Universitat Autònoma de Barcelona

Towards sustainable cities through an environmental, economic and eco-efficiency analysis of urban sanitation and drainage systems

Anna Petit Boix

Towards sustainable cities through an environmental, economic and eco-efficiency analysis of urban sanitation and drainage systems

Anna Petit Boix

Doctoral thesis

Directors: Dr. Joan Rieradevall Pons (UAB)
Dr. Xavier Gabarrell Durany (UAB)
Dr. Alejandro Josa Garcia-Tornel (UPC)

Academic tutor: Dr. Joan Rieradevall Pons (UAB)

A thesis submitted in fulfilment of the requirements for the Doctoral degree in Environmental Science and Technology

Sostenipra research group
Institut de Ciència i Tecnologia Ambientals (ICTA)
María de Maeztu program for Units of Excellence in R&D (MDM-2015-0552)
Universitat Autònoma de Barcelona (UAB)

Bellaterra, May 2017

Towards sustainable cities through an environmental, economic and eco-efficiency analysis of urban sanitation and drainage systems

By

Anna Petit Boix

A thesis submitted in fulfilment of the requirements of the

PhD degree in Environmental Science and Technology

May 2017

The present doctoral thesis was developed thanks to the pre-doctoral fellowships awarded to Anna Petit Boix by Generalitat de Catalunya (FI-DGR 2014; March to September, 2014) and the Spanish Ministry of Education, Culture and Sports (FPU13/01273; September, 2014 – until end of training). It was conducted in a "María de Maeztu" Unit of Excellence in R&D (MDM-2015-0552) thanks to the support of the Spanish Ministry of Economy and Competitiveness

Today
I’m laughing the clouds away
I hear what the flowers say
and drink every drop of rain
And I see
places that I have been
in ways that I've never seen
my side of the grass is green
Oh I can't believe that it's so simple
It feels so natural to me

[*Love is easy – Mcfly*]

Avi, ho he tornat a aconseguir

The present thesis entitled *Towards sustainable cities through an environmental, economic and eco-efficiency analysis of urban sanitation and drainage systems*, by Anna Petit Boix, was carried out at the Institute of Environmental Science and Technology (ICTA), at Universitat Autònoma de Barcelona (UAB)

Anna Petit Boix

under the supervision of Dr. Joan Rieradevall and Dr. Xavier Gabarrell, from ICTA and the Department of Chemical, Biological and Environmental Engineering (UAB), and Dr. Alejandro Josa, from the Department of Civil and Environmental Engineering and the Institute of Sustainability at the Technical University of Catalonia (UPC).

Joan Rieradevall Pons　　Xavier Gabarrell Durany　　Alejandro Josa García-Tornel

Bellaterra (Cerdanyola del Vallès), May 5, 2017

Contents

Tables

Figures

Abbreviations

1,4 DB eq	1,4 dichlorobenzene equivalent emissions
A	Acidification
AD	Abiotic depletion
ALCA	Attributional life cycle assessment
ALO	Agricultural land occupation
A/V	Biofilm area-to-liquid volume
BOD	Biological Oxygen Demand
BMP	Best Management Practices
CED	Cumulative energy demand
C_2H_4 eq	Ethylene equivalent emissions
CH_4	Methane
CFC -11 eq	Trichlorofluoromethane equivalent emissions
CLCA	Consequential life cycle assessment
CML	Institute of Environmental Sciences (Leiden)
CO_2 eq	Carbon dioxide equivalent emissions
COD	Chemical Oxygen Demand
DO	Dissolved oxygen
DU	Declared unit
E	Eutrophication
ENEI	Eutrophication Net Environmental Impact
FD	Fossil depletion
FE	Freshwater eutrophication
FET	Freshwater ecotoxicity
FST	Filter, swale, infiltration trench
FU	Functional unit
GHG	Greenhouse gas
GW	Global warming
H_2S	Hydrogen sulfide
HDPE	High-density polyethylene
HRT	Hydraulic retention time
HT	Human toxicity
ICTA	Institute of Environmental Science and Technology (UAB)
IE	Industrial ecology
ILCD	International Reference Life Cycle Data System
IPCC	Intergovernmental Panel on Climate Change
IR	Ionizing radiation
ISO	International Organization for Standardization
LCA	Life cycle assessment
LCC	Life cycle costing
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
LCSA	Life cycle sustainability assessment
LDPE	Low-density polyethylene
ME	Marine eutrophication
MET	Marine ecotoxicity
MJ	Mega joules
MD	Metal depletion

NEI	Net Environmental Impact
NLT	Natural land transformation
N_2O	Nitrous oxide
NO_3^-	Nitrate ion
NO_2^-	Nitrite ion
OD	Ozone depletion
PMF	Particulate matter formation
POC	Photochemical ozone creation
POF	Photochemical ozone formation
PO_4^{3-}	Phosphate
PP	Payback period
PVC	Polyvinylchloride
PWTP	Potable water treatment plant
Sb eq	Antimony equivalent emissions
SBR	Sulfate reducing bacteria
SETAC	Society of Environmental Toxicology and Chemisty
S-LCA	Social life cycle assessment
SO_2 eq	Sulphur dioxide equivalent emissions
Sostenipra	Sustainability and Environmental Prevention research group
TA	Terrestrial acidification
TC	Total cost
TET	Terrestrial ecotoxicity
TN	Total nitrogen
TP	Total phosphorus
TSS	Total suspended solids
UAB	Universitat Autònoma de Barcelona
ULO	Urban land occupation
UNEP	United Nations Environment Program
VFA	Volatile fatty acids
WD	Water depletion
WWTP	Wastewater treatment plant

Acknowledgements

I would like to thank a series of people who helped me during my training period. I am positive that this thesis would have not been possible without them.

Als meus de sempre: M'és impossible començar d'una altra manera que no sigui agraint infinitament l'ajuda i suport de la meva família. Sense els meus pares no hagués arribat fins aquí, i sense la meva mare no hagués marxat a l'altra punta del món quan era impensable creuar el carrer. Gràcies per la paciència i per no deixar que abandonés ni quan era difícil trobar el camí, ni quan semblava que no podia més.

Als meus directors: Vull agrair de tot cor l'oportunitat que m'han donat els Dr Joan Rieradevall, Dr Xavier Gabarrell i Dr Alejandro Josa. Gràcies per haver confiat en mi per fer aquesta feina i creure que era capaç de fer-la. He après infinitat de coses durant aquests anys. Gràcies per haver-me ajudat a saber què és la recerca.

Als de cada dia: Ha estat fantàstic començar i acabar aquest camí al costat del David i el Pere. Gràcies per veure-hi quan ningú hi veia i per ser amics de veritat! Ana, gracias por los buenos momentos y estar allí. Gràcies a tots els que heu compartit una part del vostre temps amb mi a l'ICTA (Mireia, Martí, Susana, Aniol, Perla, Ana María, Aurélie, Eva, Joan Manuel, Sara, Julia, Farah, Charlie, Ana Arahuetes, Esme, Cinzia...). I a les que estan lluny, però a prop cada dia – Esther, Violeta, Katherine i Elena.

To the ones on the other side: I would like to thank Dr Defne Apul for her warm welcome while I was in the US. I really appreciate the way you made me feel like home and thank you for listening. Thanks to Robert, Jay and Youcan. We had a great time and literally worked like nobody was watching. Also big thank you to Sonia, Ilke, Jazzy and Elizabeth – because I also learned from you.

Als qui fan el que no sé fer: Perquè no tot el que he fet, ho he fet sola, vull agrair el suport de l'equip del projecte Aquaenvec. Gràcies als inèdits i a enginyeria química pel cop de mà que em doneu de tant en tant. Albert, Pau, Núria: sento que m'ho hagueu d'explicar tot molts cops. Gràcies per tenir paciència amb les meves preguntes poc intel·ligents sobre estructures! También quiero dar las gracias al equipo de la UFSCar por la ayuda con la investigación sobre inundaciones.

But because nobody is perfect, I would also like to apologize:

Als que m'han sentit queixar-me: Perdó si mai he estat pesada!

Als meus estudiants: Si mai he demanat massa, ho sento! La meva intenció era ajudar!

To you, reader: I am deeply sorry if this dissertation is too long or too tough. All of my energy and good will were invested in this research. I hope it is interesting and useful enough to make the world a better place.

Summary

The growth of cities worldwide is associated with an increasing demand for sanitation and drainage infrastructure in the context of the water cycle. Combined with the effects of climate change, which alter the rainfall patterns, the situation of these systems in urban environments is critical. Part of the existing sewer networks require an imminent renovation, others must be constructed in developing areas, whereas stormwater runoff becomes a threat in terms of flooding because of the soil imperviousness. In this context, we must determine the best practices aimed at reducing these issues from an innovative environmental and economic viewpoint and at the same time adapt cities to climate change.

However, there is a lack of environmental and economic data that define the main impacts of sewer management and different types of flood prevention systems. To this end, the combination of different tools is key to obtaining integrated information about these infrastructures. Furthermore, it is essential to determine the implications of different sanitation and drainage management scenarios at the urban, suburban, neighborhood and building scales.

In response to this demand, this dissertation revolves around the following questions, which are studied based on the structure illustrated in **Figure X1**:

(1) What are the environmental and economic hotspots of sewer networks and the parameters that affect this outcome in medium-sized cities?

(2) Is decentralized sanitation a better environmental option in small suburban areas when dealing with peak sanitation demand?

(3) Do flood prevention strategies result in a positive environmental and economic performance when avoided damage is accounted for? If so, are green BMPs better than gray systems?

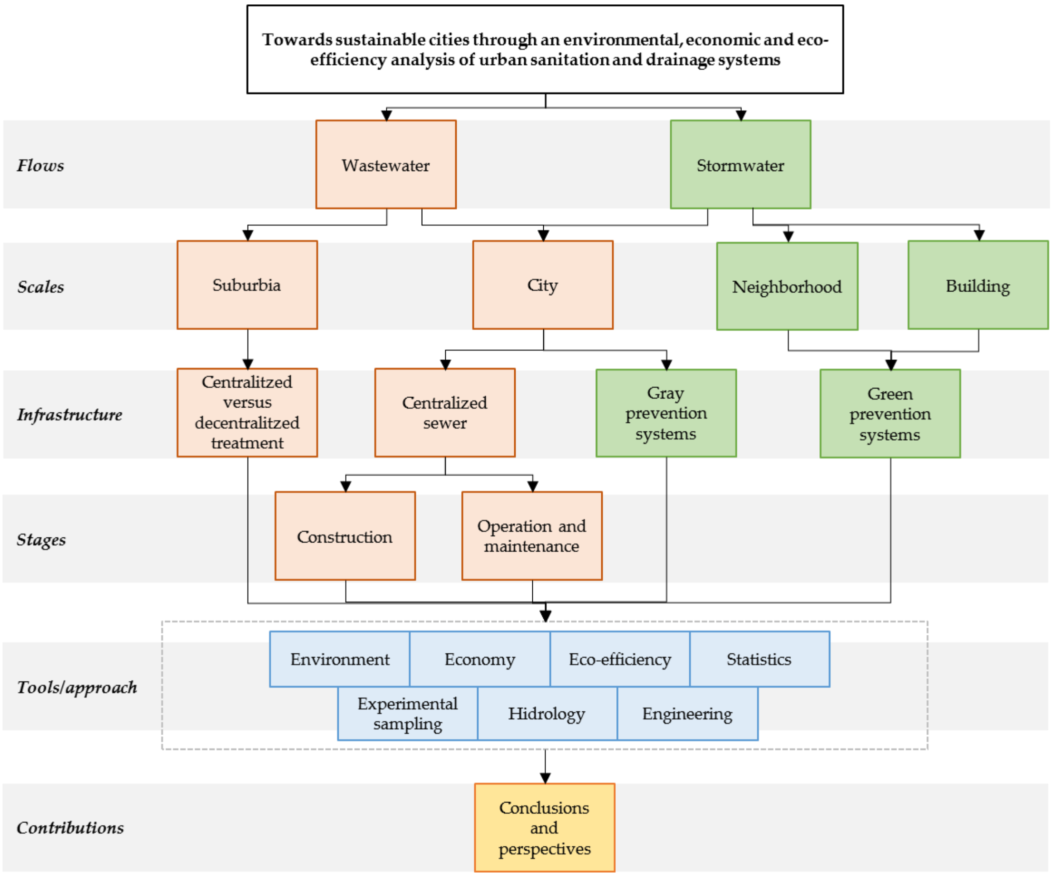


Figure X1 Elements studied in the dissertation, tools and contributions

The following sections summarize the main methods and results obtained in this doctoral thesis in the context of environmental science and technology.

Materials and methods. Integration of tools

In this dissertation, the industrial ecology framework is used by applying specific methods such as life cycle assessment (LCA) (both attributional and consequential), life cycle costing (LCC) and eco-efficiency to urban sanitation and drainage systems. This interdisciplinary research requires the incorporation of additional methods, such as statistical studies or field experimental analyses to obtain real data on the greenhouse gas (GHG) emissions resulting from sewers. This integrated framework with a life cycle perspective includes applications coming from engineering and hydrology. The scales of analysis range from cities and suburban areas to neighborhoods and buildings depending on the study.

The construction stage of sewer networks

The life cycle of sewers was widely analyzed in medium-sized cities in this dissertation. For instance, we estimated that one meter of constructive solution with a diameter of 300 mm might generate up to 120 kg CO_2 eq when pipes are made of concrete, or up to 190 kg CO_2 eq in the case of high-density polyethylene pipes. In general, concrete pipes seem to be the most environmentally friendly option in comparison with plastic pipes due to their longer lifespan, although it depends on the case. However, pipe materials are not the only factors that determine the environmental impacts of a sewer constructive solution. In some cases, the trench might contribute to 80% of the environmental impacts of the construction phase, which is a relevant issue to consider in decision-making. Through a structural parametric study we found the equivalent constructive solutions that generate the lowest environmental impact. Reducing the use of concrete or reusing the excavated soil for backfilling and protecting the pipes might represent a reduction in the environmental impacts.

The operation and maintenance stage of sewer networks associated with each city

The operation stage is challenging in the context of urban planning. When studying a sample of Spanish cities – in general -, and two contrasting cities – in particular -, we observed that urban planning is essential in the consumption of energy. The coastal city of Calafell (Mediterranean climate) was compared with the city of Betanzos (Atlantic climate). In this case, the location of Calafell's wastewater treatment plant at a higher elevation than the city resulted in Calafell consuming more pumping energy (0.47 kWh/m^3) than Betanzos (0.11 kWh/m^3), where wastewater flows gravitationally. In an additional research line, GHG emissions were found in the sewer through sampling campaigns in these two cities with different climate. These GHG include methane and nitrous oxide, which have a global warming potential that is 25 and 298 times as high as carbon dioxide, respectively. The largest emissions were mainly detected during the summer due to high temperature, and in turbulent areas of the sewer, such as pump stations.

Innovation through the eco-efficiency of sewer networks

The combination of environmental and economic values was conducted through one of the first eco-efficiency studies that apply ISO 14045:2012. In the case of Calafell and Betanzos, the eco-efficiency of both the life cycle stages and total sewer value was compared. Regardless of climate and urban form, results show that the operation stage generates the largest environmental impacts (up to 74%), whereas the installation (i.e., the trench) mostly contributes to the economic costs (70-75%). There is room for improvement in the operation stage in terms of pumping energy, whereas the installation costs depend on labor. This contribution to the life cycle costs is especially relevant in the context of the entire urban water cycle, as these might represent up to 45% of the costs in these cities. In this sense, the thesis discusses the need for managing sewers in the context of integrated wastewater management in order to improve the eco-efficiency of the system.

Centralized versus decentralized wastewater treatment in the environmental sphere

The implementation of centralized wastewater treatment systems might generate higher or lower impacts depending on the scale of application and level of demand. For this reason, in a specific study in suburban temporal settlements in the island of Minorca (Spain), we analyzed different wastewater treatment scenarios with centralized or decentralized systems to determine the most environmentally friendly options for dealing with tourist seasons. Results depend on the selected indicators and variables. However, it seems that a centralized scenario that connects the settlement to an existing treatment plant with a large treatment capacity is beneficial due to economies of scale. Based on a weighted environmental score, this scenario might generate 390 mPt/m^3. This represents a 12% reduction with respect to partial decentralization through septic tanks, or 36% reduction with respect to treating seasonal wastewater at a constructed wetland. Nevertheless, the peak population, seasonal wastewater generation, and duration of the seasonal period were varied through a probabilistic analysis to extrapolate these results to similar areas. In this case, treating wastewater in septic systems and transporting the effluent to the neighboring treatment plant can have the best average results (520 mPt/m^3). In general, results depend on the duration of the seasonal period. The centralized scenario has the highest variability (77%) because of the wastewater generation, which is associated with pumping. For this reason, seasonal and permanent settlements should be assessed in different types of analyses.

Flood analysis from a life cycle perspective

In the field of flood prevention, this thesis provides a new vision and an innovative approach, as these are the first studies that integrate the avoided impacts of damage prevention into the environmental and economic effects of investing in preventive measures. These studies are of interest in the framework of urban planning and focus on particular case studies with different climate and urban context. From a methodological perspective, flooding analyses provide some ideas in the field of LCA methods and discuss how to deal with the consequences of flooding from an integrated viewpoint and considering the sustainability of cities.

Environmental and economic balance of flood prevention systems

To determine the net balance of the environmental impacts and economic costs of certain preventive actions, two contrasting areas were studied. On the one hand, ephemeral streams pose a threat in the Mediterranean coast. In this area, we studied the historical evolution of the damage generated in three cities of the Maresme region (Catalonia, Spain) before and after the implementation of post-disaster emergency actions. These consisted of wall reconstructions, or channeling and/or adapting the streams. When comparing the impacts and costs invested in prevention with the avoided damage, it was estimated that these actions have an economic and environmental payback time of 2 and 25 years, respectively. This result highlights one of the environmental and economic consequences of investing in material-intensive (but durable) systems, which in this case use large amounts of concrete. In contrast, an existing green prevention system was also studied, this time in the city of São Carlos (Brazil), where frequent rainfall generates a large number of damaged goods every year. In this area, we observed that the eco-efficiency ratio of the system (0.38 kg CO_2 eq/€) is lower than in the actions applied in the ephemeral streams (1.2 kg CO_2 eq/€). Although these systems are not equivalent, using a green system with a carbon capture potential might be beneficial; besides, it also fosters the natural infiltration of stormwater. In this case, it was estimated that the carbon footprint of the system can be paid off once the destruction of one car is avoided.

Upcoming challenges of urban sanitation and drainage research

The results obtained in this dissertation open up future research lines. It is interesting to determine if the trends observed in the sewers of small to medium sized cities also apply in big cities, where there is a greater urban complexity. On the other hand, the centralization debate in suburban areas presents more challenges, such as studying the possibilities of wastewater treatment at different scales, climates and tourist contexts, as well as the symbiosis that natural systems can provide by storing stormwater for decentralized uses. In this field, we should also understand the applications of rainwater and/or reclaimed water in agriculture and services to close the urban water cycle in the most sustainable manner. In the case of floods, additional predictive models are needed to determine the exact consequences of flooding and its prevention. Lastly, the life cycle approach of this dissertation could be completed by analyzing the ecosystem services (i.e., benefits to humans) of different sanitation and drainage alternatives. By doing so, their benefits and impacts would be clearly defined to ease decision-making processes at the urban scale.

Resum

El creixement de les ciutats arreu del món porta associat un increment en la demanda d'infraestructures de sanejament i drenatge associades al cicle de l'aigua. Combinat amb els efectes del canvi climàtic, que estan alterant els patrons de pluviometria, la situació d'aquests sistemes en entorns urbans és crítica. Bona part de les xarxes de clavegueram existents requereixen una renovació urgent, d'altres han de ser construïdes en zones en creixement, mentre que l'escolament superficial d'aigua pluvial esdevé una amenaça quant a inundacions degut a la impermeabilització del sòl. En aquest context, cal determinar a través d'una nova visió ambiental i econòmica quines són les millors pràctiques per reduir aquestes problemàtiques i al mateix temps adaptar les ciutats al canvi climàtic.

Tot i així, existeix una manca de dades ambientals i econòmiques que descriguin els principals impactes de la gestió de les xarxes de clavegueram, així com de diferents sistemes de prevenció d'inundacions. Per aquesta finalitat, la combinació de diferents eines és clau en l'obtenció d'informació integrada sobre aquestes infraestructures. A més, també és essencial determinar les implicacions de diferents escenaris de gestió del sanejament i drenatge a escala urbana, periurbana, de barri i d'edifici.

En resposta a aquestes demandes, la tesi es desenvolupa a l'entorn de les següents preguntes, les quals s'estudien d'acord a l'estructura de la **Figura X1**:

(1) Quins són els punts crítics ambientals i econòmics de les xarxes de clavegueram i els paràmetres que afecten aquests resultats en ciutats mitjanes?

(2) És el sanejament descentralitzat una opció ambientalment millor en zones suburbanes petites quan cal gestionar una demanda pic?

(3) Tenen les estratègies de prevenció d'inundacions un comportament ambiental i econòmic positiu quan es comptabilitzen les pèrdues evitades en inundacions? En cas afirmatiu, és millor aplicar pràctiques de gestió verdes o grises?

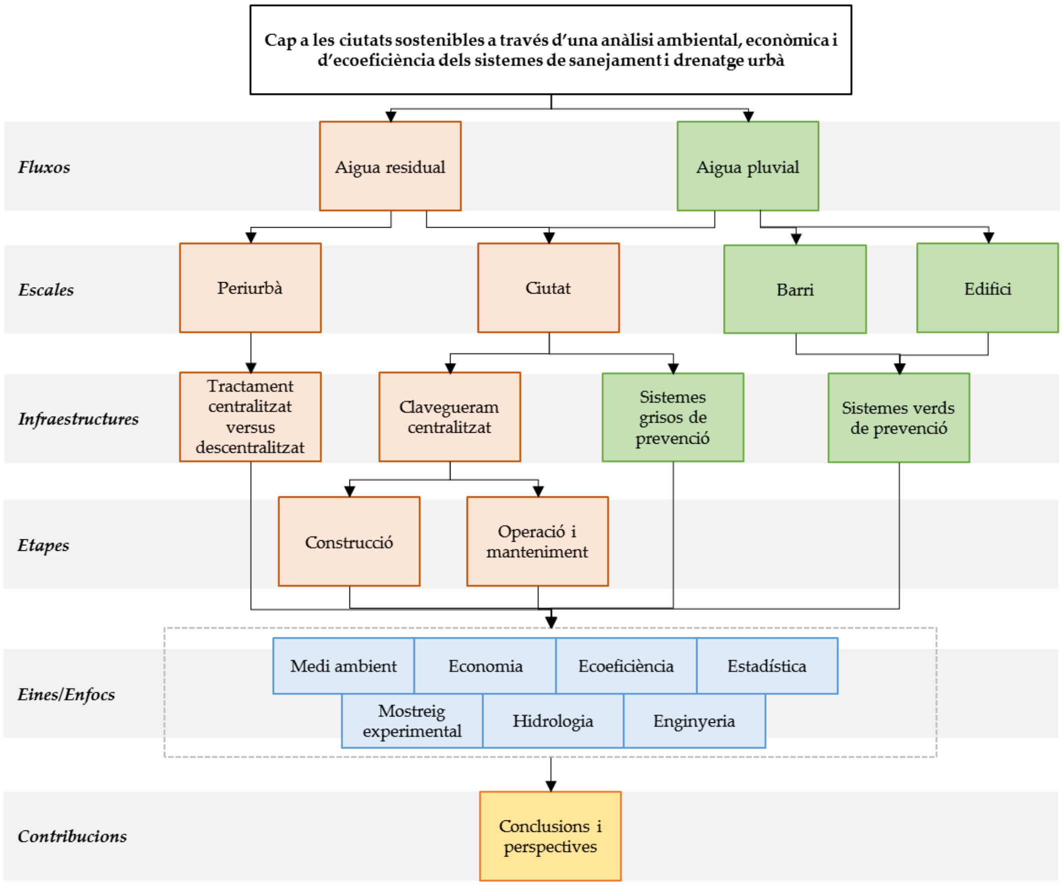


Figura X1 Elements estudiats a la tesi, eines i contribucions

Seguidament, es presenta un resum on es recullen els principals mètodes i resultats obtinguts en aquesta tesi doctoral en el marc de la ciència i tecnologia ambientals.

Materials i mètodes. Integració d'eines

En aquesta tesi doctoral es fa ús del marc de l'ecologia industrial, tot aplicant mètodes específics com l'anàlisi del cicle de vida (ACV) (atribucional i conseqüencial), l'anàlisi dels costos del cicle de vida (ACCV) i l'ecoeficiència als sistemes de sanejament i drenatge urbà. Aquesta recerca interdisciplinària requereix la incorporació de mètodes addicionals, com ara estudis estadístics o anàlisis experimentals al camp per obtenir dades reals sobre l'emissió de gasos d'efecte hivernacle (GEH) provinents del clavegueram. Dins d'aquest marc integrador amb perspectiva de cicle de vida s'inclouen aplicacions provinents de l'enginyeria i la hidrologia. Les escales d'anàlisi inclouen des de les ciutats i zones periurbanes fins als barris i edificis depenent de l'estudi.

L'etapa de construcció del clavegueram

El cicle de vida de les xarxes de clavegueram ha estat analitzat en detall en aquesta tesi en ciutats de mida mitjana. Per exemple, d'entre els resultats s'ha estimat que un metre de solució constructiva de 300 mm de diàmetre pot generar fins a 120 kg CO_2 eq si és de formigó, o fins a 190 kg CO_2 eq, si és de polietilè d'alta densitat. Quant a les canonades, generalment el formigó es presenta com la opció ambientalment millor envers els plàstics degut a la seva llarga vida útil, tot i que depèn dels casos considerats. No obstant, s'ha determinat que els materials de la canonada no són els únics determinants de l'impacte ambiental d'una solució constructiva per clavegueram. En alguns casos, la contribució de la rasa pot representar fins un 80% dels impactes ambientals de l'etapa constructiva, fet que no és irrellevant de cara a la presa de decisions. Mitjançant un estudi estructural paramètric s'han pogut trobar les solucions constructives equivalents que generen el menor impacte ambiental. Així, reduir l'ús de formigó en les rases i reutilitzar els materials del sòl excavats per reomplir i protegir les canonades pot significar una reducció dels impactes ambientals.

