freshwater toxicity by about a half. The opposite is the case for marine eco-toxicity, which increases by 50% compared to leaving the run off untreated. Therefore, the concentration of heavy metals in the runoff is an important parameter which should be considered carefully in decisions related to bio-retention units.

- The variation in the rainfall depth also affects the environmental impacts associated with the bio-retention unit. An increase in the treatment volume leads to a decrease in the environmental impacts, ranging from 12% for acidification to 62% for terrestrial eco-toxicity.

9.1.2 Economic sustainability

- The life cycle costs of the bio-retention unit are calculated at \$17 per m^3 of runoff treated or \$ 49,000 over the life time of the bio-retention unit.

- The costs are dominated by the corrective maintenance stage (76%). This is due to the costs of machinery for sludge removal (63%). Increasing the treatment volume by five times reduces the LCC by 20%.

9.2 Infiltration trench

9.2.1 Environmental sustainability

- The results show that corrective maintenance on average contributes 85% of all the environmental impacts. Therefore, this stage is identified as the target to reduce these impacts.

- Treating runoff leads to an increase in both marine aquatic and freshwater aquatic pollution, in comparison with leaving runoff untreated. These results show that an analysis of the trade-off should be carried out by the appropriate stakeholders, in order to decide on the implementation of an infiltration trench.

- The results from the sensitivity analysis related to the pollutant concentration point out that marine aquatic and freshwater aquatic environments are affected. These results show that the higher the pollutant concentration, the higher the value for these impacts. Hence, the pollutant concentration is identified as a parameter that should be carefully analysed.

- As to the variation in the stone removal frequency, the results illustrate that this parameter leads to a decrease of 70% on average for all the environmental impacts. This reduction is due to an increase in the sludge, which is increased as the stone removal frequency is increased. Therefore, based on these results this parameter should be considered with caution in the implementation of an infiltration trench. Finally it is found that the variation in the rainfall depth leads to an average reduction of 70% for all the environmental impacts.

9.2.2 Economic evaluation

- The life cycle cost of the infiltration trench amounts to US$ 65/fu or US$191,000 over the life span of this treatment practice.

- Corrective maintenance is the main cause of the costs, amounting to 89%. This is mainly due to the cost of machinery operation (70%). The increase in the rainfall depth leads to a modest decrease of 0.45% of the LCC.

9.3 Porous pavement

9.3.1 Environmental sustainability

- The construction stage is found to be the main factor responsible for all the environmental impacts, the contributions from this stage range from 76% for freshwater aquatic to 97% for terrestrial ecotoxicity. Therefore, the construction stage is identified as the stage that should be targeted in order to reduce the environmental impacts of the porous pavement.

- Porous pavement helps in the reduction of freshwater aquatic pollution, while also contributing to an increase in marine aquatic pollution, creating at the same time other environmental impacts that would have not been produced. Therefore, the installation of porous pavement should be based on the analysis of the trade-off, which can only be evaluated by the appropriate stakeholders.

- The results of the sensitivity analysis show that increasing the rainfall depth leads to a decrease that ranges from 11% for terrestrial toxicity to 67% for freshwater aquatic potential impacts. As to the variation in the pollutant concentration, the results indicate that freshwater aquatic and marine aquatic potential levels are affected; additionally this variation also shows that as pollutant concentration is increased, these impacts are increased. Therefore, both rainfall and pollutant concentration should be considered with caution when considering porous pavement as a treatment option.

9.3.2 Economic evaluation

- Total cost of porous pavement amounts to US$ 14,800 over the life cycle of this treatment practice, which equals US$ 8/fu.

- Most of the contribution to total cost is due to the construction stage (74%), which is mainly caused by the cost of materials (78%). It is found that an increase in the rainfall depth leads to a reduction of 63% of the LCC.