L'etapa d'operació i manteniment de xarxes de clavegueram relacionades a cada ciutat

L'etapa d'operació inclou importants reptes en l'àmbit del planejament urbà. Estudiant una mostra de municipis d'Espanya, en general, i dos municipis contrastats, en particular, s'observa que el planejament és clau en el consum d'energia. S'han comparat el municipi costaner de Calafell (clima mediterrani) i Betanzos (clima atlàntic). En aquest cas, la ubicació de l'estació depuradora de Calafell a una cota més elevada que el municipi fa que el consum d'energia de bombeig (0.47 kWh/m^3) sigui major que a Betanzos (0.11 kWh/m^3), on l'aigua circula per gravetat. En una línia de recerca addicional, s'han observat emissions de GEH al clavegueram a través de campanyes de mostreig en aquests dos municipis de climes diferents. Aquests GEH inclouen el metà i el monòxid de dinitrogen, els quals tenen un potencial d'escalfament global 25 i 298 cops més elevat que el diòxid de carboni, respectivament. Principalment es van detectar majors emissions durant l'estiu associades a les elevades temperatures i en zones de turbulència del clavegueram, com ara abans de les estacions de bombeig.

Innovació a través de l'ecoeficiència de xarxes

La combinació de valors ambientals i econòmics del clavegueram es du a terme a través d'un dels primers estudis d'ecoeficiència que aplica la ISO 14045:2012. En els casos de Calafell i Betanzos s'ha comparat l'ecoeficiència tant de les diferents etapes del cicle de vida com dels valors totals dels municipis. Els resultats indiquen que, independentment del clima i l'estructura urbana, l'etapa d'operació és la que genera més impactes ambientals (fins el 74% dels impactes), mentre que la instal·lació (és a dir, la rasa) contribueix més a la generació de costos econòmics (70-75%). L'operació té marge de millora quant al consum d'energia associada al bombeig, mentre que els costos de la instal·lació depenen de la mà d'obra. Aquesta contribució en els costos del cicle de vida és especialment rellevant en el context del cicle urbà de l'aigua complet, donat que poden representar fins al 45% dels costos d'aquests municipis. En aquest sentit, queda clara la necessitat de gestionar el clavegueram en context de la gestió integrada de les aigües residuals per millorar l'ecoeficiència del sistema.

Tractament de l'aigua residual centralitzat front a descentralitzat en l'esfera ambiental

La implementació d'aquests sistemes centralitzats del tractament d'aigua residual pot ser més o menys impactant depenent de l'escala d'aplicació i el grau de demanda. Per aquest motiu, en un estudi específic en assentaments urbans temporals a l'illa de Menorca s'han estudiat ambientalment diferents escenaris de tractament de l'aigua residual amb sistemes centralitzats o descentralitzats per tal de determinar les millors opcions per fer front a les èpoques turístiques. Els resultats depenen dels indicadors estudiats i les variables considerades. Tot i així, sembla que un escenari centralitzat en què es connecta l'assentament a un depuradora existent de gran capacitat és beneficiós degut a les economies d'escala. Segons una puntuació ambiental ponderada, aquest escenari generaria 390 mPt/m^3, que representa un 12% menys que descentralitzar parcialment amb fosses sèptiques o un 36% respecte a tractar el flux estacional en uns aiguamolls construïts. No obstant, mitjançant un estudi probabilístic s'ha variat la població punta, la generació d'aigua residual estival i la duració de l'època turística per tal d'extrapolar a altres zones similars. En aquest cas, tractar l'aigua en fosses sèptiques i transportar l'efluent a la depuradora veïna pot presentar els millors resultats mitjans (520 mPt/m^3). En general, els resultats depenen de la duració de l'època turística. L'escenari centralitzat és el que més variabilitat presenta (77%) en funció de la generació d'aigua, que va lligada al bombeig. És per això que cal estudiar de manera diferenciada els sistemes urbans temporals i els permanents.

Anàlisi de les inundacions des d'una perspectiva de cicle de vida

En l'àmbit de la prevenció d'inundacions, la tesi aporta una nova visió i un enfoc innovador, ja que es tracta d'uns dels primers estudis que integren l'impacte ambiental i econòmic d'invertir en mesures preventives amb l'impacte dels danys evitats. Aquests estudis poden ser de gran interès de cara a la planificació urbana i, en particular, depenent dels climes i entorns. Des d'una perspectiva metodològica, els estudis d'inundacions aporten una discussió en l'àmbit de les metodologies d'ACV i en com abordar les conseqüències de les inundacions des d'un punt de vista integrador i de la sostenibilitat de les ciutats.

Balanç ambiental i econòmic dels sistemes de prevenció d'inundacions

Per determinar el balanç net d'impactes ambientals i costos econòmics de determinades accions preventives, s'han estudiat dues zones diferenciades. Per una banda, les rieres són una amenaça a la costa mediterrània. En aquest entorn, s'ha estudiat l'evolució històrica dels danys causats a tres municipis del Maresme (Catalunya) abans i després de la implementació d'actuacions d'emergència. Aquestes consistien en la reconstrucció de murs o la canalització i/o restauració de les rieres. Tot comparant els impactes i costos invertits en la prevenció i els danys evitats, s'observa que aquestes accions es podrien amortitzar econòmicament en 2 anys i ambientalment, en 25 anys. Aquest resultat posa de manifest una de les conseqüències econòmiques i ambientals d'invertir en sistemes intensius en materials i amb alta durabilitat, com el formigó. Pel contrari, també s'ha estudiat un sistema de prevenció verd existent a la ciutat de São Carlos (Brasil), on les elevades precipitacions resulten en gran quantitat de danys cada any. En aquest entorn, s'ha observat que la ràtio d'ecoeficiència del sistema (0.38 kg CO_2 eq/€) és menor que en el cas de les actuacions a les rieres (1.2 kg CO_2 eq/€). Tot i que no són sistemes equivalents, l'ús d'un sistema verd que pot captar carboni pot ser en alguns casos beneficiós, a banda que també fomenta la infiltració natural de l'aigua. En aquest darrer cas, es va comprovar que l'impacte del sistema pot ser amortitzat un cop s'evita la destrucció d'un cotxe en termes de petjada de carboni.

Reptes de futur en la recerca del sanejament i drenatge urbà

Els resultats obtinguts obren un ventall de noves línies de recerca per al futur. És interessant determinar si les tendències observades en el clavegueram de ciutats mitjanes també tenen lloc a grans ciutats, on la complexitat urbana és major. Per altra banda, el debat de la centralització en zones periurbanes presenta més reptes de futur, com ara estudiar-ne les possibilitats de tractament d'aigua a diferents escales, climes i entorns turístics, així com la simbiosi que els sistemes naturals poden aportar en l'emmagatzemament d'aigua pluvial per un ús descentralitzat. En aquest àmbit, caldria també entendre les aplicacions de les aigües pluvials i/o regenerades en l'agricultura i els serveis per tancar el cicle de l'aigua de la manera més sostenible en sistemes urbans. En el cas de les inundacions, calen més models predictius que permetin determinar les conseqüències concretes de les inundacions i la seva prevenció. Per últim, l'enfocament de cicle de vida d'aquesta tesi es podria completar analitzant els serveis ecosistèmics (és a dir, beneficis per les persones) de diferents alternatives de sanejament i drenatge. D'aquesta manera, els seus beneficis i impactes quedarien clarament establerts de cara a la presa de decisions a escala urbana.

Resumen

El crecimiento de las ciudades alrededor del mundo lleva asociado un incremento en la demanda de infraestructuras de saneamiento y drenaje asociadas al ciclo del agua. Combinado con los efectos del cambio climático, que están alterando los patrones de pluviometría, la situación de estos sistemas en entornos urbanos es crítica. Buena parte de las redes de alcantarillado existentes requieren una renovación urgente, otras han de ser construidas en zonas en crecimiento, mientras que la escorrentía superficial de agua pluvial es una amenaza en cuanto a inundaciones debido a la impermeabilización del suelo. En este contexto, se debe determinar a través de una nueva visión ambiental y económica cuáles son las mejoras prácticas para reducir estas problemáticas y al mismo tiempo adaptar a las ciudades al cambio climático.

Aun así, existe una falta de datos ambientales y económicos que describan los principales impactos de la gestión de las redes de alcantarillado, así como de diferentes sistemas de prevención de inundaciones. Con este fin, la combinación de diferentes herramientas es clave en la obtención de información integrada sobre estas infraestructuras. Además, también es esencial determinar las implicaciones de diferentes escenarios de gestión del saneamiento y drenaje a escala urbana, periurbana, barrio y edificio.

En respuesta a estas demandas, la siguiente tesis se desarrolla en torno a las siguientes preguntas, las cuales se estudian de acuerdo a la estructura de la **Figura X1**:

(1) ¿Cuáles son los puntos críticos ambientales y económicos de las redes de alcantarillado y los parámetros que afectan estos resultados en ciudades medianas?

(2) ¿Es el saneamiento descentralizado una opción ambientalmente mejor en zonas suburbanas pequeñas cuando se debe gestionar una demanda punta?

(3) ¿Tienen las estrategias de prevención de inundaciones un comportamiento ambiental y económico positivo cuando se contabilizan las pérdidas evitadas en inundaciones? En caso afirmativo, ¿es mejor aplicar prácticas de gestión verdes o grises?

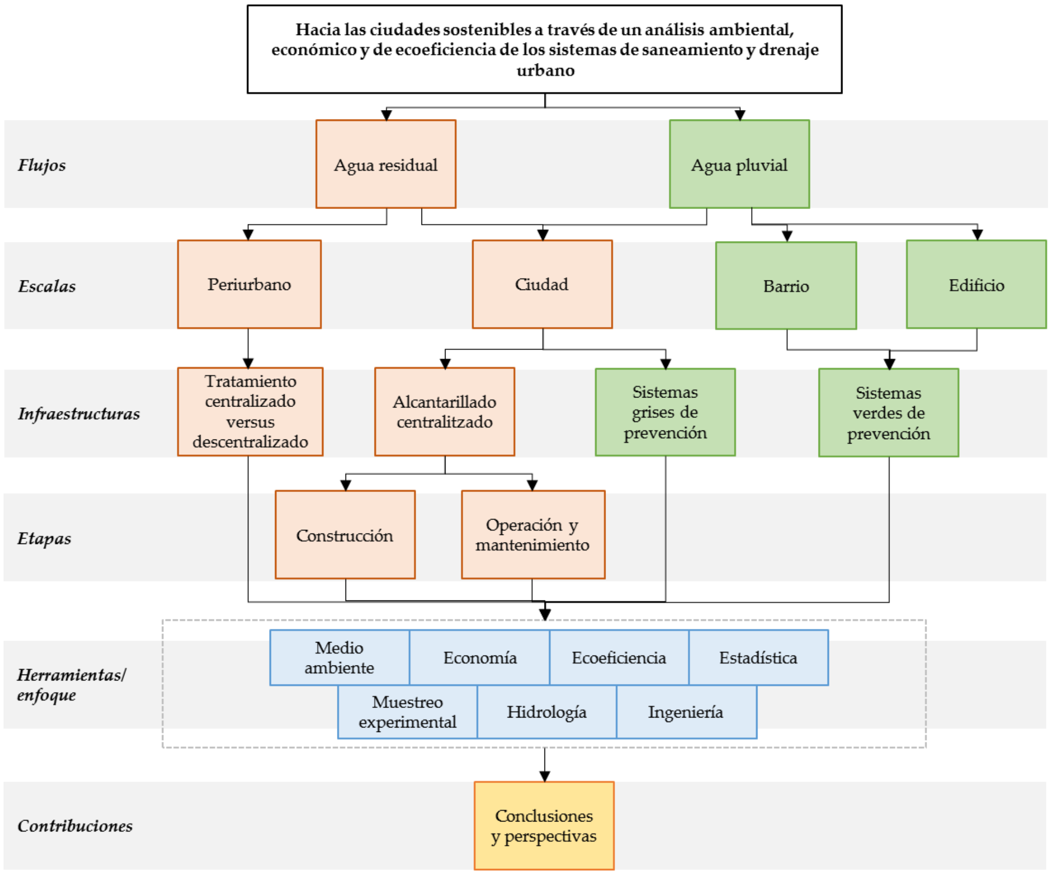


Figura X1 Elementos estudiados en la tesis, herramientas y contribuciones

Seguidamente, se presenta un resumen en el cual se recogen los principales métodos y resultados obtenidos en esta tesis doctoral en el marco de la ciencia y tecnología ambientales.

Materiales y métodos. Integración de herramientas

En esta tesis doctoral se usa el marco de la ecología industrial, aplicando métodos específicos como el análisis del ciclo de vida (ACV) (atribucional y consecuencial), el análisis de costes del ciclo de vida (ACCV) y la ecoeficiencia a los sistemas de saneamiento y drenaje urbano. Esta investigación interdisciplinaria requiere la incorporación de métodos adicionales, como estudios estadísticos o análisis experimentales en el campo para obtener datos reales sobre la emisión de gases de efecto invernadero (GEI) provenientes del alcantarillado. Dentro de este marco integrador con perspectiva de ciclo de vida se incluyen aplicaciones provenientes de la ingeniería y la hidrología. Las escalas de análisis incluyen desde las ciudades y zonas periurbanas hasta los barrios y edificios dependiendo del estudio.

La etapa de construcción del alcantarillado

El ciclo de vida de las redes de alcantarillado han sido ampliamente analizado en ciudades de tamaño mediano. Por ejemplo, de entre los resultados se ha estimado que un metro de solución constructiva de 300 mm de diámetro puede generar hasta 120 kg CO_2 eq si es de hormigón, o hasta 190 kg CO_2 eq si es de polietileno de alta densidad. En cuanto a las tuberías, generalmente el hormigón se presenta como la opción ambientalmente mejor frente a los plásticos debido a su larga vida útil, aunque depende de los casos considerados. No obstante, se ha determinado que los materiales de la tubería no son los únicos determinantes del impacto ambiental de una solución constructiva para alcantarillado. En algunos casos, la contribución de la zanja puede representar hasta un 80% de los impactos ambientales de la etapa constructiva, hecho que no es irrelevante de cara a la toma de decisiones. Mediante un estudio estructural paramétrico se han podido encontrar las soluciones constructivas equivalentes que generan el menor impacto ambiental. Así, reducir el uso de hormigón en las zanjas y reutilizar los materiales del suelo excavado para rellenar y proteger las tuberías puede significar una reducción de los impactos ambientales.

La etapa de operación y mantenimiento de las redes de alcantarillado relacionadas a cada ciudad

La etapa de operación incluye importantes retos en el ámbito del planeamiento urbano. Estudiando una muestra de municipios de España, en general, y dos municipios contrastados, en particular, se observa que el planeamiento es clave en el consumo de energía. Se han comparado el municipio costero de Calafell (clima mediterráneo) y Betanzos (clima atlántico). En este caso, la ubicación de la estación depuradora de Calafell a una cota más elevada que el municipio hace que el consumo de energía de bombeo (0.47 kWh/m^3) sea mayor que en Betanzos (0.11 kWh/m^3), donde el agua circula por gravedad. En una línea de investigación adicional, se han observado emisiones de GEI en el alcantarillado a través de campañas de muestreo en estos dos municipios de climas diferentes. Estos GEI incluyen el metano y el monóxido de dinitrógeno, los cuales tienen un potencial de calentamiento global 25 y 298 más elevado que el dióxido de carbono, respectivamente. Principalmente se detectaron mayores emisiones durante el verano asociadas a las elevadas temperaturas y en zonas de turbulencia del alcantarillado, como antes de las estaciones de bombeo.

Innovación a través de la ecoeficiencia de redes

La combinación de valores ambientales y económicos del alcantarillado se lleva a cabo a través de uno de los primeros estudios de ecoeficiencia que aplica la ISO 14045:2012. En los casos de Calafell y Betanzos se ha comparado la ecoeficiencia tanto de las diferentes etapas del ciclo de vida como de los valores totales de los municipios. Independientemente del clima y la estructura urbana, la etapa de operación es la que genera más impactos ambientales (hasta el 74% de los impactos), mientras que la instalación (es decir, la zanja) contribuye más a la generación de costes económicos (70-75%). La operación tiene margen de mejora en cuanto al consumo de energía asociada al bombeo, mientras que a los costes de la instalación dependen de la mano de obra. Esta contribución en los costes de ciclo de vida es especialmente relevante en el contexto del ciclo urbano del agua completo, dado que pueden representar hasta un 45% de los costes de estos municipios. En este sentido, se discute la necesidad de gestionar el alcantarillado en el contexto de la gestión integrada de las aguas residuales para mejorar la ecoeficiencia del sistema.

Tratamiento del agua residual centralizado frente a descentralizado en la esfera ambiental

La implementación de estos sistemas centralizados del tratamiento del agua residual puede ser más o menos impactante dependiendo de la escala de aplicación y el grado de demanda. Por este motivo, en un estudio específico en asentamientos urbanos temporales en la isla de Menorca se han estudiado ambientalmente diferentes escenarios de tratamiento del agua residual con sistemas centralizados o descentralizados para determinar las mejores opciones para hacer frente a las épocas turísticas. Los resultados dependen de los indicadores estudiados y las variables consideradas. Aun así, parece que un escenario centralizado en el que se conecta el asentamiento a una depuradora existente de gran capacidad es beneficioso debido a las economías de escala. Según una puntuación ambiental ponderada, este escenario generaría 390 mPt/m^3, que representa un 12% menos que descentralizar parcialmente con fosas sépticas o un 36% respecto a tratar el flujo estacional en un humedal construido. No obstante, mediante un estudio probabilístico se ha variado la población punta, la generación de agua residual estival y la duración de la época turística para extrapolar a otras zonas similares. En este caso, tratar el agua en fosas sépticas y transportar el efluente a la depuradora vecina puede presentar los mejores resultados medios (520 mPt/m^3). En general, los resultados dependen de la duración de la época turística. El escenario centralizado es el que más variabilidad presenta (77%) en función de la generación del agua, que va ligada al bombeo. Por este motivo, se deben estudiar de forma diferenciada los sistemas urbanos temporales y los permanentes.

Análisis de las inundaciones desde una perspectiva de ciclo de vida

En el ámbito de la prevención de inundaciones, se aporta una nueva visión y un enfoque innovador, pues se trata de los primeros estudios que integran el impacto ambiental y económico de invertir en medidas preventivas con el impacto de los daños evitados. Estos estudios pueden ser de gran interés de cara a la planificación urbana y, en particular, dependiendo de climas y entornos. Desde una perspectiva metodológica, los estudios de inundaciones aportan una discusión en el ámbito de las metodologías de ACV y en cómo abordar las consecuencias de las inundaciones desde un punto de vista integrador y de la sostenibilidad de las ciudades.

Balance ambiental y económico de los sistemas de prevención de inundaciones

Para determinar el balance neto de los impactos ambientales y costes económicos de determinadas acciones preventivas, se han estudiado dos zonas diferenciadas. Por un lado, las rieras son una amenaza en la costa mediterránea. En este entorno, se ha estudiado la evolución histórica de los daños causados en tres municipios del Maresme (Cataluña, España) antes y después de la implementación de actuaciones de emergencia. Comparando los impactos y costes invertidos en la prevención y los daños evitados, se observa que estas acciones se podrían amortizar económicamente en 2 años y ambientalmente, en 25 años. Este resultado pone de manifiesto una de las consecuencias ambientales y económicas de invertir en sistemas intensivos en materiales y de larga durabilidad, como el hormigón. Por el contrario, también se ha estudiado un sistema de prevención verde existente en la ciudad de São Carlos (Brasil), donde las elevadas precipitaciones resultan en gran cantidad de daños cada año. En este entorno, se ha observado que la ratio de ecoeficiencia del sistema (0.38 kg CO_2 eq/€) es menor que en el caso de las actuaciones en las rieras (1.2 kg CO_2 eq/€). Aunque no son sistemas equivalentes, el uso de un sistema verde que puede captar carbono puede ser beneficioso en algunos casos, además de que también fomenta la infiltración natural del agua. En este último caso, se comprobó que el impacto del sistema puede ser amortizado una vez se evita la destrucción de un coche en términos de huella de carbono.

Retos de futuro en la investigación del saneamiento y drenaje urbano

Los resultados obtenidos abren un abanico de nuevas líneas de investigación de futuro. Es interesante determinar si las tendencias observadas en el alcantarillado de ciudades medianas también tienen lugar en grandes ciudades, donde la complejidad urbana es mayor. Por otro lado, el debate de la centralización en zonas periurbanas presenta más retos de futuro, como estudiar las posibilidades del tratamiento de agua a diferentes escalas, climas y entornos turísticos, así como la simbiosis que los sistemas naturales pueden aportar en el almacenaje del agua pluvial para un uso descentralizado. En este ámbito, se deberían comprender las aplicaciones de las aguas pluviales y/o regeneradas en la agricultura y servicios para cerrar el ciclo del agua de la manera más sostenible en sistemas urbanos. En el caso de las inundaciones, son necesarios más modelos predictivos que permitan determinar las consecuencias concretas de las inundaciones y su prevención. Por último, el enfoque de ciclo de vida de esta tesis se podría completar analizando los servicios ecosistémicos (es decir, beneficios para las personas) de diferentes alternativas de saneamiento y drenaje. De esta forma, sus beneficios e impactos quedarían claramente establecidos en vistas a la toma de decisiones a escala urbana.

Preface

This thesis was developed during the period from October 2013 to April 2017 in compliance with the PhD program in Environmental Science and Technology of the Universitat Autònoma de Barcelona. This training period took place within the Sostenipra research group (2014 SGR 1412) at the Institute of Environmental Science and Technology (ICTA), including an international mobility at the University of Toledo (Ohio, USA) from April to June 2016, and teaching assistance at the Department of Chemical, Biological and Environmental Engineering. The thesis was supported by the pre-doctoral fellowships awarded by Generalitat de Catalunya (FI-DGR 2014; March to September, 2014) and the Spanish Ministry of Education, Culture and Sports (FPU13/01273; September, 2014 – until end of training). In addition, research was conducted in a "María de Maeztu" Unit of Excellence in R&D (MDM-2015-0552) thanks to the support of the Spanish Ministry of Economy and Competitiveness.

The dissertation addresses urban wastewater and drainage systems through an integrated approach that combines environmental and economic costs and benefits of different urban water systems. The novelty of this research is the assessment of a set of conventional and alternative solutions using innovative and integrated methods that will support decision-making at different urban scales. The systems under analysis are thoroughly analyzed to provide sound results. Additionally, engineering, hydrologic, urban planning and experimental tools were applied accordingly.

Different parts of the dissertation were elaborated in the framework of funded research projects. The eco-efficiency assessment of sewers was conducted within the LIFE+ Aquaenvec project (LIFE10/ENV/ES/520) "Assessment and improvement of the urban water cycle ecoefficiency using LCA and LCC". The stormwater management studies based on Brazilian case studies were supported by the "Smart Parks" project funded by the Spanish (HBP-2012-0216) and Brazilian governments (CAPES ref. 5206).

This thesis is composed of five parts as illustrated in **Figure X2**. Each part consists of a group of chapters that deal with a variety of contents related to the main topic of the dissertation.

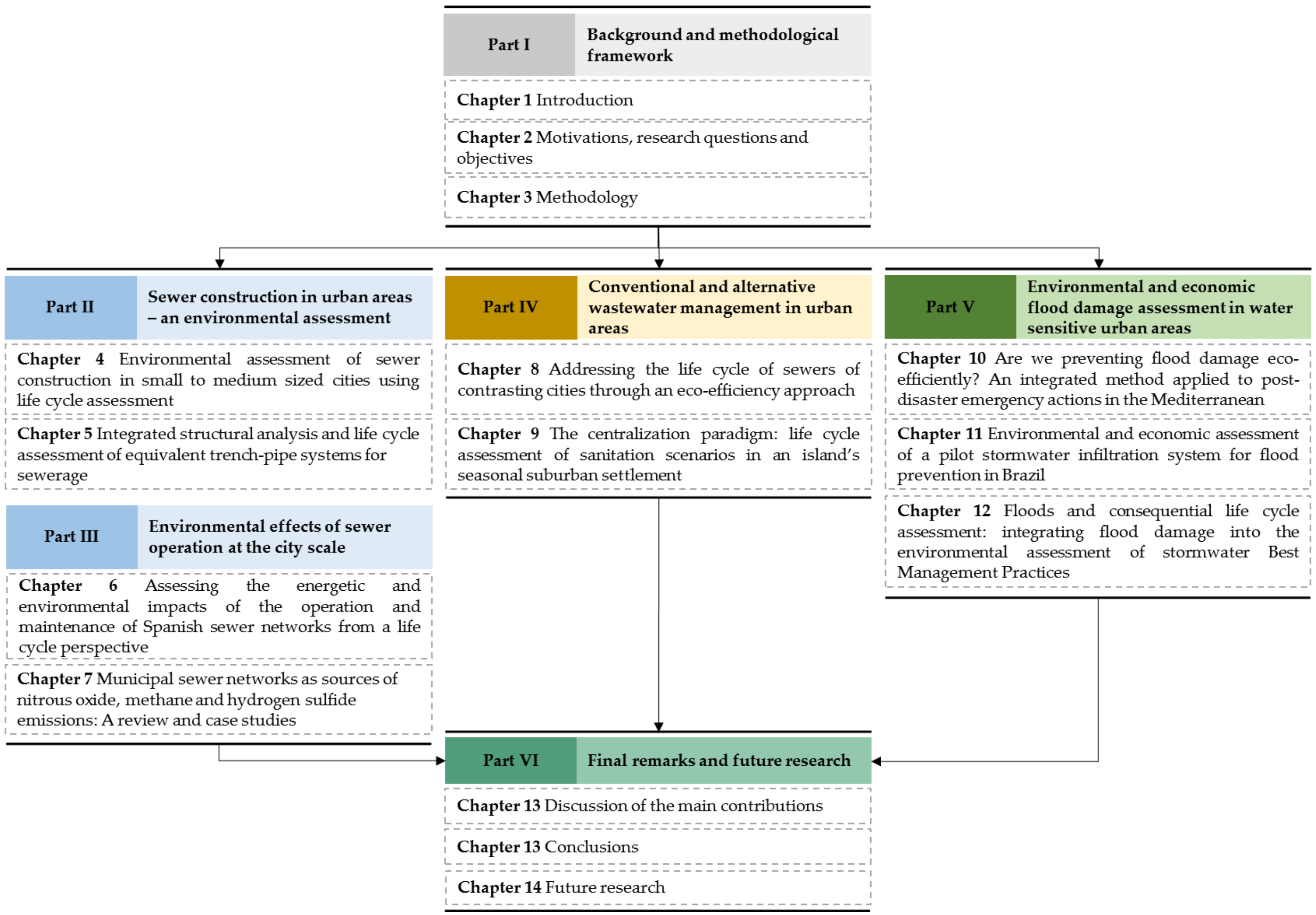


Figure X2 Structure of the dissertation

Part I. Background and methodological framework

Part I is composed of three chapters. **Chapter 1** [*Introduction*] explains the issues associated with sustainability in cities and the potentialities of industrial ecology for quantifying urban sustainability. It later focuses on the effects of population growth and climate change on current sanitation and drainage systems and discusses the need for environmentally sound infrastructure. The motivations that inspired the elaboration of this dissertation are presented in **Chapter 2** [*Motivations, research questions and objectives*], along with the questions that addresses this research. **Chapter 3** [*Methodology*] provides a general description of the tools applied in each chapter and the case studies under analysis.

Part II. Sewer construction in urban areas – an environmental assessment

Part II addresses the environmental impacts of the construction of sewer networks. First, **Chapter 4** [*Environmental assessment of sewer construction in small to medium sized cities using life cycle assessment*] conducts a life cycle assessment (LCA) of constructive solutions and compares different types of pipe materials (i.e., concrete, fibrocement, polyvinylchloride, and high-density polyethylene), trench designs and diameters. Besides comparing, it offers a basic method for calculating the impacts of urban sewers that is tested on a case study. **Chapter 5** [*Integrated structural analysis and life cycle assessment of equivalent trench-pipe systems for sewerage*] integrates LCA and a parametric study to obtain technically feasible designs based on construction regulations. Equivalent designs are compared and assessed under traffic stress conditions and pavement degradation.

Part III. Environmental effects of sewer operation at the city scale

Part III deals with the elements involved in the operation of sewers. **Chapter 6** [*Assessing the energetic and environmental impacts of the operation and maintenance of Spanish sewer networks from a life cycle perspective*] is a statistical analysis of a sample of cities that aims to determine the trends and parameters that affect the energy consumption in sewers. **Chapter 7** [*Municipal sewer networks as sources of nitrous oxide, methane and hydrogen sulfide emissions: A review and case studies*] has two main parts. First, it reviews current literature on greenhouse gas emissions generated in sewers to understand the potential sources of methane and nitrous oxide that result from wastewater degradation. Second, it provides the first experimental results of sampling campaigns conducted in two case study cities to determine the main drivers for these emissions and their contribution to a sewer's life cycle impacts.

Part IV. Conventional and alternative wastewater management in urban areas

Part IV is composed of two different studies. **Chapter 8** [*Addressing the life cycle of sewers of contrasting cities through an eco-efficiency approach*] applies the eco-efficiency method to two cities with a centralized sewer system by combining environmental and economic values. This study aims to find differences between these cities and determine the main reasons. **Chapter 9** [*The centralization paradigm: life cycle assessment of sanitation scenarios in an island's seasonal suburban settlement*] discusses the environmental performance of centralized and decentralized sanitation scenarios in the context of suburban settlements in islands that deal with an increased sanitation demand during the seasonal periods.

Part V. Environmental and economic flood damage assessment in water-sensitive urban areas

Part V addresses the environmental and economic effects of stormwater management systems under different conditions. **Chapter 10** [*Are we preventing flood damage eco-efficiently? An integrated method applied to post-disaster emergency actions in the Mediterranean*] presents a novel approach that estimates the environmental and economic balance and payback time of post-disaster emergency actions. These are applied in ephemeral streams in the Mediterranean and feed on historical records and insurance data. **Chapter 11** [*Environmental and economic assessment of a pilot stormwater infiltration system for flood prevention in Brazil*] is based on an existing green prevention system located in Brazil. The chapter estimates the impacts of this system and its potential infiltration benefits. **Chapter 12** [*Floods and consequential life cycle assessment: integrating flood damage into the environmental assessment of stormwater Best Management Practices*] discusses the integration of flood damage into the CLCA of prevention systems and proposes a methodology that is tested on the case study presented in **Chapter 11**.

Part VI. Final remarks and future research

Part VI is the last section of the dissertation. **Chapter 13** [*Discussion of the main contributions*] brings together the knowledge generated in this research to present the main contributions and applications. **Chapter 14** [*Conclusions*] answers the questions posed in **Chapter 2** based on the results of the analysis. Finally, **Chapter 15** [*Future research*] proposes upcoming research challenges and applications that might be interesting in the field of urban sustainability and the water cycle resulting from the work conducted.

Dissemination and training

This thesis is based on a set of peer-reviewed or submitted articles and the inventory data published at the LCADB.Sudoe database (http://lcadb.sudoe.ecotech.cat):

- ✓ **Petit-Boix A**, Sanjuan-Delmás D, Gasol C, Villalba G, Suárez-Ojeda ME, Gabarrell X, Josa A, Rieradevall J, 2014. Environmental assessment of sewer construction in small to medium sized cities using life cycle assessment. Water Resources Management 28, 979–997 (doi: 10.1007/s11269-014-0528-z)
- ✓ **Petit-Boix A**, Sanjuan-Delmás D, Chenel S, Marín D, Gasol C, Farreny R, Villalba G, Suárez-Ojeda ME, Gabarrell X, Josa A, Rieradevall J, 2015. Assessing the energetic and environmental impacts of the operation and maintenance of Spanish sewer networks from a life-cycle perspective. Water Resources Management 29, 2581–2597 (doi: 10.1007/s11269-015-0958-2)
- ✓ Eijo-Río E, **Petit-Boix A**, Villalba G, Suárez-Ojeda ME, Marin D, Amores MJ, Aldea X, Rieradevall J, Gabarrell X, 2015. Municipal sewer networks as sources of nitrous oxide, methane and hydrogen sulphide emissions: A review and case studies. Journal of Environmental Chemical Engineering 3(3): 2084–2094 (doi: 10.1016/j.jece.2015.07.006)
- ✓ **Petit-Boix A**, Sevigné-Itoiz E, Rojas-Gutierrez LA, Barbassa AP, Josa A, Rieradevall J, Gabarrell X, 2015. Environmental and economic assessment of a pilot stormwater infiltration system for flood prevention in Brazil. Ecological Engineering 84, 194-201 (doi: 10.1016/j.ecoleng.2015.09.010)
- ✓ **Petit-Boix A**, Roigé N, de la Fuente A, Pujadas P, Gabarrell X, Rieradevall J, Josa A, 2016. Integrated structural analysis and life cycle assessment of equivalent trench-pipe systems for sewerage. Water Resources Management 30, 1117–1130 (doi: 10.1007/s11269-015-1214-5)
- ✓ **Petit-Boix A**, Arahuetes A, Josa A, Rieradevall J, Gabarrell X, 2017. Are we preventing flood damage eco-efficiently? An integrated method applied to post-disaster emergency actions. Science of the Total Environment 580, 873-881 (doi: 10.1016/j.scitotenv.2016.12.034)
- ✓ **Petit-Boix A**, Sevigné-Itoiz E, Rojas-Gutierrez LA, Barbassa AP, Josa A, Rieradevall J, Gabarrell X. Floods and consequential Life Cycle Assessment: integrating flood damage into the environmental assessment of stormwater Best Management Practices. Accepted with minor revisions
- ✓ **Petit-Boix A**, Arnal C, Marin D, Josa A, Gabarrell X, Rieradevall J. Addressing the life cycle of sewers through an eco-efficiency approach. The case of opposite urban environments. Accepted with major revisions

In addition, some preliminary results were presented in international conferences of interest:

- ✓ Sanjuan-Delmás D, **Petit-Boix A**, Marín D, Gabarrell X, Josa A, Rieradevall J "Drinking water and sewer networks in Spanish medium-sized cities: an environmental assessment from an industrial ecology perspective" Oral presentation. ISIE Americas 2016 Meeting (May 25-27, 2016; Bogotá, Colombia)
- ✓ Sanjuan-Delmás D, **Petit-Boix A**, Marín D, Gabarrell X, Josa A, Rieradevall J "Environmental assessment of drinking water supply and sewer networks for small to medium cities". Poster.

LCM2015. Mainstreaming Life Cycle Management for sustainable value creation (August 1 – September 2, 2015; Bordeaux, France)

- ✓ Arnal C, Marín D, Termes M, Planas M, **Petit-Boix A**, Sanjuan-Delmás A, Amores MJ, Prieto M, Aldea X "Life Cycle Costing assessment of water supply and sewer networks in small-medium cities" Poster. Mainstreaming Life Cycle Management for sustainable value creation (August 1 – September 2, 2015; Bordeaux, France)
- ✓ **Petit-Boix A**, Sanjuan-Delmás D, Arnal C, Marín D, Gabarrell X, Josa A, Rieradevall J "LCA and LCC Integration for Supporting Decisions in the Design and Construction of Sewer Networks in Future Smart Cities" Poster. CILCA2015. VI Conferencia Internacional de Análisis de Ciclo de Vida en Latinoamérica (July 13-16, 2015; Perú, Lima) – ***Third place at poster awards***
- ✓ Eijo-Río E, **Petit-Boix A**, Amores MJ, Marín D, Villalba G, Suárez-Ojeda ME, Rieradevall J, Gabarrell X "Gas emissions in municipal sewer networks in two climatic regions" Poster. SETAC Europe 25th Annual Meeting (May 3-7, 2015; Barcelona, Spain)
- ✓ Rojas-Gutierrez LA, **Petit-Boix A**, Barbassa AP, Teixeira B, Josa A, Rieradevall J, Gabarrell X "Environmental assessment of stormwater infrastructures built with the best management practices" Oral presentation. ICFM 6. 6th International Conference on Flood Management. "Floods in a changing Environment" (September 16-18, 2014; São Paulo, Brazil)
- ✓ **Petit-Boix A**, Sevigné E, Barbassa AP, Teixeira B, de Lima Nascimento Sirio D, Rojas-Gutierrez LA, Marin D, Rieradevall J, Gabarrell X "Environmental consequences of stormwater harvesting for flood prevention" Oral presentation. SETAC Europe 24th Annual Meeting (May 11-16, 2014; Basel, Switzerland)

Furthermore, additional training and knowledge was obtained through collaborations in a number projects, articles, conferences, books, and scientific activities during the PhD training process.

Participation in projects:

- ✓ LIFE SAVING-E: Two-Stage Autotrophic N-remoVal for maINstream sewaGe treatment (LIFE14-ENV_ES_000633). EU funds
- ✓ Desdemona: DESarrollo de una DEpuradora urbana autosuficiente energéticamente Mediante la eliminación autOtrófica de Nitrógeno en la línea principal de Aguas y la recuperación de fósforo. MINECO funds
- ✓ Collaborative Research: IRES (International Research Experience for Undergraduates) - Life Cycle Management and Ecosystem Services Applied to Urban Agriculture. National Science Foundation funds
- ✓ Fertilecity (CTM2013-47067-C2-1-R). Agrourban sustainability through rooftop greenhouses. Ecoinnovation on residual flows of energy, water and CO_2 for food production. MINECO funds
- ✓ Collaborative Research: Analysis of Decentralized Harvested Rainwater Systems using the Urban Water Infrastructure Sustainability Evaluation (uWISE) Framework. National Science Foundation funds

- ✓ M-ECO, Investigación para la mejora de la sostenibilidad del sector de la madera y mueble en Andalucía a través de la Eco-innovación. Junta de Andalucía funds

Participation in articles and book chapters:

- ✓ **Petit-Boix A**, Sanyé-Mengual E, Llorach-Massana P, Sanjuan-Delmás D, Sierra-Pérez J, Vinyes E, Gabarrell X, Rieradevall J. Research on strategies towards sustainable cities from a life cycle perspective: a review. Accepted with major revisions
- ✓ Devkota J, **Petit-Boix A**, Phillips R, Vargas-Parra V, Josa A, Gabarrell X, Rieradevall J, Apul D. Life Cycle Assessment of Rainwater Harvesting in Urban Neighborhoods: Implications of Urban Form and Water Demand Patterns in the US and Spain. Under review
- ✓ Angrill S, **Petit-Boix A**, Morales-Pinzón T, Josa A, Rieradevall J, Gabarrell X, 2017. Urban rainwater runoff quantity and quality – A potential endogenous resource in cities? Journal of Environmental Management 189, 14-21 (doi: 10.1016/j.jenvman.2016.12.027)
- ✓ Angrill S, Segura-Castillo L, **Petit-Boix A**, Rieradevall J, Gabarrell X, Josa A, 2017. Environmental performance of rainwater harvesting strategies in Mediterranean buildings. International Journal of Life Cycle Assessment 22(3), 398-409 (doi: 10.1007/s11367-016-1174-x)
- ✓ Sanjuan-Delmás D, **Petit-Boix A**, Martínez-Blanco J, Rieradevall J, 2016. Environmental metabolism of educational services. Case study of nursery schools in the city of Barcelona. Energy Efficiency 9(5), 981-992 (doi: 10.1007/s12053-015-9403-x)
- ✓ Sanjuan-Delmás D, **Petit-Boix A**, Gasol CM, Farreny R, Villalba G, Suárez-Ojeda ME, Gabarrell X, Josa A, Rieradevall J, 2015. Environmental assessment of drinking water transport and distribution network use phase for small to medium-sized municipalities in Spain. Journal of Cleaner Production 87, 573-582 (doi: 10.1016/j.jclepro.2014.09.042)
- ✓ Sanjuan-Delmás D, **Petit-Boix A**, Gasol CM, Villalba G, Suárez-Ojeda ME, Gabarrell X, Josa A, Rieradevall J, 2014. Environmental assessment of different pipelines for drinking water transport and distribution network in small to medium cities: a case from Betanzos, Spain. Journal of Cleaner Production 66, 588-598 (doi: 10.1016/j.jclepro.2013.10.055)
- ✓ **Petit-Boix A**, Sanjuan-Delmás D, Martínez-Blanco J, Rieradevall J, 2013. La petjada ambiental d'un infant a l'escola bressol. Guix d’Infantil 73, 33-36
- ✓ Gabarrell X, Rieradevall J, Josa A, Oliver-Solà J, Mendoza JMF, Sanjuan-Delmás D, **Petit-Boix A**, Sanyé-Mengual E, 2015. Life Cycle Management Applied to Urban Fabric Planning, In: Life Cycle Management. Part V: Implementation and Case Studies of Life Cycle Management in Different Business and Industry Sectors. In LCA Compendium - The Complete World of Life Cycle Assessment (ISBN: 978-94-017-7220-4). Pages 307-317

Collaboration in journals

From November, 2015, until present, the abstracts of the Journal of Industrial Ecology were translated into Spanish.

Yale JOURNAL OF INDUSTRIAL ECOLOGY

Resúmenes en Español
Revista de Ecología Industrial
Volumen 21, Número 2

Traducido / organizado por

Spanish Abstracts
Journal of Industrial Ecology
Volume 21, Issue 2

Translated / organized by

Xavier Gabarrell Durany (UAB), Bart van Hoof (UA), Cristina Madrid-López (Yale),
David Sanjuan Delmas (UAB), Jorge Sierra (UAB, CUD-Zaragoza),
Viviana Salas (UA), Anna Petit Boix (UAB), Violeta Vargas (UPC),
Santiago Londoño (UA), Esther Sanyé-Mengual (UNIBO),
Joan Rieradevall (UAB), Gara Villalba (UAB), Pere Muñoz (IRTA)

© 2017 by Yale University

www.wileyonlinelibrary.com/journal/jie Spanish Abstracts

Participation in conferences:

- ✓ Uzunogullari D, Apul D, Gruden C, **Petit-Boix A** "Net Environmental Impact of Urban and Conventional Agriculture Estimated Using Urban Water Metabolism" Poster. 2017 Borchardt Conference. 24th Triennial Symposium on Advancements in Water & Wastewater (February 21-22, 2017; Ann Arbor, Michigan, US)
- ✓ Duch C, Peña A, Roch M, Nieto S, Rieradevall J, **Petit-Boix A** "Eco-efficiency of the meat and leather production in a transitional stockbreeding system located in a Mediterranean island" Poster. LCAFood2016. 10th International Conference on Life Cycle Assessment of Food 2016 (October 19-21, 2016; Dublin, Ireland)
- ✓ **Petit-Boix A**, Feng Y, Burian S, Apul D, Josa A, Rieradevall J, Gabarrell X "Environmental map of water and food self-sufficiency: integrating rainwater harvesting and urban agriculture in different climates" Poster. LCAFood2016. 10th International Conference on Life Cycle Assessment of Food 2016 (October 19-21, 2016; Dublin, Ireland)
- ✓ Benítez M, Codina G, Domènech J, Moreno J, Sanz S, Rieradevall J, **Petit-Boix A** "Environmental and eco-efficiency assessment of artisan cheese production with Protected Designation of Origin in the Mediterranean island of Minorca" Poster. LCAFood2016. 10th International Conference on Life Cycle Assessment of Food 2016 (October 19-21, 2016; Dublin, Ireland)
- ✓ Toboso S, Aynès J, Liarte R, Torres C, Muñiz I, Rieradevall J, **Petit-Boix A** "Integrating geographic, social and environmental tools for urban garden sustainability in Barcelona" Oral presentation. GROWING IN CITIES. Interdisciplinary Perspectives on Urban Gardening International Conference (September 9-10, 2016; Basel, Switzerland)

- ✓ Sanjuan-Delmás D, **Petit-Boix A**, Rieradevall J, Gabarrell X "Ecoinnovación en el sector del agua en Sostenipra" Poster. META2016. XII Reunión de la Mesa Española de Tratamiento de Aguas (June 20-22, 2016; Madrid, Spain)
- ✓ Llorach-Massana P, **Petit-Boix A**, Sanyé-Mengual E, Rieradevall J, Gabarrell X, Garcia Lozano R, Gasol CM, Vázquez V, Rodríguez G, Rodríguez-Acuña R "Tools for implementing and communicating eco-design in the furniture sector" Oral presentation. ISIE Americas 2016 Meeting (May 25-27, 2016; Bogotá, Colombia)
- ✓ Feng Y, Burian S, **Petit-Boix A**, Josa A, Rieradevall J, Gabarrell X, Apul D "Life Cycle Assessment of the Cost and the Social and Ecosystem Services of Rainwater Harvesting in Dry Climates" Poster. EWRI. World Environmental & Water Resources Congress 2016 (May 22-26, 2016; Palm Beach, Florida, US)
- ✓ Sanjuan-Delmás D, **Petit-Boix A**, Arnal C, Marín D, Gabarrell X, Josa A, Rieradevall J "Ecoefficiency assessment of the drinking water distribution network of Calafell (Spain)" Poster. GCPC2015. Global Cleaner Production and Sustainable Consumption Conference 2015 (November 1-4, 2016; Sitges, Spain)
- ✓ **Petit-Boix A**, Sanyé-Mengual E, Llorach-Massana P, Rieradevall J, Gabarrell X, Garcia Lozano R, Gasol CM, Vázquez V, Rodríguez R, Rodríguez G "Sustainable production through innovation in the furniture sector: promoting policy-making and eco-design tools from a joint research-industry project" Oral presentation. GCPC2015. Global Cleaner Production and Sustainable Consumption Conference 2015 (November 1-4, 2016; Sitges, Spain)
- ✓ Sanyé-Mengual E, **Petit-Boix A**, Ometto A, Camargo Felicio M, Gonçalves LM,Siloto da Silva R, Bovo Peres R, Barbassa AP, Teixeira BAN, Rieradevall J, Gabarrell X "Towards the design of "Smart Parks": development of industrial ecology tools for industrial and service parks in Brazil and Spain" Oral presentation. CILCA2015. VI Conferencia Internacional de Análisis de Ciclo de Vida en Latinoamérica (July 13-16, 2015; Perú, Lima)
- ✓ Sanjuan-Delmás D, Arnal C, **Petit-Boix A**, Gabarrell X, Josa A, Rieradevall J "Comparison of HDPE and ductile iron pipes for drinking water supply networks in future smart cities through LCA and LCC" Poster. CILCA2015. VI Conferencia Internacional de Análisis de Ciclo de Vida en Latinoamérica (July 13-16, 2015; Perú, Lima)
- ✓ **Petit-Boix A**, Pereira Silva S, Ometto A, Yuri Hanai F, Barbassa AP, Josa A, Gabarrell X, Rieradevall J "Integrated assessment of water flows and urban water networks in smart parks" Oral presentation. ISIE Conference 2015. Taking Stock of Industrial Ecology (July 7-11, 2015; Guildford, UK)
- ✓ Sanjuan-Delmás D, **Petit-Boix A**, Gasol CM, Villalba V, Suárez-Ojeda ME, Gabarrell X, Josa A, Rieradevall J "LCA of PVC and HDPE pipes for drinking water distribution networks in cities" Poster. IWA World Water Congress & Exhibition (September 21-26, 2014; Lisbon, Portugal)

Part I

Background and methodological framework

Chapter 1

Picture: Residential area in the Village of Ottawa Hills (Ohio, USA)

Introduction

Chapter 1 Introduction

This chapter elaborates on the need to consider sustainability in urban water management. First, problems associated with urban expansion are presented in the context of sustainable development, considering the specific application of industrial ecology tools to improve the performance of cities. Second, the need for sustainable water management systems is discussed.

1.1 Cities and sustainability

It is no longer an assumption but a fact that our society is highly dependent on resources. We rely on technologies that improve our quality of life and wellbeing, but a cost is always associated with any decision. Deciding on a place to live or work or the type of products that we consume has a direct effect on the society, economy and environment. In the following sections, we explore the concept of sustainability, how it relates to cities, which are population hotspots, and specific tools that we can apply to quantify and evaluate the performance of a city.

1.1.1 Key issues towards urban sustainability

Sustainability is a central value in most political agendas at the national, regional and local level since the Brundtland Report defined in 1987 that sustainable development is a "development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (WCED 1987). To achieve sustainability, action plans should account for the environmental, economic and social effects that they might trigger and the potential interconnections among these three pillars. In this context, the Agenda 21 program, which resulted from the Río Conference in 1992, promoted the implementation of sustainability actions based on different program areas (United Nations, UN 1992).

Within the 2030 Agenda for Sustainable Development, the United Nations Development Program (UNDP) defined in 2015 a set of seventeen strategic goals that covered a variety of core dimensions related to sustainability. One of these was to make cities inclusive, safe, resilient and sustainable (United Nations, UN 2016). In fact, cities are known population hubs and a focus of concern given the increase in urban population that the world experienced during the last century. At present, more than half of the world's population lives in urban areas (The World Bank 2016) and this rate is not expected to decrease in the coming years. The expansion of cities and development of new metropolis is associated with an increasing demand for resources to meet the needs of potential new citizens.

The form these urban areas take will directly affect the per capita resource use and impact generation in the short and long term, which requires immediate action (McDonald 2008). Research has already acknowledged that cities contribute to local and global impacts, such as climate change or resource depletion (Kennedy et al. 2012). Nevertheless, the extent to which cities relate to these effects is still a controversial matter. For instance, UN-HABITAT (2011) reported that cities are responsible for 70% of the greenhouse gas emissions (GHG). Satterthwaite (2008) rebutted this claim by stating that we need to clearly define the boundaries of a city when accounting for its performance, as the production of food, energy or other products mainly occurs outside of the city's physical boundaries and might alter the results. According to this author, less than 50% of the GHG emissions are likely to be produced within city boundaries.

Given these concerns about cities, a number of studies have dealt with strategies, frameworks and political agendas aimed at increasing sustainability and resilience in urban areas (Satterthwaite 1997; Kousky and Schneider 2003; McDonald 2008; Alberti 2010; North 2010; Opschoor 2011). Interestingly, it is still not clear what a sustainable city is, although many definitions and characteristics have been suggested. Recent reports claim that sustainable cities should be conceived as the integration of social and economic development, environmental management, and urban governance (UN 2013). For a city to be sustainable, we must ensure urban quality and reduce the impact on global resources (Alberti 1996). Alberti and Susskind (1996) highlighted the need to reinvent and benefit from sustainability and its applications, this being a choice that requires leadership at any geographical level.

In this context, institutions and decision-makers should be provided with tools that enable the quantification and assessment of urban sustainability and the issues involved in their performance. Industrial ecology is one of the disciplines that facilitates this process, as discussed in the following section.

1.1.2 Industrial ecology applied at the city scale

The concept of 'industrial ecology' (IE) implies that human economic activities can be approached as ecological systems, seeking to integrate activities and cyclization of resources (Graedel 1996). One of the main differences between humans and other species is that human activities not only involve biological processes, but also an industrial metabolism (Rees and Wackernagel 1996). Allenby (1992) defined IE as a means to approach and maintain sustainable development. To this end, regulations, costs, social behavior, and design of products and processes should be interrelated to effectively implement the IE approach in industrialized systems (Frosch 1992).

In this sense, a multidisciplinary context is one of the key aspects of IE mentioned by Garner and Keoleian (1995). According to these authors, IE is composed of a set of additional concepts, i.e., systems analysis, material and energy flows and transformations, analogies to natural systems, and closed-loop systems. A number of tools support these processes, such as material flow analysis (MFA), life cycle methods (i.e., life cycle assessment (LCA), life cycle costing (LCC), social life cycle assessment (S-LCA) and life cycle sustainability assessment (LCSA)), design for the environment (DfE) or eco-efficiency.

When the concept of IE was developed in the early 90s, it mainly focused on the industrial metabolism. In the late 90s, urban metabolism (UM) became one of the metabolisms included within IE, along with industrial and socio-economic (Kennedy 2016). However, the concept of UM was not new, as it resulted from the first definition coined by Wolman (1965). In a broader context, Kennedy et al. (2007) defined UM as "the sum total of the technical and socio-economic processes that occur in cities, resulting in growth, production of energy, and elimination of waste". The mainstream UM school quantifies urban metabolic stocks and flows through simplified units to ultimately estimate urban sustainability indicators and GHG emissions, and improve the design and management of a city (Kennedy et al. 2010). Metabolic flows are composed of the inputs and outputs of energy, water, nutrients and waste, whereas stocks consist of built infrastructure, such as buildings, roads or water and energy supply systems (**Figure 1.1**). As a result, products and services are provided to the society. Within these stocks and flows, this dissertation deals with water systems, particularly wastewater transport and treatment, and stormwater drainage.