9.4 Sustainability comparison of structural practices

- Based on the assumptions made in this work, the bio-retention unit appears to be environmentally the most sustainable option for diffuse water pollution.
- Porous pavements are the best option for economic and social aspects.
- Overall, the porous pavements are the best option, followed closely by the bio-retention unit.

9.5 Recommendations

- Based on the results of this work, application of porous pavements is recommended in the Mexico City region for controlling the diffuse water pollution. Bio-retention units also show good sustainability performance.
- The trade-offs between the different sustainability aspects are important and should be considered carefully before any decisions are made on diffuse water pollution treatment. This also includes the trade-offs with the additional life cycle impacts generated by the treatment options, compared to the impacts from the untreated runoff.
- Non-structural practices should also be considered in addition to structural practices. Thus it is recommended that appropriate regulations and educational programs be developed for a more efficient treatment of diffuse water pollution.
- By 2030 the National Water is expecting to have diffuse pollutant sources under control, therefore in order to reach this goal more research about controlling this kind of pollution is recommended.

9.6 Future work

The following future work is recommended:

- Consideration of different designs of structural treatment practices to explore how this might affect the environmental impacts and the economic costs.
- Analysis of other pollutants in the runoff in addition to heavy metals considered here.
- Consideration of the application of the different structural treatment practices in other countries and regions.
- Consideration of different non-structural practices and the best combination with structural practices.

Appendix A

This section is aim at describing the environmental impact categories considered in the CML impact characterisation method (Guinee,et al. 2001).

Impact category	**Definition**
Abiotic depletion potential elements	This impact category measures the depletion of natural resources, including energy resources. It is based on ultimate reserves and is expressed relative to Sb for ADP elements and MJ for ADP fossil, respectively.
Abiotic depletion potential fossil	
Acidification potential	This impact category measures the acidifying potential from acidifying pollutants. It is expressed relative to SO_2.
Eutrophication potential	This impact category measures the potential impact from nutrients (such as N and P). It is expressed relative to PO^{-3}_4
Global warming potential	This impact category measures the potential from greenhouse pollutants. It is expressed relative to CO_2.
Ecotoxicity: Freshwater aquatic Marine aquatic Terrestrial	This impact category measures the impact of toxic substances on aquatic and terrestrial ecosystems. It is expressed relative to DCB.
Human toxicity potential	This impact category measures the potential of toxic substances on human health. It is expressed relative to DCB.
Photochemical ozone creation potential	This impact category measures the potential of formatting reactive chemical compounds due to the action of sunlight on air pollutants. It is expressed relative to ethylene.
Ozone depletion potential	This impact category measures the impact of different substances that contribute to the reduction of ozone depletion. It is expressed relative to trichlorofluoromethane or R-11.

Appendix B

The estimation of the pollutant accumulated per kg of sediment removed from the bio-retention unit is shown in this section. The estimation of the pollutant per year is described in Table A, while the estimation of pollutant accumulated per kg of sediment is described in Table B.

Table A Pollutant balance per year

Parameter	Pollutant load in	Pollutant load out	Pollutant load accumulated
TSS	262.13	107.47	154.65
Cd	6.14E-04	6.14E-5	5.53E-04
Cr	7.54E-04	7.54E-5	6.79E-04
Cu	2.94E-03	2.06E-04	2.73E-03
Ni	9.65E-04	6.75E-05	8.79E-04
Pb	4.12E-03	2.89E-04	3.83E-03
Zn	1.64E-02	1.15E-03	1.53E-02

Table B Pollutant accumulation over 4 years per kg of sediment

Parameter	Pollutant accumulation (kg/year)	Pollutant accumulated over 4 years	mg pollutant/kg sediment
TSS	154.65	618.62	
Cd	5.53E-04	2.21E-03	3.57
Cr	6.79E-04	2.72E-03	4.39
Cu	2.73E-03	1.09E-02	17.67
Ni	8.79E-04	3.59E-03	5.80
Pb	3.83E-03	1.53E-02	24.79
Zn	1.53E-02	6.10E-02	98.65