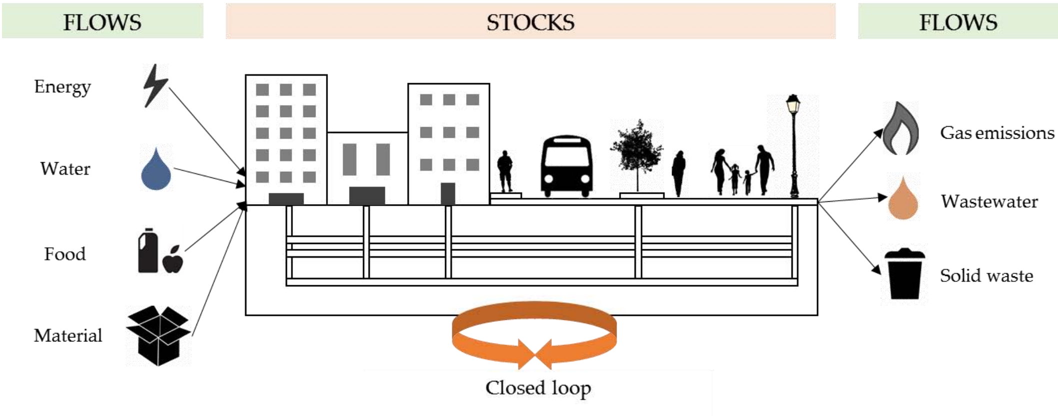


Figure 1.1 Simplified diagram of the metabolism of cities

1.2 The role of water in the metabolism of cities

Water is also included in the fifth strategic goal defined by the United Nations (2016), which calls for an increase in water quality, accessibility and sanitation standards. This section deals with the state of the art and challenges associated with the urban water cycle in the context of climate change adaptation and mitigation.

1.2.1 Water and cities: supply, sanitation and drainage

Increasing the access to water and sanitation involves the adaptation of the urban water cycle to the demand for these services. In general, the conventional structure of the urban water cycle consists of the stages shown in **Figure 1.2**. Freshwater is extracted from different types of water bodies depending on the area, such as rivers, lakes or reservoirs, or even seawater when desalination processes are the main water source. Once the potable water treatment plant (PWTP) provides water that complies with human consumption standards, potable water flows through a transport network that connects the PWTP with the city or cities that benefit from a centralized treatment. Additionally, a complex distribution network supplies water to the users. At the consumption point, water is used for potable (e.g., drinking, personal care) and non-potable purposes (e.g., toilet flushing, irrigation). These activities produce an outflow of urban wastewater, which includes wastewater generated by households, services and small industries (Council of the European Union 1991).

Sewers are responsible for transporting urban wastewater to centralized wastewater treatment plants (WWTPs), which are the most common treatment option in metropolis and medium-sized cities. Smaller cities might benefit from decentralized treatment systems, such as septic tanks, when connecting to a conventional WWTP is unfeasible due to their location and population size. This alternative is also common in low-income communities because of its reduced costs (Butler and Davies 2000). In conventional centralized systems, sewers might be either combined or separate if they transport waste- and stormwater through the same or different pipelines. Finally, wastewater is treated at the WWTP and discharged into the environment in compliance with the outflow quality standards that are applicable to the discharge area. Alternatively, reclaimed wastewater can be used for non-potable purposes, which also applies in the case of rainwater.

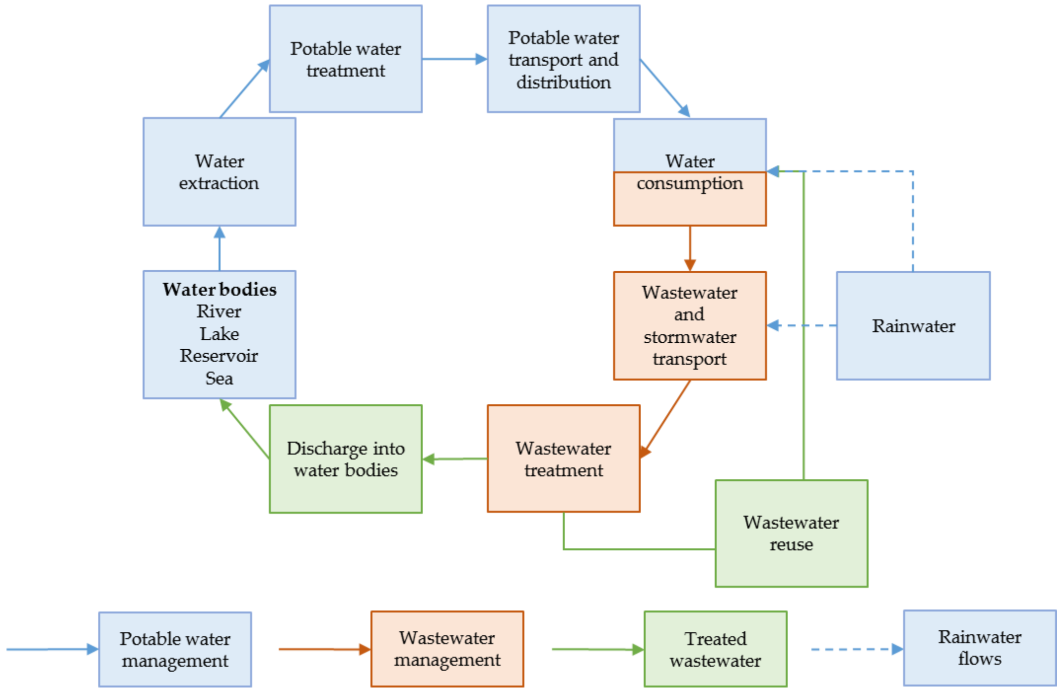


Figure 1.2 Diagram of the stages of the urban water cycle. The colors refer to the type of water that is managed in each step

Although the urban water cycle is a basic pillar in any city, community or settlement, access to water and sanitation is still a concern in the 21st century. The urban exodus might not always imply an improved water supply and sanitation system. In the period from 1990 to 2008, urban population increased by 1,099 million people, and 96% and 74% of these new urban residents gained access to improved drinking water and sanitation, respectively (WHO/UNICEF 2012). These figures are smaller in rural areas, with only 47% of the population having access to these systems worldwide (WHO/UNICEF 2012). This situation is also relevant in developing countries, especially in sub-Saharan Africa and Southern Asia, where less than 50% of the population has access to improved sanitation facilities (WHO/UNICEF 2015).

The European context is also very diverse (**Table 1.1**). The extreme difference among countries in terms of sanitation services is noteworthy. Central European countries have the highest sanitation rate, with 94% of the population being connected to WWTPs that offer a tertiary wastewater treatment in 77% of the cases (Eurostat 2015). In contrast, West Balkan and South Eastern areas have up to 43% of the population served by sewers that do not transport wastewater to a WWTP. This is a concern when dealing with severe eutrophication and aquifer pollution events that arise due to the discharge of untreated wastewater into water bodies. In this sense, 2,160 tons of nitrogen were released every day in 2013 in the European Union (28 countries). This is not irrelevant, especially depending on the discharge location. Until recently, several coastal locations released untreated sewage into the sea, for instance in Malta (WSC 2011) or Spain (Vida and Arroyo 2005).

In terms of global environmental impacts, European water and wastewater facilities account for less than 2% of the CO_2 eq emissions of the economy (Eurostat 2015). At the global level, the European Commission (2009) estimated that WWTPs generate 9% and 3% of the methane (CH_4) and nitrous oxide (N_2O) emissions worldwide. However, depending on the calculation methods and assumptions, the relative importance of water and wastewater systems can be higher or lower, as demonstrated in the Third Catalan Climate Change Report (Institut d'Estudis Catalans and Generalitat de Catalunya 2016). This distortion in the results poses a question: what should be included in the sustainability assessments of the urban water cycle, and what are the challenges that might alter this outcome? In the following section, these additional challenges are described with the objective of identifying potential areas of interest in sustainability assessments.

Table 1.1 Global European water and wastewater management data based on Eurostat (2015)

Parameter	Values for European Union (EU-28)
Gross surface and groundwater abstraction (million m^3) (2013)	30,306
Volumetric wastewater generation and discharge (million m^3) (2013)	
Point sources	7,750
Urban wastewater treatment	3,570
Discharges of WWTPs	4,340
Discharges to inland water bodies	7,680
Discharges to the sea	340
Point source pollutant emissions (tons/day) (2013)	
Total nitrogen	2,160
Total phosphorus	284
Available sanitation services (%) (2012)	
Population with wastewater treatment	Range: 8 - 94%
Population with wastewater collection, but without treatment	Range: 0.2 - 43%
GHG emissions (2014)	
Water supply, sewerage and waste management (million tons CO_2 eq)	45.4
Contribution to sectorial emissions	2%

1.2.2 *Sustainability challenges of current water management models*

Assessing the sustainability of the urban water cycle is complex due to the number of variables involved in its performance. This dissertation focuses on the metabolism of water in cities, specifically those elements related to the sustainability of sanitation and drainage. To this end, **Figure 1.3** screens potential challenges of sanitation and drainage systems from a causal perspective. This chain is an adaptation of the DPSIR (Driving forces, Pressure, State, Impact, Response) framework, which is used in the definition of strategies towards integrated environmental assessments (EEA 1999). In this case, we seek to define the relationship between driving forces, pressure on water resources, responses to these pressures and final impacts.

In the context of cities and sustainability, two potential driving forces can be identified, i.e., the increase in urban population and the variations in precipitation patterns that result from climate change. As stated in **Section 1.2.1**, cities are population hotspots, but the availability of sanitation systems is not homogenous. Combined with the sanitation requirements of future urban residents, the pressure on water infrastructure becomes apparent. The demand for sewer pipelines and wastewater treatment increases, adding up to the existing pressure on the system. It is difficult to estimate the total length of sewer that has already been constructed in the world, but it is a fact that millions of kilometers of aging sewers call for an urgent upgrade (de Feo et al. 2014; WWAP 2012). In response to this demand, urban sanitation systems should be renewed and adapted based on sustainability criteria to provide this service at the minimum costs.

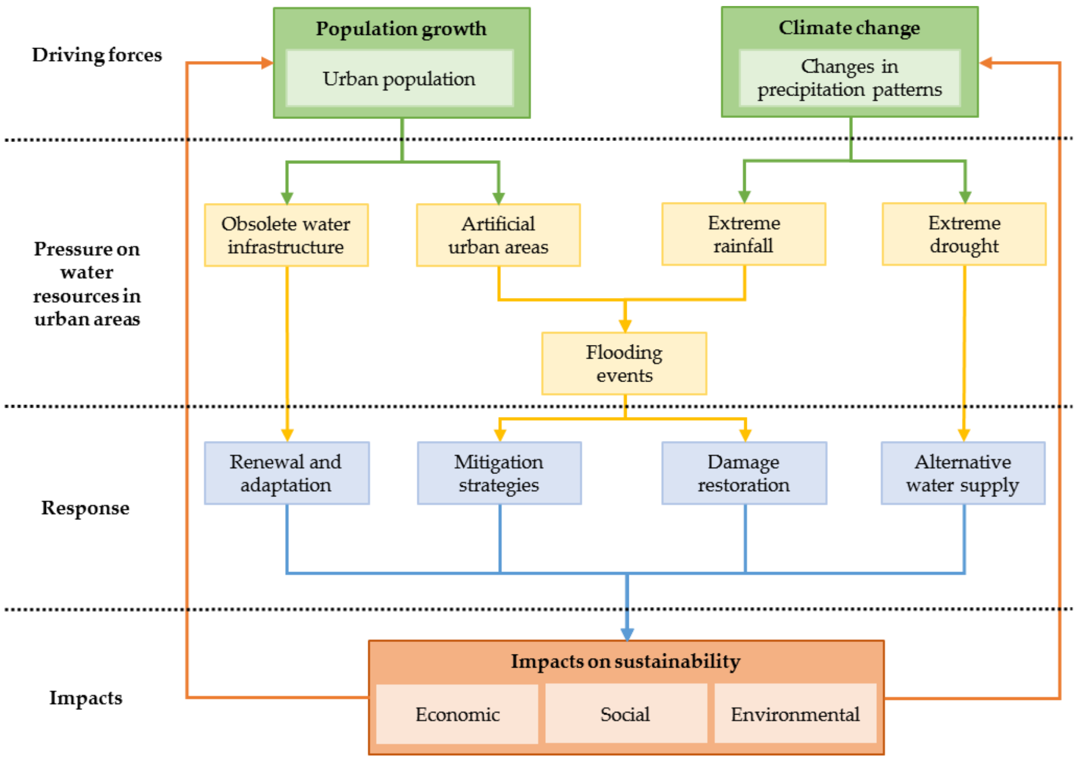


Figure 1.3 Sustainability challenges of sanitation and drainage systems from a causal perspective

At the same time, climate change poses a threat by increasing the intensity and frequency of both flooding and drought events. Wet regions might experience an increase in the average rainfall, whereas dry areas, including some parts of the subtropics, might become drier (Kirtman et al. 2013). These anomalies become even more relevant in urban areas, where artificial surfaces reduce the natural infiltration potential of the soil. Direct consequences are the reduction in the aquifer recharge and groundwater availability, as well as an increase in the urban runoff that most probably results in urban floods (Butler and Davies 2000). In fact, the historical evolution of flooding events and associated damage presented an incremental trend in the past decades (**Figure 1.4**). This highlights the importance of future actions and planning aimed at reducing the risk and sustainability impacts of mitigating and restoring flood damage.

The responses to these pressures ultimately lead to impacts on water sustainability. The extent to which responses affect the environment depends on the type of solutions applied in each case. Improving and/or increasing the access to water and sanitation systems can be done by overexploiting the existing infrastructure or implementing novel strategies that reduce the environmental and economic costs. These strategies can be adapted to the specific features of the area, such as population density or climate. **Section 1.2.3** explores the benefits and applications of different types of systems for sanitation and drainage. These should account for both the risk and damage mitigation potential to obtain a detailed overview of the sustainability aspects affected by management decisions.

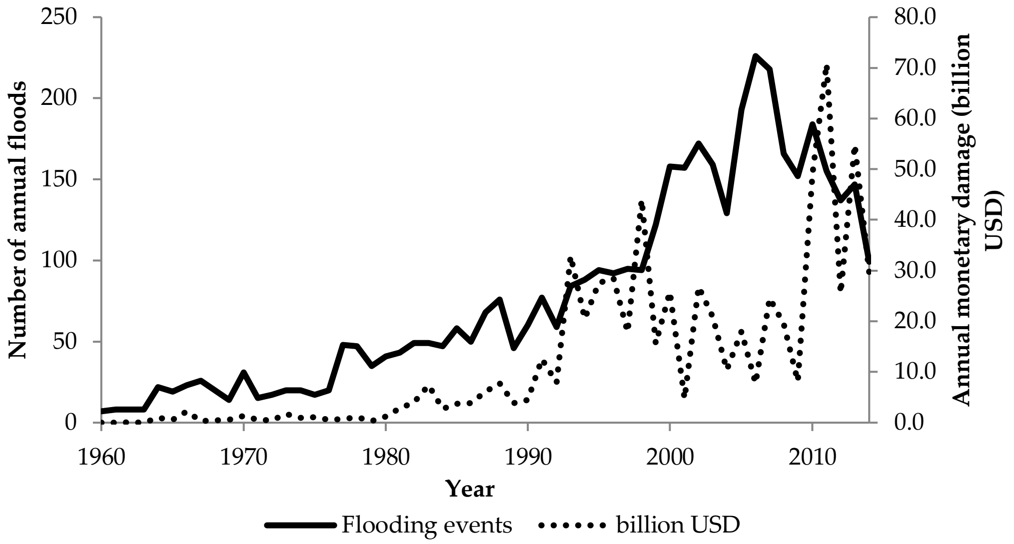


Figure 1.4 Evolution of flooding events and associated monetary damage from 1960 to 2014. Based on Guha-Sapir et al. (2009)

Figure 1.3 also illustrates a closed loop, meaning that the impacts on sustainability that result from local and regional actions also have an effect on climate change patterns and population wellbeing. To improve the overall performance of this cycle, the following section presents a comparison between conventional and alternative sanitation and drainage systems applied at the urban scale.

1.2.3 Conventional and alternative sanitation and drainage systems

Based on the urban water cycle presented in **Section 1.2.1**, different strategies can be implemented to deal with both the sanitation demand and risk prevention. In this sense, conventional and alternative systems provide different types of configurations and potential effects on urban environments (**Figure 1.5**). Deciding on one type or the other might compromise stormwater management, as the rationale behind urban drainage is more than the mere transport of water from one place to another through a sewer system (Butler and Davies 2000).

Most urban areas rely on traditional drainage systems that usually connect to a centralized WWTP. The transport of combined waste- and stormwater flows presents some limitations. On the one hand, pipes have a larger diameter because the water flows are larger than in separate sewers. This oversizing results in a reduced self-cleaning potential, greater resource consumption, and unnecessary construction costs (Bizier 2007). On the other hand, intense rainfall events might generate combined sewer overflows (CSO) when the flow capacity is exceeded. CSO can account for 50% of the total pollution registered in receiving waters due to the discharge of untreated wastewater (Malgrat et al. 2004). In addition, the limited stormwater retention capacity increases the flooding risk in the area. In this sense, adapting and renovating the existing networks, and restoring the damage generated by insufficient infrastructure implies direct economic and environmental costs that could be avoided.

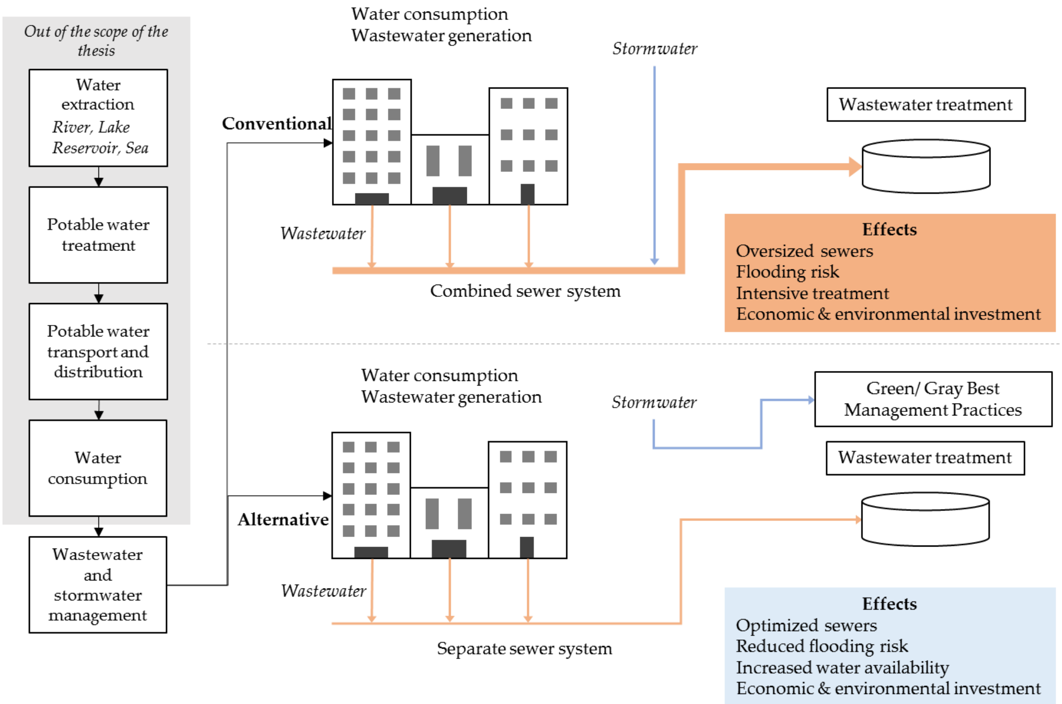
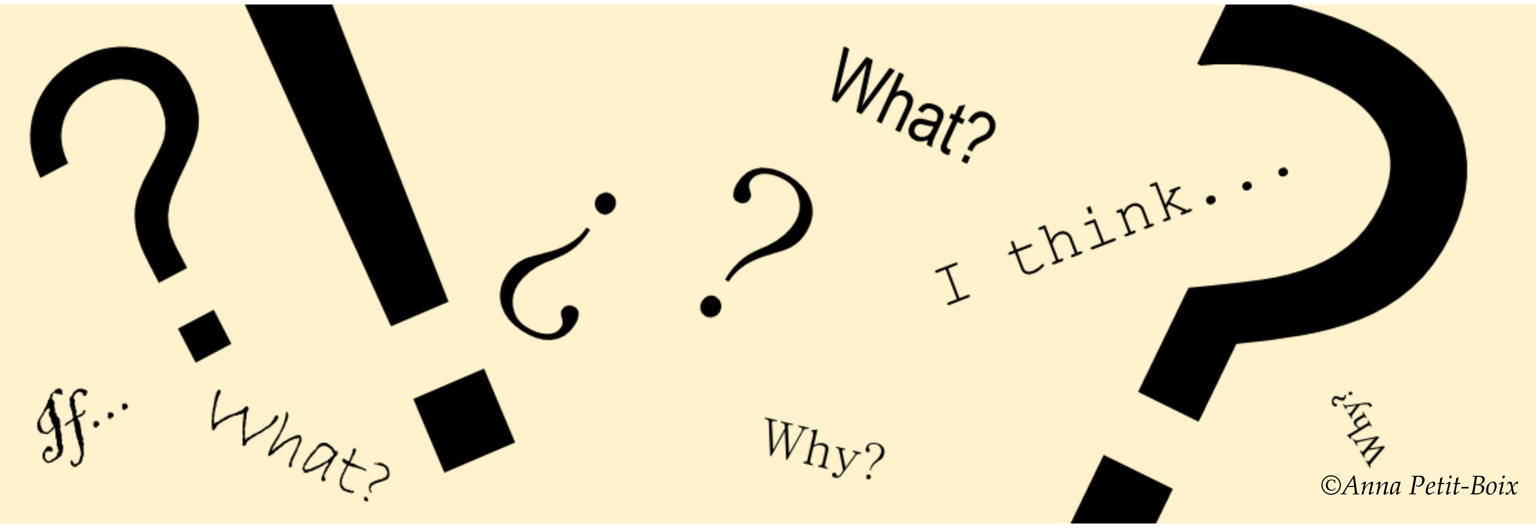


Figure 1.5 Simplified diagram of conventional and alternative sanitation and drainage systems

An alternative is the integration of stormwater Best Management Practices (BMP) into current sanitation and drainage plans. These are defined as "methods that have been determined to be the most effective, practical means of preventing or reducing pollution from nonpoint source" (Lee 2005, based on EPA-97/12). Sewers and detention tanks belong to the traditional ("gray") BMPs, but separate sewers can be a valid solution for preventing CSOs (Butler and Davies 2000; Semadeni-Davies et al. 2008) and re-using stormwater for non-potable purposes, such as irrigation. Green infrastructure is the second BMP category, which includes a variety of multifunctional integrated systems that are meant to provide additional ecosystem services to the environment and society (European Commission 2013). These strategies can also involve the construction of optimized sewers that are adapted to the wastewater production rates. At the same time, they deal with stormwater through naturalized systems that foster its retention and/or infiltration through vegetation, such as swales or permeable pavements, or through a direct re-use, such as rainwater harvesting systems. The application of these alternatives is attractive in water-stressed areas, either dry or wet, because they provide additional water sources while reducing undesired urban runoff (USEPA 2013b). For all these reasons, determining the potential benefits and impacts of these systems is worth analyzing in the context of climate change adaptation and mitigation.

Chapter 2



Picture: Questioning ourselves

Motivations, research questions and objectives

Chapter 2 Motivations, research questions and objectives

Based on the background information presented in **Chapter 1**, this chapter connects the main ideas on IE tools and the water cycle that inspired the elaboration of this dissertation. The resulting research questions and objectives are also presented.

2.1 Motivations of this dissertation

This dissertation builds upon the existing IE concepts and methods and their application in the field of sanitation and drainage systems (**Figure 2.1**). In general, it seeks to understand the life cycle of this infrastructure in order to provide decision-makers with environmental and economic results that help them manage the water cycle from an integrated perspective. In addition, the impacts and benefits quantified through this approach might feed the IE strategies applied so far. The social dimension is also essential, but due to its complexity, it was out of the scope of this research.

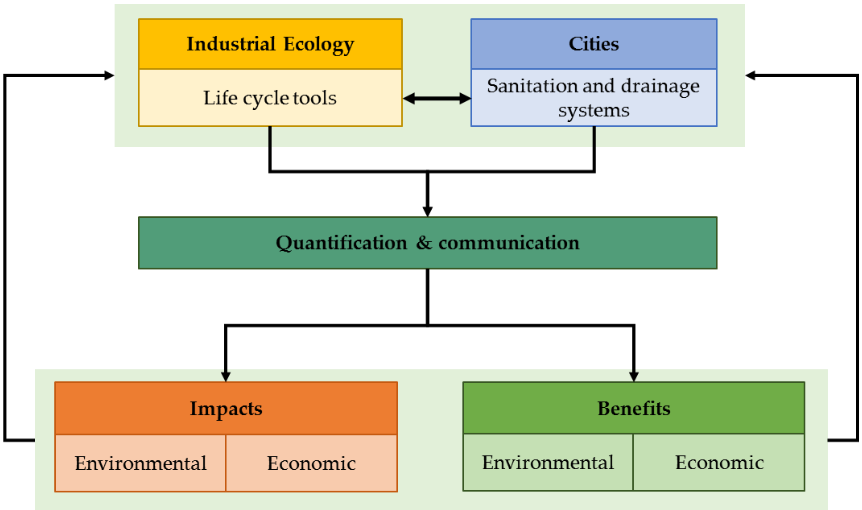


Figure 2.1 Motivations of this dissertation

Given the water and sustainability goals defined at a global context (UN 2016), a number of issues need to be dealt with when quantifying and assessing the environmental and economic performance of sanitation and drainage systems. Based on the management challenges presented in **Section 1.2.2**, we divided the motivations of this dissertation into two main groups, i.e., sewers and flood prevention.

Adaptation of existing and future sewer networks

Sewer networks must be adapted and/or renewed due to aging and wastewater demand, and they must be properly designed according to durability and water flows. However, the environmental impacts and economic costs that result from this process should be assessed in order not to compromise sustainability. In this sense, analyzing the entire life cycle of this infrastructure might be the key to understanding how sewers contribute to the environmental and economic performance of an urban area.

To do so, we need to identify all of the materials and processes involved in the construction and operation of sewers. So far, a variety of authors have studied the environmental impacts of sewers (Morera et al. 2016; Vahidi et al. 2015, 2016; Venkatesh et al. 2009; Risch et al. 2015) and a few have provided economic data (Akhtar et al. 2014; Thomas et al. 2016; Murla et al. 2016). However, integrating these results based on both general and specific case studies is essential to detecting the main factors that affect sewer performance. Site-specific features, such as topography or location (Roux et al. 2011), might influence the way sewers demand energy and materials, but these might be common in different cities. In this sense, we should identify where, how and why sewers have a different or similar performance.

In addition, if WWTPs contribute to the direct CH_4 and N_2O emissions (European Commission 2009), sewers might also play a similar role, as the wastewater degradation process starts in sewers. This is apparent because of the typical odor and corrosion issues that occur in the networks (USEPA 1974; Guisasola et al. 2008; Sharma et al. 2008; Vollertsen et al. 2008). For this reason, we need to quantify the direct GHG emissions generated in sewers and include them in the environmental impacts of this system. This process is not straightforward, but a first simple attempt might be helpful.

Furthermore, the way these results are communicated to users and decision-makers might influence their level of understanding and, thus, the level of success in transferring scientific knowledge to the society. Eco-efficiency assessments, which combine the environmental and economic dimensions of a product system, are a field to explore because they were quite recently regulated through ISO 14045:2012. Applying this method to the case of sewers might be helpful to determine the relationship between environmental impacts and economic costs.

Finally, some studies suggested that there is a strong relationship between the distance to urban centers and adverse environmental effects (McDonnell and Pickett 1990). However, Alberti (2010) concluded that we do not have enough information about the effects of different urban patterns, housing options and types of infrastructure (i.e., conventional and/or alternative) on ecological systems, to cite a few examples. For this reason, we need to study different types of urban/suburban areas to determine the best sanitation strategies (e.g., centralized or decentralized) from an environmental standpoint. This will provide information about the need to implement additional sewer pipelines or to prioritize alternative decentralized treatment systems.

Climate change mitigation and adaptation to intense rainfall events

The increasing frequency and intensity of rainfall and flooding events calls for the implementation of flood prevention systems that help cities adapt to climate change. However, not all the solutions are equally valid from an environmental and economic perspective. *A priori,* gray BMPs seem to perform environmentally worse than green alternatives because of the role of vegetation in carbon sequestration (De Sousa et al. 2012). So far, a number of studies have assessed the environmental impacts of different types of

BMPs, such as green roofs, bio-retention basins or rainwater harvesting systems (Kosareo and Ries 2007; Angrill et al. 2016; Flynn and Traver 2013), to cite a few.

Nevertheless, accounting for the benefits of these systems is associated with a complex modeling background. In the context of urban planning, local authorities might be interested in preventive actions that reduce the flooding risk and its associated urban damage. For this reason, cost-benefit analyses are typically conducted to determine the feasibility of these systems (Zhou et al. 2012; Jonkman et al. 2008). However, an environmental equivalent is needed to identify the feasibility of prevention systems in terms of damage prevention. So far, there is a lack of this type of analysis, but it might add up to the current IE methods by providing an integrated approach that covers some of the consequences of flooding.

This dissertation aims to meet this demand for novel approaches in the field of sanitation and drainage in an attempt to provide useful information to decision-makers and improve the eco-efficiency of current urban models.

2.2 Research questions and objectives

The goal of this dissertation is to assess the eco-efficiency of sanitation and drainage systems to determine the best alternatives in an urban context. To this end, we formulated the following research questions:

Question 1	What are the environmental and economic hotspots of sewer networks and the parameters that affect this outcome in medium-sized cities?
Question 2	Is decentralized sanitation a better environmental option in small suburban areas when dealing with peak sanitation demand?
Question 3	Do flood prevention strategies result in a positive environmental and economic performance when avoided damage is accounted for? If so, are green BMPs better than gray systems?

To answer these questions, different objectives were defined accordingly:

	Objective	Research question	Chapter
I	To determine the environmental impacts of the construction of sewers in urban areas from a life cycle approach		**4 & 5**
II	To study the variables that influence the energy consumption in the operation of sewers	**Q1**	**6**
III	To identify potential direct GHG emissions in sewers associated with wastewater degradation		**7**
IV	To assess the eco-efficiency of sewers in cities with contrasting urban and climatic features		**8**
V	To determine the environmental feasibility of centralized and decentralized sanitation scenarios in the context of suburban temporary settlements	**Q2**	**9**
VI	To provide a novel method for quantifying the eco-efficiency performance of flood prevention strategies	**Q3**	**10 & 12**
VII	To estimate the environmental and economic impacts of stormwater BMPs through life cycle tools		**10-12**

Chapter 3

Picture: Tools at grandad's studio

Methodology

Chapter 3 Methodology

This chapter describes the set of methods applied in the dissertation and the areas that were selected to exemplify the results in real settings. General and detailed data are provided for each of the tools.

3.1 Method overview

A number of methods were used in the course of this dissertation to deal with the quantification of the environmental and economic performance of sanitation and drainage systems (**Figure 3.1**).

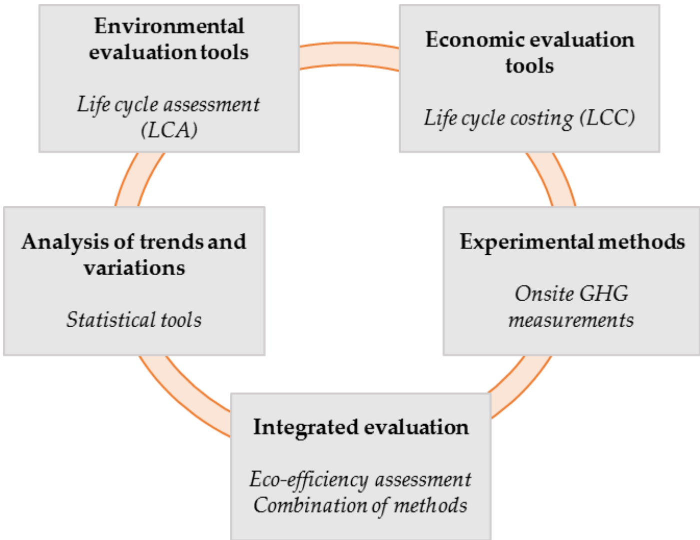


Figure 3.1 Overview of the methods applied in the dissertation

In the context of IE methods, life cycle assessment (LCA), life cycle costing (LCC) and eco-efficient assessment were applied to determine the environmental and economic investment in this infrastructure. However, these were also integrated into novel approaches to optimize the material use or identify potential benefits at different scales, among others. Experimental data were also needed to obtain real values on direct GHG emissions generated in sewers. In order to identify trends and variations in the impacts of sanitation and drainage systems in different urban areas, statistical tools were applied.

The extent to which each method was used is shown in **Table 3.1**. All of the chapters include environmental indicators, such as CO_2 emissions, with the aim of offering an environmental perspective in all cases. Chapters 8, 10 and 11 provide additional economic information on sewers and flood prevention strategies. Integrated evaluations depend on the analysis. In Chapter 5, LCA is combined with structural parametrization to define equivalent sewer designs based on regulations and select the ones with the lowest environmental impacts. Chapters 8 and 10 make use of the eco-efficiency approach, whereas Chapter 12 integrates a new dimension to LCA studies by expanding the system boundaries of prevention systems to include urban damage. Statistical tools were used in Chapter 6 to identify potential variations in the energy demand of sewers depending on their location. Chapter 9 includes a Monte Carlo simulation to predict the variability and sensitivity of the environmental results of (de)centralized sanitation to a set of key variables. Onsite experimental methods were needed in Chapter 7 to estimate the direct GHG emissions generated in sewers. These methods are described in the following sections.

Table 3.1 Methods applied in each part and chapter of the dissertation

Sections		Environmental tools	Economic tools	Integrated evaluation	Statistical tools	Experimental methods
II	Chapter 4	•				
	Chapter 5	•		•		
III	Chapter 6	•			•	
	Chapter 7	•				•
IV	Chapter 8	•	•	•		
	Chapter 9	•			•	
V	Chapter 10	•	•	•		
	Chapter 11	•	•			
	Chapter 12	•		•		

3.2 Environmental tools: life cycle assessment

Environmental data are present in all of the studies included in this dissertation, which were obtained through different procedures. However, life cycle assessment (LCA) is the main environmental quantification tool applied in most of the analyses. The methodological framework is described in the following sections along with relevant features of the LCAs performed during this research.

3.2.1 Theoretical framework

As defined by ISO 14040:2006 - 14044:2006, LCA is "a compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle". To obtain these environmental impacts, four iterative phases are involved (**Figure 3.2**), which are described in this section based on the ISO standards.

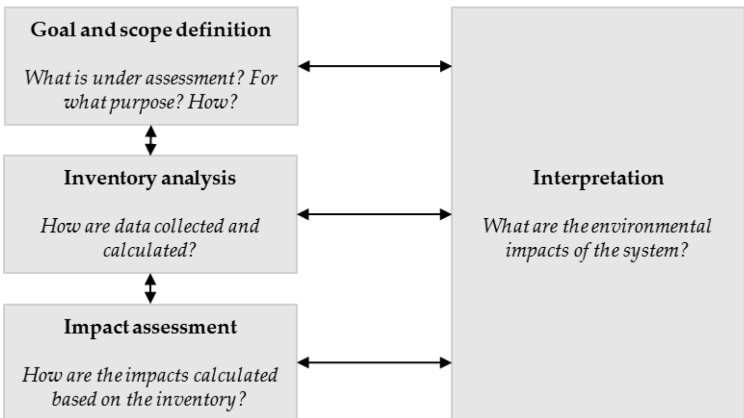


Figure 3.2 Phases of the LCA method. Based on ISO 14040:2006

a) Goal and scope definition

In the first phase, the system under analysis is defined based on a specific purpose. The goal should explicitly indicate the applications and motivations of the analysis, target audience, and intended comparative reports. The scope defines the system that is being studied and a set of elements that will ultimately affect the environmental results:

- The function, functional unit and reference flows
- The system boundaries, allocation processes and assumptions
- The required data quality and quantity
- The impact assessment methods and categories applied
- The assumptions, limitations and critical aspects of the analysis

Quantifying through LCA implies that a product system is associated with a certain function. For this reason, input and output data always refer to a measurable functional unit (FU) that quantifies and qualifies a system's function through the questions "what", "how much", "how well", and "for how long" (EC-JRC 2010a).

The system boundaries of the analysis are central to defining the processes included within the LCA. The life cycle stages, processes and flows included in and excluded from the assessment must be clearly stated so that the results are interpreted and compared accordingly. Quantitative cut-off criteria should be defined, when needed. The system boundaries can be from cradle to gate, from cradle to grave or from cradle to cradle, among other options. In addition, when different co-products or functions result from the same system, assigning environmental burdens to each of them follows a hierarchical decision-making process according to ISO 14044:2006:

(1) Allocation should be avoided by
    (a) dividing the unit process into more sub-processes for which data can be collected, or
    (b) expanding the system to include additional functions of the co-products
(2) When step (1) is not possible and allocation cannot be avoided, each of the co-products or functions should be assigned part of the inputs and outputs of the system based on physical relationships (e.g., mass).
(3) When physical relationships cannot be obtained, other ways of allocation should be applied, such as economic allocation.

b) Inventory analysis

The creation of the life cycle inventory (LCI) is the second phase of the LCA method where data on the inputs and outputs is collected based on mass balance principles. These include:

- Inputs from nature, such as water or minerals
- Inputs from technosphere, i.e., processed items such as electricity or manufactured materials
- Outputs to technosphere, such as by-products
- Outputs to nature, for example emissions to the air, water or soil.

These data can be obtained from different types of sources. Primary or foreground data result from fieldwork, lab experiments or data registered in the system under analysis. Secondary or background data can be retrieved from reports, databases or previous literature. An example of background data are LCI databases, such as the ecoinvent database (Weidema et al. 2013), which provides key data on the life cycle of materials and processes.

There are two main LCI modeling principles, i.e., attributional (ALCA) or consequential (CLCA) (EC-JRC 2010a). The ALCA approach, also known as "accounting" or "descriptive", was defined as a "system modelling approach in which inputs and outputs are attributed to the functional unit of a product system by linking and/or partitioning the unit processes of the system according to a normative rule" (Sonnemann and Vigon 2011). In contrast, CLCA is a change-oriented approach "in which activities in a product system are linked so that activities are included in the product system to the extent that they are expected to change as a consequence of a change in demand for the functional unit" (Sonnemann and Vigon 2011). In other words: Whereas ALCA describes the system as it is or was, CLCA integrates market mechanisms and looks at the consequences of decisions, which might happen outside of its immediate system boundaries (Weidema et al. 2009; Earles and Halog 2011; EC-JRC 2010a).

c) Impact assessment

Using the LCI, the life cycle impact assessment (LCIA) phase calculates the potential environmental impacts of the system through a set of impact categories. To do so, ISO 14040:2006 defines three mandatory steps, i.e., (1) selection of impact assessment models and indicators, (2) classification of the LCI into category indicators, and (3) characterization of each item through the impact assessment models to obtain the LCIA results.

When selecting impact categories, the type of information estimated and communicated through each indicator must be considered. In this sense, LCIA methods link the LCI with areas of protection (AoP) (i.e., at the endpoint level) or somewhere along the impact pathway (i.e., at the midpoint level) (EC-JRC 2010b). Midpoint-oriented methods are at an intermediate position of the impact pathways where a common mechanism exists for a group of substances (Jolliet et al. 2004; EC-JRC 2010b). An example is the contribution of CO_2 or CH_4 emissions to global warming due to radiative forcing. Characterization at the endpoint level involves indicators that can be more easily interpreted because they are closer to societal concerns (Hertwich and Hammitt 2001), although the modeling principles and choices are more complex and result in more uncertainty (Bare et al. 2000). The AoP include the impacts to human health, natural resources and ecosystems. Midpoint and/or endpoint indicators can be optionally normalized, weighted or grouped in order to provide a single value to the target audience and reduce the complexity of the interpretation process (Bare et al. 2000). This process should not only include scientific choices, but also social and economic preferences.

d) Interpretation

In the last phase, the environmental results are interpreted to identify hotspots and improvement actions based on the selected set of categories. Iterations can be conducted to improve the LCI, modeling methods or objectives. Here, data consistency, completeness and sensitivity are evaluated (EC-JRC 2010a). Finally, the study provides conclusions, recommendations and limitations, and can be compared to other systems if needed.

3.2.2 LCA applications in this research

Seven of the chapters of this dissertation explicitly use LCA as the main environmental assessment method. Different methodological aspects were considered depending on the goal of the analysis. The particular specifications are explained in each chapter.

a) Goal and scope definition

A set of FUs was defined in the different research topics under assessment. The most common FU was one cubic meter (m^3) of wastewater or stormwater collected, transported and/or treated, depending on the specific goals of the study. In the case of sewer infrastructure, an additional concept was used to simplify the calculation process, i.e., declared units (DU). According to the environmental product declaration (EPD) rules for construction products (EN 15804:2011), a DU is defined when the precise function of a product is not known, usually due to the multiple applications of basic products, such as beams or other concrete structures. In this sense, the DUs used in this dissertation were one linear meter of pipeline and one unit of sewer appurtenance (i.e., a sewer's supporting elements, such as pumps or manholes) (see Chapters 4 and 5). More complex FUs were used in Chapters 10 and 12 when dealing with the balance of flood and damage prevention. The system boundaries of the LCA studies were generally from cradle to grave, except for sewer infrastructure, where the end of life was not included. No allocation procedures were applied in any of the analyses.

b) Inventory analysis

Water companies were the main suppliers of primary data, such as wastewater consumption or features of the sanitation systems. Sewer studies implied an intensive relationship with water managers especially through the LIFE+ Aquaenvec project (LIFE10/ES/ENV/520), which involved local water agencies that provided raw data on municipal systems. In addition, stormwater prevention systems were assessed using real data obtained via the Brazilian "Smart Parks" project (HBP-2012-0216, CAPES ref. 5206) (Chapters 11 and 12) and the archive of the Catalan Water Agency (Chapter 10). The Spanish Insurance Compensation Consortium also provided real flood damage estimates that were used in Chapter 10. For this reason, primary data is believed to have a good quality.

Two main types of background data were applied. On the one hand, construction-related datasets were retrieved from construction databases, particularly MetaBase ITeC (2010). This database offers open data on the material, energy and economic requirements of a wide set of constructive solutions that were needed in the LCA studies. On the other hand, background LCI unit processes were obtained from ecoinvent v2.2 (Frischknecht et al. 2005) and ecoinvent v3 (Weidema et al. 2013).

The general modeling principle applied in the LCA studies was the ALCA approach. However, Chapter 12 attempts to assess flood prevention systems from a CLCA perspective by integrating avoided flood damage as one of the consequences of planning decisions. However, it is not a pure CLCA because it focuses on a simplified approach and not all of the processes involved were studied based on market mechanisms.

c) Impact assessment

Different LCIA methods and approaches were selected based on the purpose of the assessment and suitability (**Table 3.2**). In all of the cases, the Simapro software (PRé Consultants 2010, 2014) was used in the modeling of the environmental impacts. The first chapters (4 to 7) use the CML IA method (Guinée et al. 2002), but the ReCiPe method (Goedkoop et al. 2009) was applied from Chapters 8 to 12 in order to use the most updated models. These two methods include similar indicators, but ReCiPe covers a wider range of impact areas. Note that the global warming indicator is called climate change in ReCiPe, but the same name was used for the sake of homogenization. The latter was a common indicator in all chapters because it is a well-known category and it is easy to communicate. The presentation rules are common in all chapters as recommended by Heijungs (2013).

Table 3.2 LCIA methods and impact indicators used in each chapter of the dissertation

LCIA methods and indicators	**Chapters**								
	4	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**
CML IA method (Guinée et al. 2002)									
Abiotic depletion	•	•							
Acidification	•	•							
Eutrophication	•	•							
Global warming	•	•	•	•					
Ozone depletion	•	•							
Human toxicity	•	•							
Photochemical ozone creation	•	•							
ReCiPe (H) (Goedkoop et al. 2009)									
Midpoint									
Global warming					•	•	•	•	•
Ozone depletion					•	•	•	•	•
Terrestrial acidification					•	•	•	•	•
Freshwater eutrophication					•	•	•	•	•
Marine eutrophication					•	•	•	•	•
Human toxicity					•	•	•	•	•
Photochemical oxidant formation					•	•	•	•	•
Particulate matter formation					•	•			
Terrestrial ecotoxicity					•	•			
Freshwater ecotoxicity					•	•			
Marine ecotoxicity					•	•			
Ionizing radiation					•	•			
Agricultural land occupation					•	•			
Urban land occupation					•	•			
Natural land transformation					•	•			
Water depletion					•	•	•	•	•
Metal depletion					•	•	•	•	•
Fossil depletion					•	•	•	•	•
Endpoint									
Human health					•				
Ecosystems					•				
Resources					•				
Weighted endpoints									
Eco-points					•	•			
Cumulative energy demand (Hischier et al. 2010)	•	•			•	•	•	•	•

The CML IA indicators shown in **Table 3.2** were selected based on the reporting requirements stated by the EPD rules for construction products (EN 15804:2011). A similar rationale was applied in Chapters 10 to 12. In Chapter 8, which tries to simplify and discuss the communication issues, the complete set of midpoint and endpoint indicators was shown, as some results were represented using eco-points. These are (dimensionless) single scores that result from weighting the endpoints. For the sake of transparency, all of the indicators were presented. In Chapter 9, all of the midpoints and eco-points are also included, but special emphasis is placed on a group of indicators that are more interesting in the field of wastewater treatment. The impact categories are described in **Table 3.3**.

The ReCiPe method has three perspectives, i.e., Egalitarian, Hierarchist, and Individualist, each of them considering a different timeframe. Because the Hierarchist (H) view accounts for a timeframe of 100 years, it is commonly used, as it is based on common policy principles (Goedkoop et al. 2009). This was also the approach selected in the analyses that used the ReCiPe method.

Table 3.3 Description of the impact categories used in the studies based on their following baseline reports

Impact categories	Acronym	Units	Definition
CML IA method. Based on Guinée et al. (2002)			
Abiotic depletion	AD	kg Sb eq	Extraction of mineral and energy resources based on the ultimate reserves and extraction rates
Acidification	A	kg SO_2 eq	Emission of acidifying pollutants that have an impact on soil, groundwater, etc.
Eutrophication	E	kg PO_4^{3-} eq	Emission of macronutrients (here phosphorus) that increase the biomass production in aquatic and terrestrial ecosystems.
Global warming	GW	kg CO_2 eq	Contribution of GHG emissions the radiative forcing of the atmosphere, which increases the earth's temperature
Ozone depletion	OD	kg CFC-11 eq	Thinning of the ozone layer that increases the ultraviolet radiation
Human toxicity	HT	kg 1.4-DB eq	Emission of toxic substances to the environment that are health detrimental.
Photochemical ozone creation	POC	kg C_2H_4 eq	Formation of tropospheric ozone due to the reaction of air pollutants with sunlight.
ReCiPe method. Based on Goedkoop et al. (2009)*			
Freshwater eutrophication	FE	kg P eq	Emission of phosphorus compounds that result in the phosphorus enrichment of freshwater and foster algae proliferation
Marine eutrophication	ME	kg N eq	Emission of nitrogen compounds that result in the nitrogen enrichment of seawater and foster algae proliferation

Impact categories	Acronym	Units	Definition
Particulate matter formation	PMF	kg PM10 eq	Emission of a mixture of organic and inorganic substances with a diameter <10μm (PM_{10}) that can reach the lungs when inhaled and ultimately cause health issues
Terrestrial ecotoxicity	TET	kg 1,4-DB eq	Persistence, accumulation and effect of chemicals released to urban air that can be later inhaled by humans
Freshwater ecotoxicity	FET	kg 1,4-DB eq	Emission of chemicals to rural air that have a potential fate and effect on freshwater bodies
Marine ecotoxicity	MET	kg 1,4-DB eq	Emission of chemicals that have a potential fate and effect on seawater
Ionizing radiation	IR	kBq U^{235} eq	Release of radioactive material to the environment
Agricultural land occupation	ALO	m^2a	Effects of occupying agricultural land during a specific period
Urban land occupation	ULO	m^2a	Effects of occupying urban land during a specific period
Natural land transformation	NLT	m^2	Effects of transforming natural systems into artificial areas
Water depletion	WD	m^3	Consumption of freshwater both as a scarce and abundant resource depending on the extraction area
Metal depletion	MD	kg Fe eq	Extraction of minerals and associated quality loss
Fossil depletion	FD	kg oil eq	Consumption of the group of resources that contain hydrocarbons due to anthropogenic activities
Human health	HH	DALY	Disability-adjusted loss of life years associated with different types of diseases
Ecosystems	ED	species.year	Decreased quality of an ecosystem associated with the loss of species during a year
Resources	RA	$	Increase in the market prices of basic resources that results from its depletion
Cumulative energy demand. Based on Hischier et al. (2010)			
Cumulative energy demand	CED	MJ	Primary energy use from renewable and non-renewable origin

* The categories defined in the CML IA section are not provided again for the ReCiPe method, as the meaning of the concepts is similar or the same. Note that the photochemical oxidant formation (POF) can be considered an equivalent to the photochemical ozone creation. This also applies in the case of terrestrial acidification (TA)

3.3 Economic tools: life cycle costing

Economic results are also provided in this dissertation and most of them were obtained through the life cycle costing (LCC) method. The LCC of buildings and construction assets was standardized through ISO 15686-5:2008, which was consulted due to the nature of this research. Based on the guidelines published by the UNEP-SETAC Life Cycle Initiative (Swarr et al. 2011), the phases for conducting an LCC are very similar to those included in the LCA method (**Section 3.2**), as illustrated in **Figure 3.3**. The main difference is the third phase, which is now called "aggregation" and consists of selecting the economic indicators of interest.

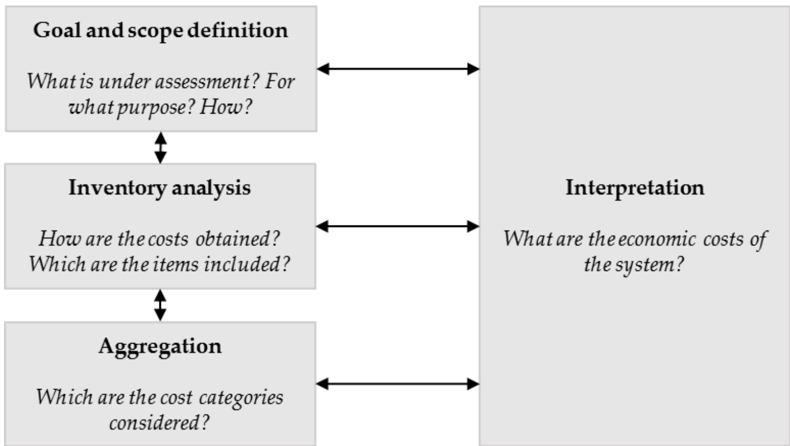


Figure 3.3 Phases of the LCC method. Based on Swarr et al. (2011)

Pure LCC studies are not part of this dissertation, as they are always combined with environmental LCA data to provide integrated results for decision-making. All of them were conventional LCCs that accounted for private life cycle costs (Swarr et al. 2011). Details on the specific processes are provided in each chapter.

3.4 Integrated evaluation: eco-efficiency assessment

Combining different types of tools is one of the main contributions of this dissertation. These integrated evaluations were conducted at different levels, i.e., adapting existing methods and coupling complementary disciplines. One of the relevant approaches is the eco-efficiency assessment. This method lacked a standardized framework until the publication of ISO 14045:2012, which is entirely devoted to normalizing the application of eco-efficiency assessments. According to this standard, eco-efficiency is an "aspect of sustainability relating the environmental performance of a product system to its product system value". For this reason, this tool was used when combining economic and environmental data. The phases of an eco-efficiency assessment are similar to those of an LCA (**Section 3.2**), but including two parallel environmental and value quantification processes that are finally combined (**Figure 3.4**). Because a pure eco-efficiency approach is applied in Chapter 8, a more detailed explanation of the specific assumptions and modeling principles is presented in its methodological section.

It is worth mentioning that the results presented in Chapter 8 belong to a set of values that were officially certified by AENOR (a Spanish certification agency) as the first Spanish analysis with an ISO 14045 award in the context of the LIFE+ Aquaenvec project (Aquaenvec 2015). The results of the assessment were arranged in an Excel file and a certified evaluator made a random selection of values. These were then traced back through equations, assumptions and raw data in order to justify the calculation process and prove their validity.

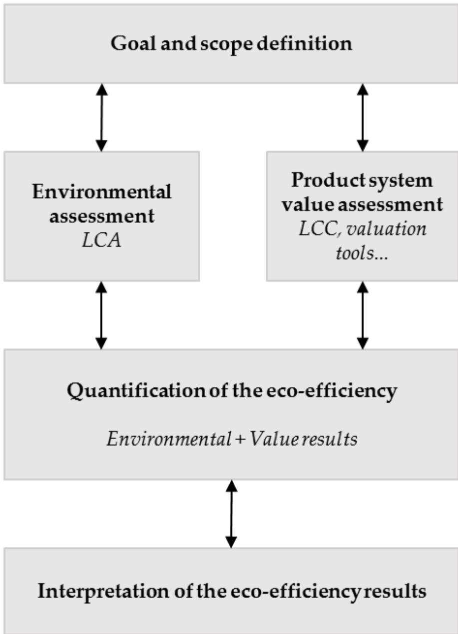


Figure 3.4 Phases of an eco-efficiency assessment. Based on ISO 14045:2012

3.5 Experimental and statistical tools

Experimental methods were only used in Chapter 7 in the detection of GHG emissions from real-life sewers. The particular specifications of this process are presented in the corresponding section of this chapter. Here, a list of analytical methods is provided so that the features of the gas analyzer and analytical procedure can be consulted (**Table 3.4** and **Table 3.5**). Personnel from a water company were responsible for setting up and managing the equipment at the sampling sites, and later providing measurement records. Wastewater samples were also collected at each site, but they were processed by one of the partners of the LIFE+ Aquaenvec project.

Statistical tools, such as correlation models and Monte Carlo simulations, were applied in Chapters 6 and 9 and their particularities are explained in these chapters, as these are not general methods that apply to all of the studies.

Table 3.4 Technical features of the gas analyzer and gas conditioner

Technical features of the sampling cell	
Principle:	Non-dispersive IR
Technology:	Infrared transduction
Range (CH_4):	0-500 ppm
Range (N_2O):	0-50 ppm
Error:	< 1% or 0.5 ppm_v
Linearity:	< 1% or 0.5 ppm_v
Repeatability:	< 1% or 0.5 ppm_v
Response time (T90):	<20 seconds

Technical features of the gas conditioner	
Gas flow:	100Nl/h
Inlet gas dew point:	65ºC
Inlet temperature:	140ºC max.
Outlet gas dew point:	3ºC
Environmental temperature:	5-45ºC
Cooling power:	160W
Condensate discharge capacity:	300 ml/min
Filter:	1μm
Rotameter: with needle valve	Range: 10-100Nl/h
Gas connections:	Inlet: DN6 SS316 Outlet: DN6 PP
Dimensions:	430 x 225 x 300 mm
Weight:	20kg approx.

Table 3.5 Analytical methods applied during the experimental process

Analytical methods	
COD	5220C Chemical Oxygen Demand. Closed reflux, Tritimetric Method, Standard Methods (APHA, 2005) modified by Soto et al. (1989)
TOC	Shimadzu analyzer (TOC-L)
Solids	2540B Solids. Total solids dried at 103-105ºC Standard Methods (APHA 2005) 2540D Solids. Total Suspended Solids dried at 103-105ºC Standard Methods (APHA 2005) 2540E Solids. Fixed and Volatile Solids Ignited at 550ºC Standard Methods (APHA 2005)
Anions	4110B Determination of anions by ion chromatography. Ion chromatography with Chemical Suppression of Eluent Conductivity. Standard Methods (APHA 2005) Equipment: Advanced Compact IC System (861, Metrohm)
Cations	Equipment: Advanced Compact IC System (861, Metrohm)

3.6 Case study selection

The methodological framework explained in **Sections 3.2 – 3.5** was applied to different infrastructures and at different scales (**Figure 3.5**). Wastewater management was studied in cities and suburban areas. In the case of cities, centralized sewers were the infrastructure of interest. However, because sewers might not be the best option in suburban areas, decentralized and centralized infrastructure were compared at this scale. Stormwater management was assessed at the city, neighborhood and building scales through the implementation of gray and green prevention systems.

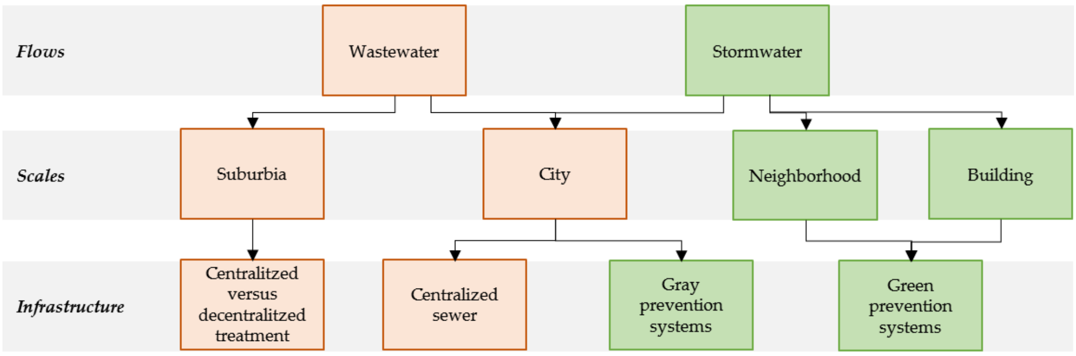


Figure 3.5 Flows, systems and infrastructure covered in this dissertation

Each analysis is based on particular case studies that are sensitive to water issues. **Figure 3.6** illustrates the location of these areas and the chapter where they are addressed. A detailed description of each case study area is provided in their respective chapters for a better understanding.

In general, most of the analyses were conducted in Spain, but covering different types of urban areas and climates for increasing the representativeness of the results. The eco-efficiency of sewers was studied in Calafell (Catalonia) and Betanzos (Galicia). These cities were selected in the context of the LIFE+ Aquaenvec project, as they were representative of the Mediterranean and Atlantic climate, respectively, and presented different water use patterns and urban form (Aquaenvec 2015). An environmental calculation method was tested on Betanzos' sewer (Chapter 4), an eco-efficiency assessment (i.e., LCA and LCC) of both cities was performed in Chapter 8, and Chapter 7 assessed the direct GHG emissions resulting from both sewers. To determine broader operational differences among cities, a sample was studied in the green-shaded areas shown in **Figure 3.6** (Chapter 6).

The analysis of centralized and decentralized sanitation systems was performed under particular conditions (Chapter 9). In this case, the study focused on tourist areas that have an increased sanitation demand in the summer but limited treatment options. To assess the centralization paradigm, the island of Minorca was selected because it has a number of suburban seasonal settlements that need to be re-designed in terms of sanitation in order not to compromise sustainability. Minorca is a Reserve of the Biosphere and, as such, preserving the environmental conditions of the region is essential.

Stormwater management was assessed in two regions that are affected by the flooding risk under different circumstances. On the one hand, the Maresme region (Catalonia) is a well-known area that is affected by intense flooding due to its coastal location and the presence of ephemeral streams that are always dry except for the occasional rainfall events. Chapter 10 deals with the investment in (gray) post-disaster emergency actions in this area and the resulting flood damage.

On the other hand, green BMPs were assessed in another flood-sensitive area, this time in São Carlos (Brazil). In the framework of the "Smart Parks" project (HBP-2012-0216, CAPES ref. 5206), the hydrologic, environmental and economic performance of a particular green BMP was studied, i.e., a filter, swale and infiltration trench. Chapter 11 includes an LCA and LCC of this system, together with its potential to reducing stormwater runoff. This study was conducted at the building scale, because this was the real-life setting of this infrastructure. Chapter 12 exemplifies the integration of damage prevention into the CLCA of BMPs through a theoretical analysis of a neighborhood in São Carlos. The FST was hypothetically located in available green spaces of the neighborhood to show how the method proposed in this chapter can be used in other cases.

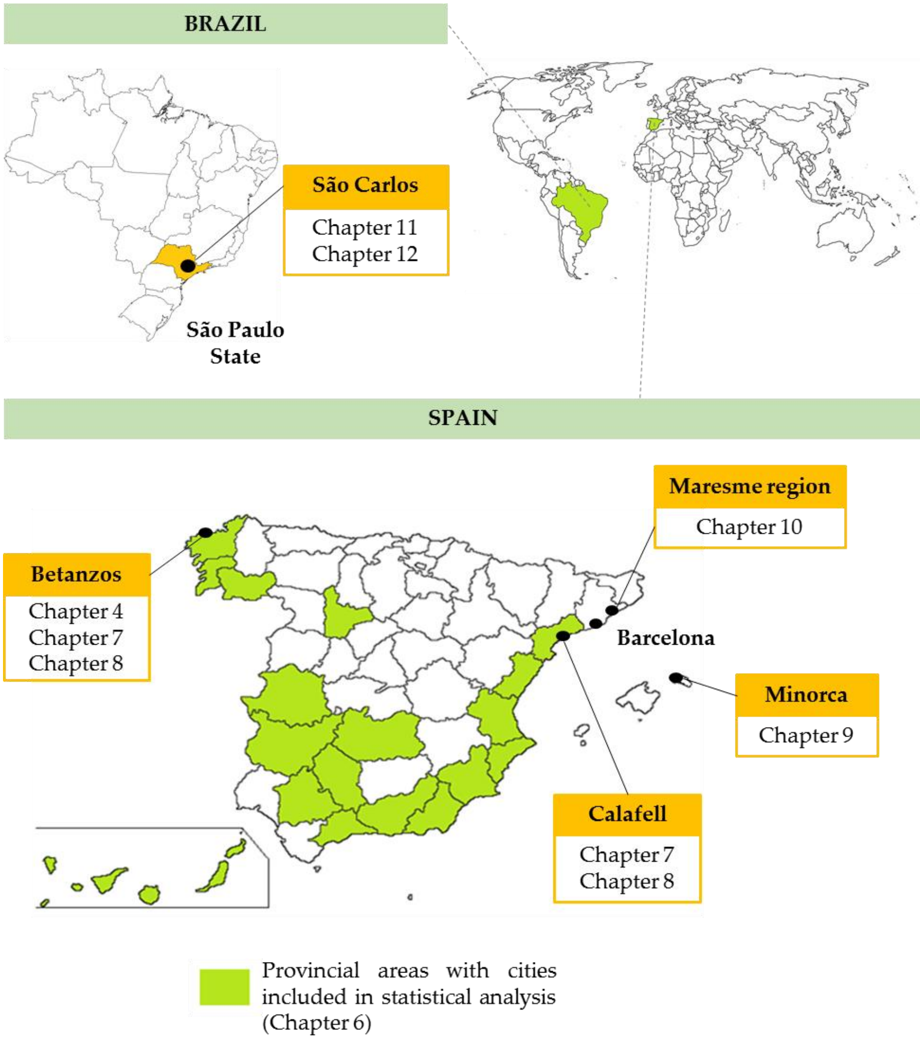


Figure 3.6 Geographic distribution of the case study areas

Part II

Sewer construction in urban areas – an environmental assessment

Chapter 4

Picture: Collection of manhole covers from around the globe

Environmental assessment of sewer construction in small to medium sized cities using life cycle assessment

Chapter 4 Environmental assessment of sewer construction in small to medium sized cities using life cycle assessment

This chapter is based on the following paper:

Petit-Boix, A., Sanjuan-Delmás, D., Gasol, C., Villalba, G., Suárez-Ojeda, M., Gabarrell, X., Josa, A., Rieradevall, J., 2014. Environmental assessment of sewer construction in small to medium sized cities using life cycle assessment. Water Resources Management 28, 979–997

doi: 10.1007/s11269-014-0528-z

DDD link: http://ddd.uab.cat/record/141547

Abstract

In a world with an increasing urban population, analyzing the construction impacts of sanitation infrastructures through life cycle assessment (LCA) is necessary for defining the best environmental management strategies. In this study, the environmental impacts of one linear meter of sewer constructive solution were analyzed for different pipe materials and diameters used in Southern Europe; a unit of different sewer appurtenances (pump, manhole and inspection chamber) was also considered. The impacts of the pipe materials were compared considering different lifespan periods and high-density polyethylene (HDPE) turned out to be the worst option, being polyvinyl chloride (PVC) and concrete the most favorable ones. Few data are available on the material and energy flows in the installation stage; therefore, a comparative analysis of trenches with sand and concrete bedding was conducted. The results show that the installation stage represents up to 80% of the total life cycle impact of the constructive solutions. Concrete pipes with half-concrete/half-sand bedding are the best option and produce 20-30% of the impact of HDPE pipes with concrete bedding. Hence, designers should focus not only on the pipe but also on the trench model. A methodology was presented to enable the impact aggregation of different sewer elements, and Betanzos (Spain) was selected to conduct a pilot study in small cities. In the future, studies will need to incorporate the operation and maintenance stage, as it is not standard and varies according to the physical features of the cities. Finally, this study provides basic concepts for developing eco-efficiency indicators.

Keywords: pipe, appurtenance, LCI, urban, construction, smart cities

Chapter 5

Picture: Construction of a sewer stretch in Basel (Switzerland)

Integrated structural analysis and life cycle assessment of equivalent trench-pipe systems for sewerage

Chapter 5 Integrated structural analysis and life cycle assessment of equivalent trench-pipe systems for sewerage

This chapter is based on the following paper:

Petit-Boix, A., Roigé, N., de la Fuente, A., Pujadas, P., Gabarrell, X., Rieradevall, J., Josa, A., 2016. Integrated structural analysis and life cycle assessment of equivalent trench-pipe systems for sewerage. Water Resources Management 30, 1117–1130.

doi: 10.1007/s11269-015-1214-5

DDD link: http://ddd.uab.cat/record/146023

Abstract

The demand for sanitation infrastructures is increasing due to a rise in the urban population. To meet the need for wastewater collection, the construction of sewer networks must comply with a series of technical parameters that indicate whether a solution is feasible or not. Considering that this construction implies a series of environmental impacts, this study coupled a structural analysis of one linear meter of sewer constructive solutions with their life cycle impacts. Different pipe materials (i.e., concrete, polyvinylchloride (PVC) and high-density polyethylene (HDPE)) were combined with different trench designs and their environmental performance was assessed using life cycle assessment (LCA). These solutions complied with technical parameters consisting of traffic loads and pavement conditions, among others. Concrete pipes embedded in granular matter result in fewer environmental impacts, such as the global warming or cumulative energy demand. Further, re-using the excavated soil results in up to 80% of environmental savings with respect to extracting new materials. Concerning traffic loads and pavement conditions, failures in plastic pipes could be avoided if these are embedded in concrete. Moreover, the environmental impacts of this solution are similar to those resulting from the substitution of pipes that do not comply with the mechanical requirements of the construction site. Therefore, proper planning is needed to provide cities with sewers that are resilient to time and external loads and reduce the urban environmental impacts.

Keywords: sewer, pipe, trench, construction, LCA, urban design

Part III

Environmental effects of sewer operation at the city scale

Chapter 6

Picture: Maintenance truck cleaning a retention tank in Calafell (Catalonia, Spain)

Assessing the energetic and environmental impacts of the operation and maintenance of Spanish sewer networks from a life cycle perspective

Chapter 6 Assessing the energetic and environmental impacts of the operation and maintenance of Spanish sewer networks from a life cycle perspective

This chapter is based on the following paper:

Petit-Boix, A., Sanjuan-Delmás, D., Chenel, S., Marín, D., Gasol, C., Farreny, R., Villalba, G., Suárez-Ojeda, M., Gabarrell, X., Josa, A., Rieradevall, J., 2015. Assessing the energetic and environmental impacts of the operation and maintenance of Spanish sewer networks from a life-cycle perspective. Water Resources Management 29, 2581–2597

doi: 10.1007/s11269-015-0958-2

DDD link: http://ddd.uab.cat/record/132053

Abstract

The environmental impacts resulting from sewer networks are best analyzed from a life cycle perspective to integrate the energy requirements into the infrastructure design. The energy requirements for pumping wastewater depend on the configuration of the city (e.g., climate, population, length of the sewer, topography, etc.). This study analyzes and models the effect of such site-specific features on energy consumption and related effects in a sample of Spanish cities. The results show that the average annual energy used by sewers (6.4 kWh/capita and 0.014 kWh/m^3 of water flow) must not be underestimated because they may require up to 50% of the electricity needs of a typical treatment plant in terms of consumption per capita. In terms of global warming, pumping results in an average of 2.3 kg CO_2 eq/capita. A significant positive relationship was demonstrated between the kWh consumed and the length of the sewer and between other factors such as the population and wastewater production. In addition, Atlantic cities can consume five times as much energy as Mediterranean or Subtropical regions. A similar trend was shown in coastal cities. Finally, a simple predictive model of the electricity consumption was presented that considers the analyzed parameters.

Keywords: Energy, sewer, LCA, operation, city

Chapter 7

Picture: Gas sampling in a manhole in Betanzos (Galicia, Spain)

Municipal sewer networks as sources of nitrous oxide, methane and hydrogen sulfide emissions: A review and case studies

Chapter 7 Municipal sewer networks as sources of nitrous oxide, methane and hydrogen sulfide emissions: A review and case studies

This chapter is based on the following paper:

Eijo-Río, E., Petit-Boix, A., Villalba, G., Suárez-Ojeda, M.E., Marin, D., Amores, M.J., Aldea, X., Rieradevall, J., Gabarrell, X., 2015. Municipal sewer networks as sources of nitrous oxide, methane and hydrogen sulphide emissions: A review and case studies. Journal of Environmental Chemical Engineering 3(3), 2084–2094

doi: 10.1016/j.jece.2015.07.006

DDD link: http://ddd.uab.cat/record/145988

Abstract

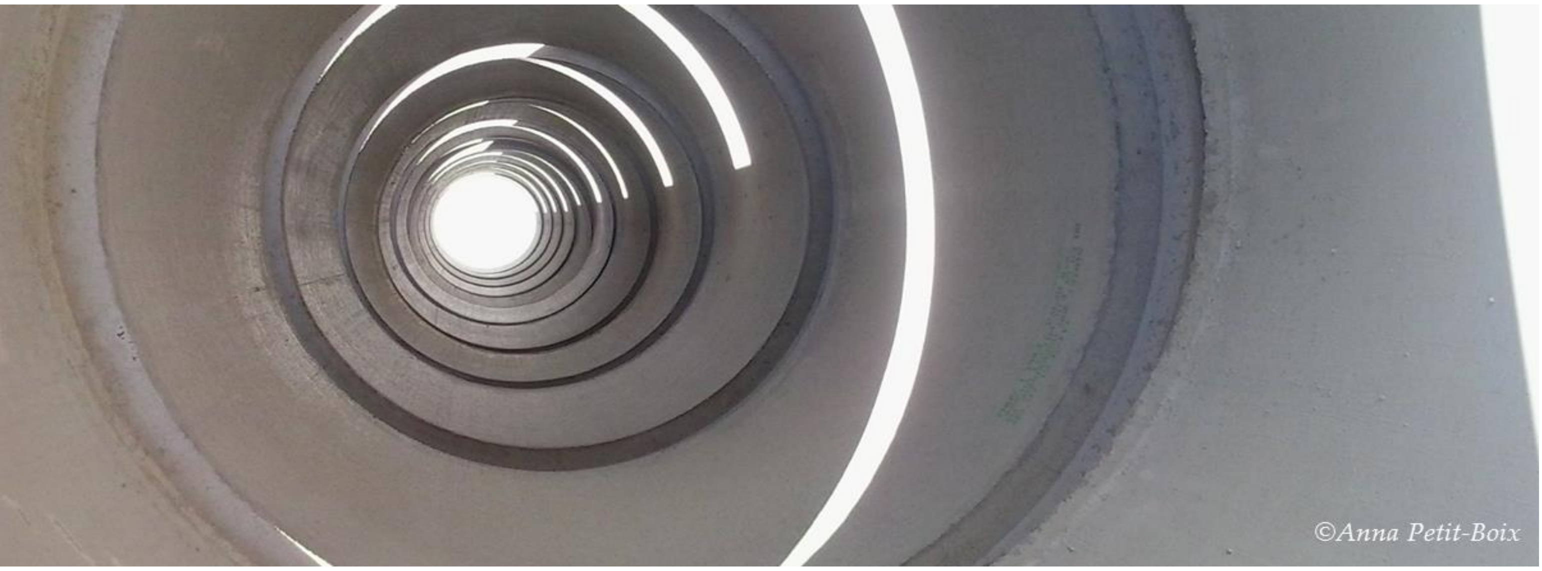
Sewers are known as longitudinal reactors where gases such as methane, nitrous oxide and hydrogen sulfide can be produced. However, gaseous emissions in wastewater infrastructures have been mainly assessed in wastewater treatment plants (WWTP). The aim of this study is to present a critical review of studies that quantify the generation of these gases in sewers and to identify the existing research gaps. Differences in sampling methods and site selection, as well as a very limited number of studies, result in incoherent comparisons amongst sewers. In order to address some of these gaps, sampling campaigns were conducted in two Spanish cities in winter and summer. Results showed that wet wells were the most important sources of gases with concentrations up to 317 μg CH_4/L_{air} and 6.8 μg N_2O/L_{air}. Regarding emission factors, in the case of Calafell, the estimated annual emissions were 18.6 kg CH_4/year and 0.3 kg H_2S/year in summer and 3.8 kg CH_4/year and 0.5 kg H_2S/year in winter. Regarding Betanzos, these values were 24.6 kg CH_4/year and 0.5 kg N_2O/year in summer and 10 kg CH_4/year in winter. The summer campaign resulted in greater gas concentration than in the winter season for both cities, suggesting that temperature is a key parameter. We conclude that gas emissions from sewers are significant compared to those of WWTPs, resulting in an important contribution to the carbon footprint. Further work is needed to assess the gas production along the entire sewer networks, which can result in very different emission factors depending on the sewer components.

Keywords: Sewers, greenhouse gases, urban wastewater, N_2O, CH_4, H_2S, sustainability

Part IV

Conventional and alternative wastewater management in urban areas

Chapter 8



Picture: Concrete pipes piled at the storage area (Lleida, Spain)

Addressing the life cycle of sewers in contrasting cities through an eco-efficiency approach

Chapter 8 Addressing the life cycle of sewers in contrasting cities through an eco-efficiency approach

This chapter had the following collaborators:

Petit-Boix, A., Arnal, C., Marín, D., Josa, A., Gabarrell, X., Rieradevall, J.

Abstract

Evaluating the sustainability of the urban water cycle is not straightforward, although a variety of methods have been proposed. Given the lack of integrated data about sewers, we applied the eco-efficiency approach to two case studies located in Spain with contrasting climate, population, and urban and sewer configurations. Our goal was to determine critical variables and life cycle stages and provide results for decision-making. We used life cycle assessment (LCA) and life cycle costing (LCC) to evaluate their environmental and economic impacts. Results showed that both cities have a similar profile albeit their contrasting features, i.e., operation and maintenance (O&M) was the main environmental issue (50-70% of the impacts) and pipe installation registered the greatest economic capital expenditure (70-75%) due to labor. The location of the wastewater treatment plant (WWTPs) is an essential factor in our analysis mainly due to the topography effects, e.g., annual pump energy was thirteen times greater in Calafell. Using the eco-efficiency portfolio, we observed that sewers might be less eco-efficient than WWTPs and that we need to envision their design in the context of an integrated WWTP-sewer management to improve sewer performance. In terms of methodological approach, the bi-dimensional nature of eco-efficiency enables the benchmarking of product systems and might be more easily interpreted by the general public. However, there are still some constraints that should be addressed to improve communication, such as the selection of indicators discussed in the paper.

Keywords: water cycle, life cycle assessment, life cycle costing, industrial ecology, eco-innovation

8.1 Introduction

Meeting sustainability standards in cities is essential to ensuring the provision of urban services at low environmental, economic and social costs. One of these services is the urban water system, which calls for special attention given the increasing demand for water and sanitation that results from growing urban populations (UN 2012). However, evaluating the sustainability of this system is not straightforward. A variety of methods can be applied to assess the performance of the urban water cycle. For instance, multiple indicators have been used to cover some environmental, economic, socio-cultural, and/or functional criteria (Balkema et al. 2002; Venkatesh and Brattebø 2013; Hellström et al. 2000; van Leeuwen et al. 2012; Muga and Mihelcic 2008; Lemos et al. 2013; Fragkou et al. 2016). These were often combined through multi-criteria approaches to assess diverse sustainability objectives at different scales (Makropoulos et al. 2008; Marques et al. 2015).

Still, objective and comparable quantification is a challenge. We need to provide robust models and data to water facility managers so that they can apply the most viable options. In this respect, the eco-efficiency concept, normalized through ISO 14045:2012, can be particularly useful. This standard describes *eco-efficiency assessment* as "a quantitative management tool which enables the study of life-cycle environmental impacts of a product system along with its product system value for a stakeholder". This tool lacks the social dimension of sustainability (Ehrenfeld 2005), but eco-efficiency is especially attractive because it might provide intrinsic information about potential social benefits (Ekins 2005). For instance, the product system value might be defined through consumer preferences. The need for a consistent approach (Brattebø 2005) is covered through ISO 14045:2012, which sets a methodological framework for assessing the eco-efficiency of products and systems.

In the field of urban water management, the eco-efficiency of sewer networks is worth analyzing. In general, there is an apparent interest in the absolute and relative environmental impacts of wastewater treatment plants (WWTP) around the world (Corominas et al. 2013), which have mainly been evaluated through life cycle assessment (LCA). However, few of these LCAs include the pipe infrastructure (Loubet et al. 2014). As opposed to WWTPs, which are generally affected by climatic conditions, a particularity of sewers is the effect of urban configuration on the energy required to operate the system (Petit-Boix et al. 2015a). Within the existing literature, articles have mainly focused on the environmental impacts of the construction or full life cycle of sewers (Morera et al. 2016; Vahidi et al. 2015, 2016, Petit-Boix et al. 2016, 2014; Venkatesh et al. 2009; Risch et al. 2015) and a few studies have provided economic data (Akhtar et al. 2014; Thomas et al. 2016; Murla et al. 2016). Only Lorenzo-Toja et al. (2016) conducted an eco-efficiency benchmarking of WWTPs following ISO 14045:2012, but sewers were not analyzed.

In this context, what are the hotspots that might alter the eco-efficiency of sewers? Our goal was to apply the eco-efficiency approach to sewer networks in order to determine critical variables and life cycle stages and provide results and discussion for decision-making in the context of the urban water cycle. To address our questions, we based our assessment on two cities with contrasting urban conditions and climate in an attempt to represent major areas of the globe, i.e., an Atlantic city with year-round population and a Mediterranean, coastal city with seasonal population. To quantify the eco-efficiency of the systems, we followed the guidelines described in ISO 14045:2012.

8.2 Materials and Methods

4.2.1 Case study definition

To answer our research question, we studied two Spanish cities with different urban and climatic features in the framework of the LIFE+ Aquaenvec project (LIFE10/ENV/ES/520). These cities present different conditions that we used to test whether the eco-efficiency of sewers varies depending on the climate, population, and sewer and urban configurations. Betanzos is located in the northwest of Spain and has an Atlantic climate that results in more than 1,000 mm of rainfall every year. Wastewater flows from households to a WWTP located at sea level and most of the network is a gravity sewer due to the topography. In contrast, Calafell is a coastal, Mediterranean city with an annual rainfall of around 500 mm. Because of land price and usually odor control, the WWTP was constructed inland and 40 m above sea level, which results in greater pumping requirements and length of sewer than in Betanzos (**Table 8.1**). Both cities can be considered medium-sized based on their population (10,000 – 50,000 inhabitants), although in Calafell it usually doubles in the summer (Idescat 2016).

The sewer components were identified through the water managers and the SGO (Operation Management System) and CONTEC (Technical Control of the Integral Water Cycle) databases (©Suez services company 2012). These were mainly combined sewers, with a portion of stormwater network in Calafell. The network has a total length of 77 and 173 km in Betanzos and Calafell, respectively. They consisted of concrete, fibrocement, high-density polyethylene (HDPE) and polyvinylchloride (PVC) pipes. Both cities had a greater share of plastic pipes, i.e., 66% of PVC in Betanzos and 73% of HDPE in Calafell, and diameters of 300-315 mm dominated (50-75% of the network). In the case of sewer appurtenances, the number of manholes and inspection chambers was estimated assuming one unit every 50 meters of sewer (Petit-Boix et al. 2014). The CONTEC database provided the number of scuppers, wastewater connections and submersible pumps (©Suez services company 2012). However, the power of the pumps was unknown and we tested a scenario with a 60 m^3/h pump to account for pump production, which is the highest flow we found in construction databases (MetaBase ITeC 2010). We did know the real electricity consumption of the system, and the wastewater production was registered at the WWTP.

Table 8.1 Main features of the sewer networks in the cities analyzed. Based on data provided by the local water managers for the year 2011.

Features		**Betanzos**	**Calafell**
Climate		Atlantic	Mediterranean
Year-round population		13,537	24,984
Total equivalent population (year-round + tourism)		13,672	38,211
Wastewater production (m^3/year)		1,145,699	3,349,749
Pipeline			
Material	**Diameter (mm)**	**Length (m)**	
Concrete	250	721	929
	300	22,081	22,718
	350	0	7,704
	400	0	3,697
	500	120	7,053
	600	381	896
	700	0	1,065
	800	0	1,883
	1000	0	553
	1200	819	251
Fibrocement	400	1,518	0
HDPE	110	727	0
	200	243	11,856
	250	0	4,438
	300	0	67,660
	350	0	2,208
	400	0	21,540
	450	0	2,489
	500	0	9,369
	700	0	1,680
	800	0	4,766
	1200	0	700
PVC	75	819	0
	90	943	0
	110	57	0
	200	3,431	0
	250	8,654	0
	315	35,249	0
	400	1,700	0
Total pipeline		77,462	173,454
Appurtenances		**Number of units**	
Manhole		1,578	4,154
Inspection chamber (30x30 cm)		21	201
Inspection chamber (50x50 cm)		1,556	3,458
Inspection chamber (100x100 cm)		22	495
Submersible pump (60 m^3/h)		13	15
Scupper		1,599	1,199
Wastewater connection		2,638	9,239
Pumping energy (kWh/year)		121,591	1,577,054

4.2.2 *Eco-efficiency assessment method*

The methodological framework of an eco-efficiency assessment combines the environmental and value assessment of a product system (ISO 14045:2012). To do so, ISO 14045:2012 includes two specific requirements for choosing eco-efficiency indicators. The ratio between the environmental and value dimensions can either depict an improved environment at the same product system value or an improved product system value at the same environmental effect. These results can be represented through eco-efficiency portfolios that illustrate the pathway towards the desired eco-efficiency and can be used in the benchmarking of a product system using optimization functions. For a given functional unit (FU), practitioners should define the indicators applied, as these are not provided by the standard and might vary depending on the analysis.

In general, this ISO standard is relatively open and flexible in terms of methodological approaches. LCA is the method selected to conduct the environmental analysis based on ISO 14040:2006. In the case of the value assessment, the standard calls for an integration of the full life cycle of the product system, but does not establish a specific method for this type of analysis. Based on this ISO, the system value (i.e., its worth or desirability) can be functional, monetary or intangible (i.e., esthetic, cultural, etc.). Because of the life cycle perspective, we typically apply life cycle costing (LCC; ISO 15686-5:2008) and assess the monetary value of a product system. We used this method in our analysis to assess the costs associated with the sewer infrastructure.

4.2.3 *Goal and scope definition*

In this study, we aim to assess the eco-efficiency of sewers by combining the environmental and economic dimensions through the LCA and LCC methods. The FU was the transport of one m^3 of urban wastewater from the households to the WWTP in a medium-sized city through a sewer network. We considered different lifespans depending on the pipe material and sewer component. We assumed 100 years for concrete pipes (U.S. Army Corps of Engineers 1998; CPSA 2010) and 50 years for plastic pipes (UNE 53331:1997). A lifespan of 50 years was assigned to all types of appurtenances except for submersible pumps, which were replaced every 10 years (Petit-Boix et al. 2014).

To determine the environmental and economic results per FU, we followed the method proposed by Petit-Boix et al. (2014). Based on EN 15804:2011, we set different declared units for the construction assets, i.e., one linear meter of pipe-trench constructive solutions and one unit of each appurtenance. To account for the total impacts of the system, we scaled to the total sewer components (**Table 8.1**) and combined with the operation and maintenance (O&M) of the system.

The same system boundaries were considered in the LCA and LCC (**Figure 8.1**). These included the raw material procurement, pipe production, transport to the construction site, pipe installation and trench preparation, and O&M. The demolition was excluded because it was negligible (Petit-Boix et al. 2014; Gabarrell et al. 2013). The end-of-life stage was not accounted for because the pipe can be either disposed of or left underground. The LCA does not include the emissions that result from wastewater degradation on its way to the WWTP, such as methane, nitrous oxide, and hydrogen sulfide, because a model is still needed to predict the emissions of the entire network. According to Eijo-Río et al. (2015), these emissions might represent at least 4% of the O&M impacts and should be accounted for in future assessments.

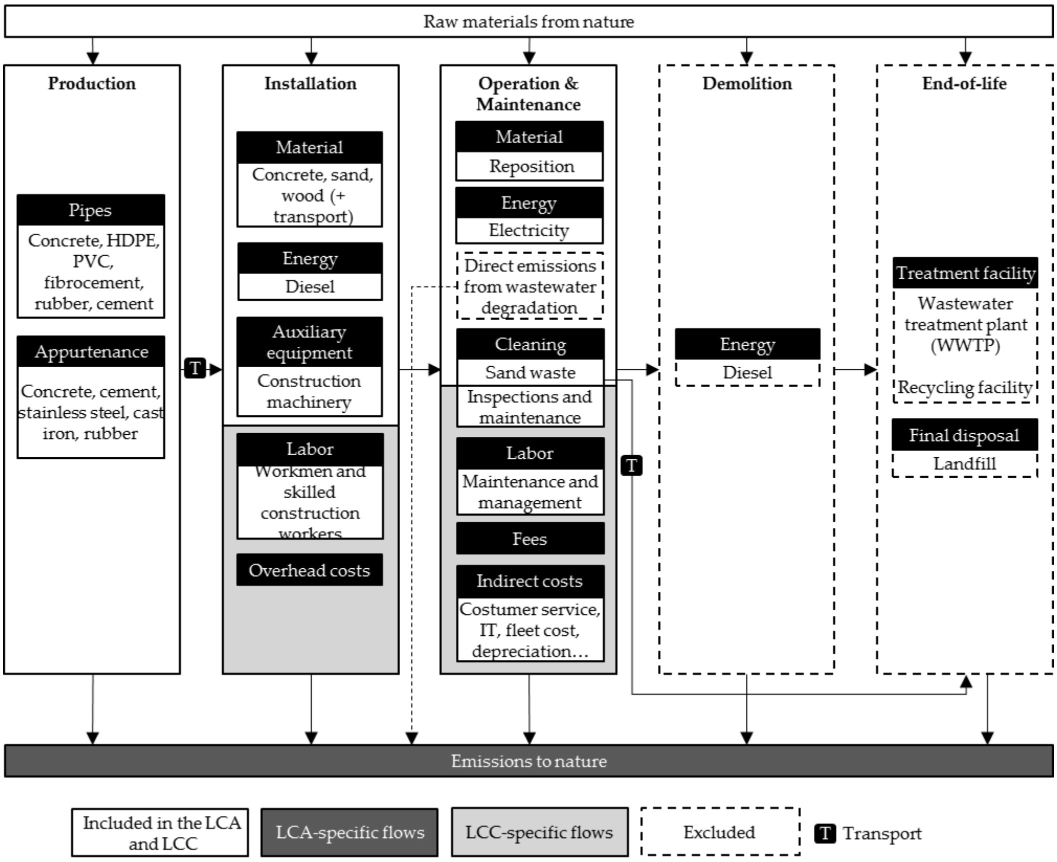


Figure 8.1 System boundaries of the LCA and LCC of sewers

4.2.4 Life cycle inventory (LCI)

Table 8.2 provides a detailed inventory of the material and energy flows involved in the life cycle of the sewers in Calafell and Betanzos. Note that the production and installation stage includes the pipes and appurtenances. The LCI of sewer components is provided in **Chapters 4, 5** and their associated annex. When inventorying the pipelines, we considered different trench designs depending on the pipe material. Based on the results reported by Petit-Boix et al. (2016), we adapted the worst designs to show the maximum environmental impacts of the system. Concrete pipe trenches had a bedding factor of 4, which is the highest safety level (EN 1916:2002) and consists of the largest amount of concrete bedding. Plastic pipes were embedded in sand imported from other areas. The excavated soil was left aside for other purposes and its management was outside of our scope. The appurtenance design was adapted from the literature (Petit-Boix et al. 2014) and databases (MetaBase ITeC 2010; CYPE Ingenieros 2015).

Data on the energy consumed in the installation process and the materials used in the pipe production were retrieved from MetaBase ITeC (2010). A truck covered a distance of 100 km to transport plastics and metals to the construction site and 30 km to transport the remaining materials (Doka 2003). The ecoinvent v2.2 database (Frischknecht et al. 2005) was used to obtain background information on the life cycle of the materials and processes involved.

Table 8.2 Life cycle inventory divided into the construction and O&M in the case study cities. Data per FU (1 m^3 of wastewater)

	Flows	Betanzos		Calafell	
		Quantity	Units	Quantity	Units
Production and installation	Mass concrete pipe	2.2E-02	kg	1.8E-02	kg
	Fibrocement pipe	9.6E-04	kg	0	kg
	PVC for pipe and appurtenance	1.4E-03	kg	9.9E-05	kg
	HDPE for pipe	2.0E-05	kg	6.3E-03	kg
	Synthetic rubber	2.9E-06	kg	7.6E-05	kg
	Polyurethane	1.3E-05	kg	0	kg
	Extrusion process	1.4E-03	kg	6.4E-03	kg
	Cement mortar	5.8E-03	kg	4.6E-03	kg
	Concrete	8.2E-05	m^3	6.1E-05	m^3
	Concrete block	5.6E-02	kg	5.2E-02	kg
	Sand	4.2E-01	kg	4.1E-01	kg
	Gravel	1.5E+00	kg	1.3E+00	kg
	Wood	1.7E-05	m^3	1.3E-05	m^3
	Cast iron	3.3E-03	kg	2.7E-03	kg
	Stainless Steel	4.1E-06	kg	1.3E-06	kg
	Metal product manufacturing	3.4E-03	kg	2.7E-03	kg
	Diesel	1.6E-01	MJ	1.4E-01	MJ
	Transport	1.3E-01	tkm	1.2E-01	tkm
Operation and Maintenance (O&M)	Electricity	1.1E-01	kWh	4.7E-01	kWh
	Mass concrete pipe	0	kg	0	kg
	Fibrocement pipe	1.9E-05	kg	0	kg
	PVC for pipe	1.4E-03	kg	9.9E-05	kg
	HDPE for pipe	2.0E-05	kg	6.3E-03	kg
	Synthetic rubber	2.9E-06	kg	7.6E-05	kg
	Polyurethane	1.3E-05	kg	0	kg
	Extrusion process	1.4E-03	kg	6.4E-03	kg
	Cement mortar	4.7E-03	kg	3.9E-03	kg
	Concrete	1.4E-05	m^3	1.1E-05	m^3
	Concrete block	5.6E-02	kg	5.2E-02	kg
	Sand	3.4E-01	kg	3.5E-01	kg
	Gravel	1.2E+00	kg	1.2E+00	kg
	Wood	1.3E-05	m^3	1.1E-05	m^3
	Cast iron	3.5E-03	kg	2.7E-03	kg
	Stainless Steel	1.9E-05	kg	7.2E-06	kg
	Metal product manufacturing	3.5E-03	kg	2.7E-03	kg
	Diesel	1.2E-01	MJ	1.1E-01	MJ
	Transport	1.0E-01	tkm	9.7E-02	tkm
	Sand removal and transport	n.d.	tkm	4.5E-03	tkm

The O&M consisted of different variables. First, the energy required to pump wastewater to the WWTP was provided by the facility managers. The electricity was modeled using the Spanish mix of the year 2011 (REE 2012), which is the year that we obtained data from. Second, the length of sewer and number of appurtenances renovated every year was unknown. We assumed a certain number of repositions depending on their lifespan, e.g., when a component had a 50-year lifespan, we considered one reposition in 100 years. Only in the case of fibrocement did we assume a 2% renovation rate according to local estimates. Third, maintenance involved cleaning tasks, e.g., removal and transport of sediments accumulated in the sewers. This data was available in the case of Calafell and we considered the number of trips covered by the inspection and maintenance service (48 trips/year; 75 km) and the average amount of sediments (4,000 kg/year). We could not apply these estimates to Betanzos, but assumed a negligible maintenance based on onsite observations.

All the construction flows were translated into monetary values through MetaBase ITeC (2010) and CYPE Ingenieros (2015). The value assessment includes additional costs, such as labor, overheads, fees and indirect costs (**Figure 8.1**). Labor and overhead data related to construction processes were available in the aforementioned databases. O&M costs were retrieved from financial statements and invoices provided by utility managers. In this case, the reposition costs of 2011 were obtained, but the reposition rate was not available. Due to limited data availability, the economic costs were not broken down into basic flows. The total cost of each sewer component is provided in the **Appendix 5.1**.

4.2.5 Life cycle impact assessment (LCIA) and indicator selection

The selection of environmental indicators might be more complex when communicating eco-efficiency results. In the eco-efficiency portfolios, only one indicator can be represented and selecting one or another might lead to the common tradeoffs that occur in LCA studies. For this reason, we conducted the impact assessment at three levels, i.e., using midpoints, endpoints, and a single score indicator. These were modeled with the ReCiPe (H) method (Goedkoop et al. 2009) and the Simapro 8 software (PRé Consultants 2014).

We used endpoints to determine the specific damage of the production, installation, and O&M to human health, ecosystems and resources. As we were interested in determining hotspots in the eco-efficiency of sewers, endpoints might reduce the complexity of interpretations because decision-makers do not need to identify the environmental relevance of each midpoint indicator (Bare et al. 2000). This can also be achieved through an integrated single score indicator, which represents the weighted endpoints in terms of eco-points (Pt). Although endpoint and single score indicators have an increased uncertainty and subjectivity with respect to midpoints, we used them in the eco-efficiency assessment for an easier understanding and communication. Some argue that endpoints provide a more structured factor weighting when comparing (Udo de Haes et al. 2002), as they are closer to our concerns, such as health issues, and can be easily valuated (Hertwich and Hammitt 2001). Nevertheless, there is a loss of comprehensiveness and increased uncertainty in endpoint and damage analysis due to the modeling principles, assumptions and value choice (Bare et al. 2000).

For this reason, a recommendation is to provide both the midpoint and endpoint results to increase the transparency of the analysis (Kägi et al. 2016). We used a set of 18 ReCiPe midpoints and the cumulative energy demand (CED) (Hischier et al. 2010) to break down the environmental impacts into sewer components and O&M flows. By doing so, we identified the impacts generated by basic flows (e.g., materials, energy, etc.) at early stages of the cause-effect chain, such as the potential resource depletion or pollutant emissions.

4.2.6 Economic indicator selection

The economic costs were calculated differently in the construction and O&M processes. The O&M costs were based on the financial statement of the water facilities, which include the energy, fees and indirect operation costs. In the case of the pipe production and installation, the direct unit costs (DUC) resulting from the LCI were converted into equivalent annual costs (EAC). By doing so, we accounted for the annual costs of the sewer construction considering a time horizon of 100 years, which is the potential maximum lifespan of concrete pipes. **Equations 8.1, 8.2 and 8.3** illustrate the conversion of the DUC into EAC. To calculate the total unit cost (TUC), we considered that the indirect costs (IUC) were 10% of the DUC. The general costs (GUC) and industrial profit (IP) represented 13% and 6% of the execution material budget (i.e., direct plus indirect costs), respectively (BOE 2001a). A 3% interest rate was assumed to estimate the present value (PV) and EAC, with a time horizon of 100 years according to the FU of the analysis.

$$\mathrm{TUC} = \mathrm{DUC} + \mathrm{IUC} + \mathrm{GUC} + \mathrm{IP} = 1.309\ \mathrm{DUC} \qquad \textbf{Equation 8.1}$$

$$\mathrm{PV} = \frac{\mathrm{TUC}}{(1+\mathrm{i})^{\mathrm{t}}} \qquad \textbf{Equation 8.2}$$

$$\mathrm{EAC} = \frac{\mathrm{PV} \times \mathrm{i}}{1-(1+\mathrm{i})^{-\mathrm{n}}} \qquad \textbf{Equation 8.3}$$

Where TUC: total unit cost; DUC: direct unit cost; IUC: indirect unit cost; GUC: general unit cost; IP: industrial profit; PV: present value; EAC: equivalent annual costs; i: interest rate (3%); t: lifespan (present, t = 0); n: time horizon (100 years).

8.3 Results and Discussion

In this section, we identified the life cycle stages with a poor environmental and economic performance and sought possible explanations. These dimensions were compared and eco-efficiency results were discussed in the context of the urban water cycle.

4.3.1 Environmental and economic characterization of the sewers

A set of environmental and economic results is shown in **Table 8.3**. The environmental and economic hotspots were similar in both cities, but the total impacts of Betanzos and Calafell were especially different. For instance, the impacts to human health were 4.1E-07 and 6.9E-07 DALYs/m^3, respectively, whereas the total economic costs amounted to approximately one €/m^3 in both cases.

Identification of environmental hotspots

The O&M was the most relevant phase of the environmental life cycle and contributed to approximately 50% and 70% of the impacts to human health, ecosystems and resources in Betanzos and Calafell, respectively. The main difference between both cities was the energy required to pump wastewater. Within the O&M, the electricity accounted for 30% and 70% of the impacts in Betanzos and Calafell, respectively. These percent contributions to the endpoint indicators resulted from the midpoint breakdown shown in **Figure 8.2** (the acronyms and absolute midpoint results are shown in **Appendix 5.2**). The ionizing radiation (IR) is the category where electricity contributed most (around 90%), as 21% of the Spanish energy demand was covered by nuclear power in 2011 (REE 2012). The contribution of the O&M to the life cycle impacts might be even greater once the direct gas emissions are included in the assessment.

Table 8.3 Environmental and economic results of Calafell and Betanzos divided into life cycle stages (Data per FU: 1 m^3)

	Dimension	Indicators	Units	Production	Installation	O&M	Total
Betanzos	Environment	Human Health	DALY	6.1E-08	1.3E-07	2.2E-07	4.1E-07
			% contribution	*15%*	*32%*	***53%***	*100%*
		Ecosystems	species.yr	2.3E-10	1.1E-09	1.3E-09	2.6E-09
			% contribution	*9%*	*42%*	***49%***	*100%*
		Resources	$	1.7E-03	3.0E-03	5.7E-03	1.0E-02
			% contribution	*16%*	*29%*	***55%***	*100%*
	Economy	AEC	€	0.06	0.73	0.19	0.98
			% contribution	*6%*	*75%*	*19%*	*100%*
Calafell	Environment	Human Health	DALY	7.0E-08	1.1E-07	5.1E-07	6.9E-07
			% contribution	*10%*	*16%*	***74%***	*100%*
		Ecosystems	species.yr	2.8E-10	9.0E-10	2.4E-09	3.6E-09
			% contribution	*8%*	*25%*	***67%***	*100%*
		Resources	$	2.9E-03	2.6E-03	1.4E-02	1.9E-02
			% contribution	*15%*	*14%*	***71%***	*100%*
	Economy	AEC	€	0.10	0.72	0.21	1.03
			% contribution	*9%*	***70%***	*20%*	*100%*

Besides the electricity consumption, the type of sewer component, material, design and lifespan had a relevant effect on the results. The sewer components can be classified in order from most to fewest impacts (**Figure 8.2**). Depending on the impact category, plastic pipelines accounted for the largest impacts, with a maximum contribution of 40%, followed by appurtenances (30%) and concrete pipelines (10%). The larger contribution of plastics can be easily associated with the length of sewer, as these were the main materials used in the pipeline. As shown in **Appendix 5.2**, the pipe itself had an irrelevant contribution to the total impacts (<10%), but the trench played a key role in the construction stage as predicted by Petit-Boix et al. (2014, 2016). In this case, we accounted for the worst case scenario, but Venkatesh et al. (2009) chose not to include the trench materials because they were reused from the excavation. The impacts of appurtenances were notable in the midpoints related to toxicity and metal depletion because iron and steel parts were used in their construction (**Table 8.2**). Based on these results, the impacts of the O&M increased due to the reposition needs. Plastic pipelines and appurtenances had shorter lifespans (50 years) than concrete pipes (100 years), which means that the reposition was related to the components with the greatest environmental impacts. An alternative might be to implement concrete pipelines that have a longer service life and better environmental performance.

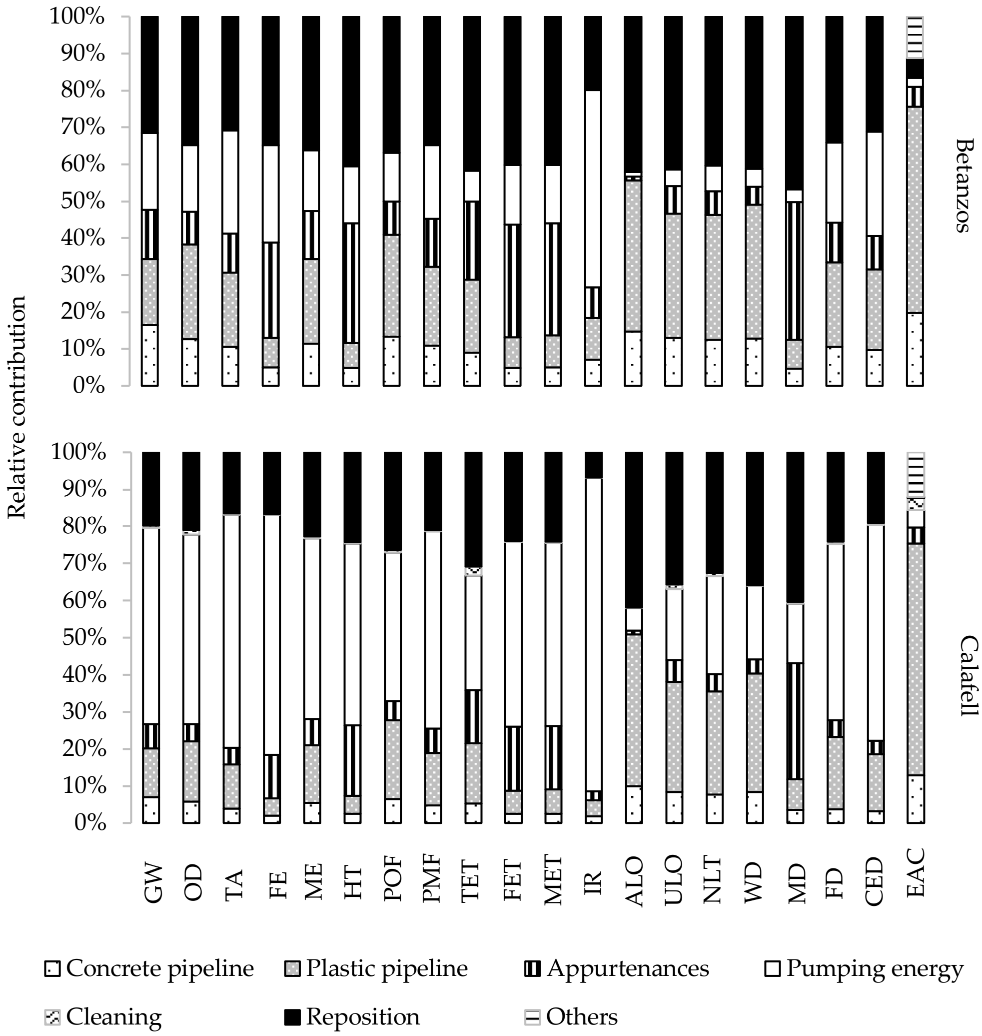


Figure 8.2 Distribution of the environmental and economic impacts of the sewer components and O&M flows of the sewers in Calafell and Betanzos

Identification of economic hotspots

Contrary to the environmental assessment, the installation stage resulted in the largest economic investment (**Table 8.3**). The cost of this stage amounted to 0.72-0.73 €/m^3, which represented 70% and 75% of the total investment made in Calafell and Betanzos, respectively. Because the O&M included the fees, personnel, and recurring maintenance costs (named "others" in **Figure 8.2**), we expected that these would account for a greater share of the annual expenses. However, they represented 10% of the total costs.

Labor was the reason why the installation was more expensive on an annual basis. The need for skilled construction workers resulted in 50% of the investment at this stage of the life cycle (see **Appendix 5.1**). In social terms, this investment might result in positive effects, as it enhances the staff recruitment in the area. It was also notable that plastic pipelines were the most expensive item in the system and accounted for 60% of the construction costs in both cities, which was also associated with the installation stage. As opposed to the LCA, note that we obtained real reposition costs and did not apply the equivalent costs of the initial construction of the sewer. In this case, the reposition was almost negligible.

Explanatory variables to local differences

The environmental and economic trends were very similar in both case studies. Nevertheless, the environmental effects of the O&M stage were greater in Calafell (e.g., 5.1E-07 DALY/m^3) than in Betanzos (e.g., 2.2E-07 DALY/m^3). Previous analyses tried to explain general variations in the O&M of sewers (Petit-Boix et al. 2015a), but an analysis of specific case studies might help to delve deeper into the differences among cities.

Different reasons might be associated with the energy requirements of these sewers, which are 0.11 and 0.47 kWh/m^3 in Betanzos and Calafell, respectively. Some of them include the rainfall patterns, population, length of sewer, and location of the WWTP. We checked each of these variables to determine which one might be contributing most to the results of each city. Betanzos was expected to have an increased energy demand because the Atlantic climate results in constant rainfall throughout the year. Additionally, tides affect this region and might result in tidewater entries to the combined sewer (Day 2000). However, these effects were not apparent when comparing Betanzos to Calafell. The influence of population can be assessed through the wastewater generation per capita. Based on **Table 8.1**, we estimated a wastewater production of 84 and 135 m^3/capita/year in Betanzos and Calafell, respectively. Nevertheless, Calafell had a seasonal inflow of 13,227 tourists that doubled the number of residents, so the total equivalent population should be applied instead (**Table 8.1)** (Idescat 2016)**.** The resulting wastewater generation was 87 m^3/capita/year, which is similar to that of Betanzos and does not explain the difference in energy consumption. The length of sewer needed to connect the city to the WWTP might influence the electricity consumption as predicted by Petit-Boix et al. (2015). Based on **Table 8.1**, the length per m^3 of wastewater was 0.05-0.07 km, which does not explain the difference between the cities.

The last variable that we addressed was the location of the WWTP. Looking at urban planning, Calafell's WWTP was located at a higher altitude than that of Betanzos. Although Betanzos had intermediate pumping stations to deal with topographic variations, the WWTP was at sea level. Consequently, the annual energy intensity of the pumping system was thirteen times greater in Calafell. A set of pumping stations directed the wastewater flow to a larger station that was responsible for connecting the city to the WWTP through a rising sewer. The WWTP was three kilometers away from this point and 40 m above sea level. For this reason, we believe that the main variable that affected the O&M was the location of the WWTP.

4.3.2 *Assessing the eco-efficiency of sewers*

At this stage, we can apply the eco-efficiency portfolios to integrate the environmental and economic results and facilitate the decision-making process. Because the economic results (EAC) are based on an LCC, we showed the environmental effects associated with economic investment, which is the eco-efficiency type called environmental intensity of production (Huppes and Ishikawa 2005a).

In **Figure 8.3** we used a single score indicator to aggregate the environmental impacts in a single unit. With this approach, we could identify the pathway towards eco-efficiency in each life cycle stage. The shaded areas highlight the location of each life cycle stage in the portfolio based on the case-study results. When compared to the other life cycle stages, every euro invested in the O&M resulted in large environmental impacts. Again, the highest values (e.g., 140 mPt/€) were associated with an increased energy consumption in Calafell, which had a low economic cost (**Figure 8.2**). The opposite situation occurred in the installation stage. Here, the economic investment was high but mainly related to labor costs, which do not have an environmental equivalent. However, these might be an indirect measure of the number of workers involved, which could be a positive social indicator. We found similar results when we used the GW instead of the single score indicator in the eco-efficiency representation (see the **Appendix 5.3**).

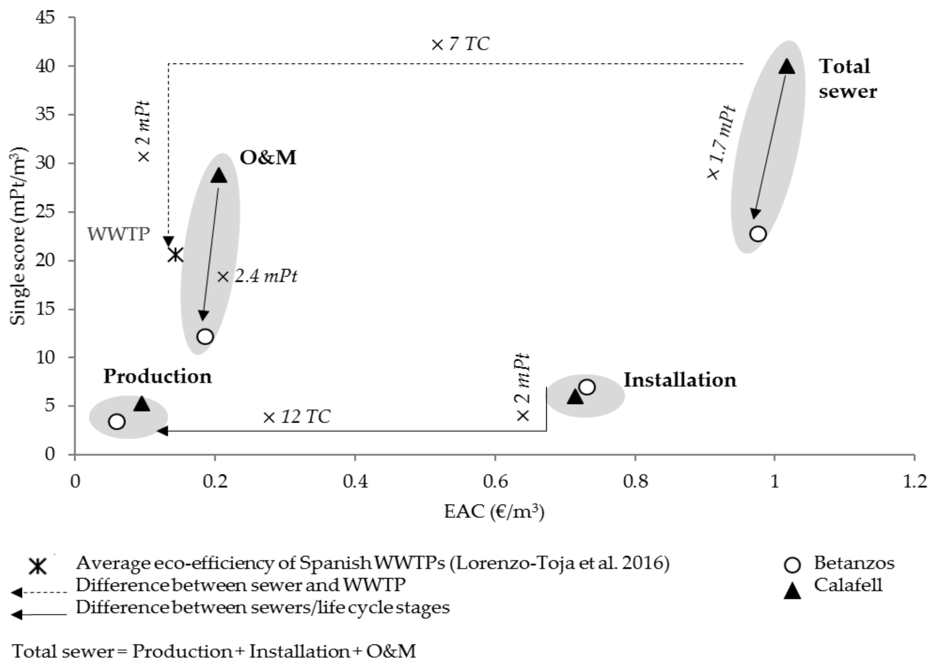


Figure 8.3 Eco-efficiency portfolio illustrating the differences amongst life cycle stages in Betanzos and Calafell

The portfolio also provides some guidance on the pathway towards eco-efficiency improvements. Given our findings on the effects of topography, a preliminary urban analysis is needed to find feasible locations of the WWTP that reduce the environmental footprint of the system. In contrast, the main concern in terms of infrastructure was the initial economic investment. When compared to the pipe and appurtenance production, the installation might double the production impacts, but the costs had the largest room for improvement, these being twelve times greater. In this case, the type of trench design, which is concrete intensive, might require more labor hours than other solutions, suggesting that an optimized system is needed.

It is also interesting to put the sewers into the context of the water cycle. As a first estimation, we compared the eco-efficiency of our case studies with the average eco-efficiency of Spanish WWTPs based on Lorenzo-Toja et al. (2016). Through this comparison, we observed that the economic and environmental costs of Calafell's sewer were seven and two times greater than the WWTP average. When comparing it to Betanzos, the environmental impacts did not differ, but the economic costs were also seven times greater in the sewer. This is related to the construction phase, as the environmental impacts of constructing WWTPs are negligible when compared to the O&M (Lorenzo-Toja et al. 2016b; Termes-Rifé et al. 2013) because of the lifetime of the infrastructure. This showed that sewers are not irrelevant in the framework of the urban water cycle, meaning that decisions made at the design phase will determine their performance.

However, this can be challenging. The complexity of WWTPs is associated with the efficiency of the O&M to meet water quality standards. In the case of sewers, indirect decisions might result in greater effects than the mere selection of pipe materials. Urban planning could have two main consequences on the eco-efficiency of sewers. A distant WWTP results in a longer pipeline; if it is located at a higher altitude than the city, this also involves more pumping energy. These decisions translate into avoidable economic investments in infrastructure and environmental burdens related to pumping energy. Therefore, there is a need to envision sewers in the context of the water cycle in order to design the system more eco-efficiently.

Still, sewer studies are underrepresented as compared to those dealing with the impacts and costs of water and wastewater treatment. Loubet et al. (2014) found that the environmental footprint of sewers is minimal in most studies, and the O&M contributed to less than 10% of the total economic costs in a water cycle analysis elaborated by Venkatesh and Brattebø (2011). Most of these analyses do not account for trench materials or appurtenances and for this reason our results might be higher. In general, more efforts are needed to characterize sewers under different conditions to better understand their role in the water cycle.

4.3.3 On the application of eco-efficiency

Similarly to life cycle analyses, eco-efficiency is a robust communication tool, but we believe that it has a greater outreach potential. The bi-dimensional nature of this approach enables the benchmarking of product systems and it can be easily interpreted by the general public. Some problems arise, though, when deciding the environmental and economic indicators that define the product system. In our case, we used a measure of the environmental intensity of production. This means that the optimum eco-efficiency accounts for reduced environmental impacts generated through reduced production costs. On the one hand, monetary costs and LCC are commonly used when addressing the economic dimension, but one might argue that this approach is not complete because it does not cover aspects such as economic growth or value creation (Haes et al. 2004). On the other hand, we selected a set of indicators, but standards should provide some guidance and discuss the suitability of aggregated indicators in the context of eco-efficiency communication tools.

We proved that the procedure presented in ISO 14045:2012 can be applied to urban systems and provided integrated data for the decision-making process. An optimized system might serve as a reference for sewer benchmarking, but we did not provide this result because several parameters are at play. For instance, policies and social perceptions should be accounted for, as these determine the location and configuration of the sewer. Our approach serves as a first step towards integrating economic and environmental variables in the context of urban sanitation, which is helpful to decision-makers and might in fact change the social aspects associated with the impacts of sewers. Urban systems are a good example of micro-scale effects to macro-scale eco-efficiency, which is an approach to consider in order to avoid tradeoffs (Huppes and Ishikawa 2005b). If service providers (e.g., pipe manufacturers) improve their eco-efficiency, these might become more competitive in the market and result in further positive effects in the context of the urban water cycle and the overall performance of our cities.

8.4 Conclusions

The eco-efficiency approach helped us to determine key hotspots in the environmental and economic performance of sewers. We studied two case studies with contrasting features in terms of population, climate, urban and sewer configuration, and yet we obtained similar trends. The critical life cycle stages were the O&M in environmental terms, and the installation in economic terms. The impacts of the O&M were associated with the location of the WWTP and the consequent energy needs. Labor was the main economic flow that affected the investment in the installation of sewers. This factor should be further assessed, as it might entail social benefits that are not directly captured by the eco-efficiency approach. Additionally, one of the sewers resulted in seven and two times as many economic and environmental impacts as an average WWTP. This means that sewers are in a critical and challenging position that calls for an integrated assessment of the urban water cycle.

To the authors' knowledge, this is the first application of ISO 14045:2012 to sewers. We believe that this is a method with a great potential in terms of communication, although integrating further social aspects should be considered. This study suggests that this type of assessment may well encourage water managers and local administrations to implement more sustainable alternatives in facility planning and management. Benchmarking their performance might be a compelling approach, as it shows their improvements with respect to similar services and drives their pathway towards a more eco-efficient behavior.

Chapter 9

Picture: Constructed wetland in Maumee Bay State Park, Oregon (Ohio, USA)

The centralization paradigm: life cycle assessment of sanitation scenarios in an island's seasonal suburban settlement

Chapter 9 The centralization paradigm: life cycle assessment of sanitation scenarios in an island's seasonal suburban settlement

This chapter had the following collaborators:

Petit-Boix, A., Josa, A., Gabarrell, X., Rieradevall, J., Apul, D.

Abstract

Islands are tourist destinations where wastewater management is an environmental issue, especially in seasonal suburban settlements that host most residents in the summer. Our goal was to identify if, in this specific case, centralized systems are beneficial due to economies of scale or if partial/total decentralization is an environmental advantage in suburban areas. We analyzed a case study in Minorca (Spain) that faces these problems and compared five scenarios, i.e., septic tanks; septic tanks combined with local wastewater treatment plant (WWTP); constructed wetland combined with local WWTP; and connection to a neighboring, larger WWTP. A life cycle assessment showed that the large WWTP was the best scenario because of economies of scale, but it scored unsatisfactorily in some indicators because of the sewer needs. In terms of eutrophication, centralization and partial decentralization through septic tanks achieved the largest improvements in phosphorus emissions, whereas the constructed wetland generated the smallest nitrogen emissions. Through a probabilistic approach, we generalized these results to other insular regions. Values were sensitive to the duration of the tourist season, which is related to the type of accommodation (i.e., mass or second-home tourism), but also to wastewater generation because of the sewer pumping demand.

Keywords: cities, industrial ecology, tourism, wastewater treatment, sustainability

9.1 Introduction

Wastewater management is crucial in islands, where limited land availability usually leads to importing resources and exporting waste. Additionally, islands are frequently tourist destinations that host a large flow of seasonal population. In this context, tourism might lead to conflicts between permanent and seasonal residents because of the pressure on water systems, among others (Müller et al. 2004). However, tourism might also result in local improvements, as cities might decide to upgrade their wastewater treatment to meet the visitors' expectations (Gössling et al. 2012). According to Scoullos (2003), sewers collect more than 80% of the total effluent generated in the Mediterranean, but only half of these networks connect to a wastewater treatment plant (WWTP), which results in significant discharges of untreated water into the sea. Considering that 40% of the world's population lives in coastal areas (Crossland et al. 2005), wastewater treatment issues become apparent. Seasonal suburban settlements are one of the areas that lack an organized wastewater management plan. These host a reduced number of residents throughout the year, but experience an abrupt population increase in the summer. In this sense, engineering solutions should be adapted to insular tourist hubs to deal with seasonal wastewater treatment at the minimum environmental costs based on existing centralized and decentralized technologies.

The environmental performance of WWTPs was studied based on the size of the population served (Rodriguez-Garcia et al. 2011; Hospido et al. 2008; Gallego et al. 2008; Lorenzo-Toja et al. 2015). Lassaux et al. (2007) determined that treating larger quantities of wastewater results in fewer environmental impacts. In fact, some studies identified environmental economies of scale, with large centralized WWTPs being beneficial for some environmental indicators (Lundin et al. 2000; Tillman et al. 1998; Shehabi et al. 2012; Cornejo et al. 2016). Both the construction and operation phase might result in lower environmental burdens per cubic meter (m^3) of wastewater in large-scale systems. However, it is not clear whether connecting a suburban area to an existing, neighboring WWTP might also result in economies of scale, as new sewerage infrastructure is required. When communities are scattered, different decentralized solutions should also be analyzed in order to meet sustainability criteria.

The particularity of decentralized and satellite systems is that they might be a solution to water stress issues and peak wastewater generation (Gikas and Tchobanoglous 2009). Decentralized systems can be applied at different scales, such as houses or suburbs, but their feasibility depends on site-specific features. For instance, septic tanks are suitable in large properties where wastewater discharge can be absorbed by the soil (Sharma et al. 2010). This absorption is possible by connecting septic tanks to sand filters or wetlands. In this sense, some authors studied the environmental impacts of septic tanks attached to wetlands (Pan et al. 2011; Fuchs et al. 2011; Yıldırım and Topkaya 2012). Lehtoranta et al. (2014) compared this system with other treatment alternatives and determined that composting toilets resulted in the lowest life cycle impacts, whereas sequencing batch reactors and bio-filters performed environmentally worse.

In this analysis, we question whether seasonal suburban settlements might benefit from economies of scale by connecting to a neighboring WWTP or whether total or partial decentralization is a better option from an environmental standpoint. Here, the seasonal sanitation demand should be considered in terms of population increase and duration of the seasonal period. Our goal was to compare the environmental performance of a set of treatment scenarios in an insular context through LCA so that the sanitation needs of the permanent and seasonal residents are covered at the minimum environmental cost. To do so, we based our assessment on a case study located in Minorca (Balearic Islands, Spain), where improving the status of wastewater management is an urgent short-term objective (Goverment of the Balearic Islands 2016). Through a Monte Carlo simulation, we assessed the sensitivity and variability of the results according to some tourist-related parameters to determine the applicability in other tourist hubs.

9.2 Materials and Methods

5.2.1 Case study and scenario definition

The Balearic Islands are a Mediterranean archipelago hosting around 13.5 million tourists every year (IBESTAT, 2015). The island of Minorca has a medium size, being smaller than other Mediterranean islands, such as Majorca, Sicily or Sardinia, but with an area of 694.79 km^2 it welcomes 1.2 million visitors annually. Some areas have become mass tourism destinations. However, seasonal suburban settlements are also popular tourist hubs where the permanent population might range between 4 and 20% of the residents (Sanyé-Mengual et al. 2014).

In Minorca, Cala Morell (CM) is one of the settlements that experiences wastewater management issues because it is not connected to conventional sanitation systems. The main features of CM are shown in **Table 9.1**. Based on GIS data (ESRI, 2016), we identified 246 buildings, most of which were detached houses. Around 244 people reside in these houses all year, but 1,390 additional residents spend the summer in CM in either private detached houses or hotels. Because water consumption is higher in the summer (Domene and Saurí 2006), wastewater management is a concern for residents and local authorities, as it compromises the environmental, economic and social conditions of the region. In this sense, wastewater was treated at a small WWTP and an infiltration trench that offered basic treatment to the area but that was unable to meet the seasonal demand. Some buildings had private septic tanks, but it was complex to estimate the exact amount.

In this analysis, we assessed different alternatives that could potentially be implemented based on their environmental performance (**Figure 9.1**). We included different degrees of decentralization for treating permanent and/or seasonal wastewater flows, considering the technologies described in previous literature (for instance, Fuchs et al., 2011; Lehtoranta et al., 2014) and local priorities. Partial decentralization was included in scenarios SC1 and SC2. In scenario SC1, we analyzed the implementation of septic tanks in all buildings, with the consequent extraction and transportation of the pre-treated wastewater to a neighboring WWTP. SC2 was similar to the sanitation system used in CM at the time of the analysis, with part of the wastewater being treated at a local facility with basic equipment and later discharged into an infiltration trench. We assumed that this system served the permanent residents, whereas seasonal wastewater flows were treated through septic tanks. SC3 and SC4 were examples of decentralized sanitation. A constructed wetland replaced the neighboring WWTP in SC3, which implies the construction of a septic tank effluent pump (STEP) pressure sewer. SC4 excluded septic tanks, and instead the total wastewater was transported through a local sewer to the local WWTP for primary sedimentation and later connected to the constructed wetland, which was in the same place of the old infiltration trench. SC5 was a completely centralized scenario taht involved the connection of CM to the neighboring WWTP through an intercity sewer.

Table 9.1 Characterization of the seasonal suburban settlement

Variable	Value	Units	Source	Assumptions
Urban features				
Total area	0.65	km^2	ESRI (2016)	
Building area	0.05	km^2		
% building area	8	%		
Number of buildings	246	u		
Average building size	212	m^2		
Average household size	3.3	People	Domene and Saurí (2006)	In low density areas
Permanent population	244	People	Sanyé-Mengual et al. (2014)	
Peak population	1,634	People		
Permanent occupancy	15	%		
Potential wastewater generation			Domene and Saurí (2006)	Average data for low-density areas. Assuming water consumption equals wastewater production
Summer (per person and day)	254.2	L		
Winter (per person and day)	161.6	L		
Permanent population				
Population	244	People		
Number of buildings	74	u		Based on household size
% buildings	30	%		
Summer – daily wastewater	839	L/building		Based on daily generation
Winter – daily wastewater	533	L/building		Based on daily generation
Seasonal population				
Population	1,390	People		Peak minus permanent residents
Number of buildings	172	u		Total minus permanent buildings
% buildings	70	%		
Household size	8.1	People		Some of the buildings are hotels
Summer – daily wastewater	2,054	L/building		Based on daily generation
Tourist season	90	Days		From June to August

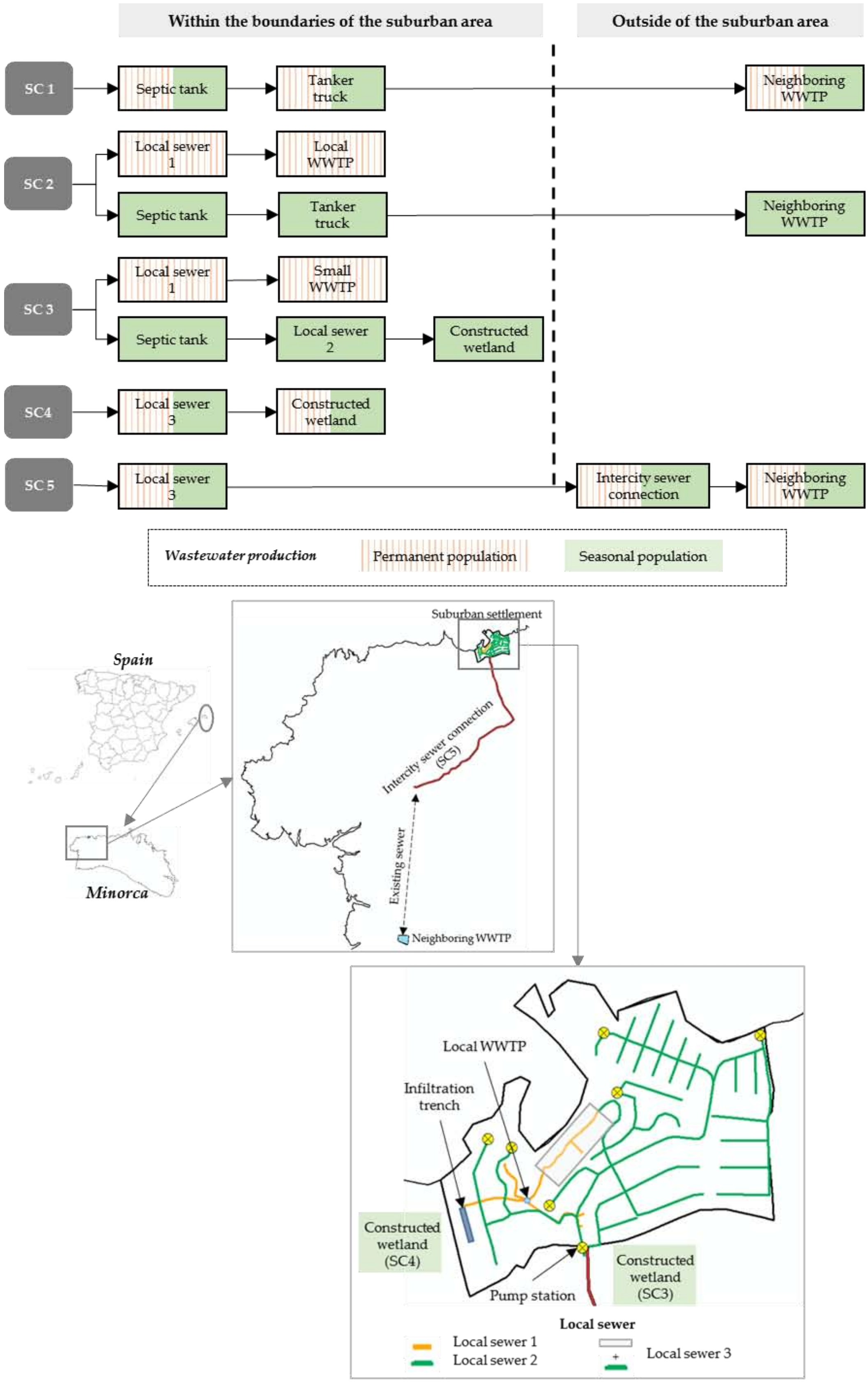


Figure 9.1 Wastewater treatment scenarios considered in the assessment and their distribution in the case study area

5.2.2 *Life cycle assessment of the treatment scenarios*

We conducted an LCA to evaluate the environmental performance of the five scenarios. In the following sections, we describe each of the steps as defined in ISO 14040:2006.

Goal and scope

To answer our research question, the functional unit of the analysis was one m^3 of wastewater produced every year by the permanent and seasonal residents of CM. Stormwater was not considered to reduce the effects of climate variability. A lifespan of 30 years was assigned to the infrastructure according to previous literature (Lehtoranta et al. 2014; Risch et al. 2015). The system boundaries included the raw material extraction, construction, operation and maintenance, and end of life. The detailed system boundaries of each scenario are shown in **Appendix 6.1**.

Life cycle inventory

The inventory of each treatment phase was elaborated based on literature and real data (see **Appendix 6.2** for details). Background life cycle data for materials and energy was retrieved from ecoinvent v3 (Weidema et al. 2013).

Influent and effluent quality: Domestic wastewater quality was estimated for total nitrogen (TN; 86.6 mg/L) and total phosphorus (TP; 16.6 mg/L) (Fuchs et al. 2011), which fall within the range defined by Kadlec and Wallace (2009). The quality of the effluent depended on the treatment technology. Septic tanks were not able to remove these components. In the local WWTP, we assumed that the infiltration trench might remove 30% and 10% of the TN and TP concentrations, respectively, thanks to soil microorganisms. Hu et al. (2007) reported a 50% removal, but we assigned more conservative values. In constructed wetlands, we considered 5 mg NH_4^+-N/L, 35 mg NO_3^--N/L, and 4 mg PO_3^{4-}-P/L (Søvik et al. 2006; Fuchs et al. 2011). The effluent of the WWTP, which was located in the city of Ciutadella, was based on average data supplied by the WWTP (>50,000 population equivalent) for the period from 2013 to 2016, i.e., 5.3 mg NH_4^+-N/L, 0.71 mg NO_2^--N/L, 15.5 mg NO_3^--N/L and 5.7 mg TP/L.

Septic tank components (SC1 and SC2): Septic tanks were sized according to the equations suggested by Crites and Tchobanoglous (1998) (see **Appendix 6.3**). A tanker truck extracted and transported wastewater and sludge to the neighboring WWTP (13 km) once a year for advanced treatment. The energy needed to pump out wastewater was estimated using similar data for sewer maintenance (Petit-Boix et al. 2015a). We calculated the sludge production through a carbon balance that accounted for the influent and effluent BOD (450 and 180 mg/L) (Kadlec and Wallace 2009) and CH_4 emissions from wastewater degradation (Diaz-Valbuena et al. 2011; US EPA 2016). CO_2 and N_2O were also included because of their global warming potential (Diaz-Valbuena et al. 2011).

STEP system + constructed wetland (SC3 and SC4): In the case of the STEP system linked to a community-based constructed wetland, wastewater was pumped out to the main sewer line an average of six times per day (Crites and Tchobanoglous 1998). The pressurized sewer included seven pump stations with two backup pumps each. We estimated the electricity consumption through the wastewater production and power of the system (1.5 kW) (Crites and Tchobanoglous 1998). Data on sewer and pump infrastructure were retrieved from Petit-Boix et al. (2014). Because septic tank effluent has a reduced amount of solids, the pipeline of a STEP system has a smaller diameter than that of SC4 and the trench is shallower (Crites and Tchobanoglous 1998).

In SC3, a horizontal subsurface flow wetland was hypothetically located in a green area that was close to the suburban settlement (**Figure 9.1**). In SC4, it replaced the infiltration trench and was connected to the local WWTP, where primary sedimentation took place. Wetland and material sizing were calculated based on a nitrogen effluent of 5 mg NH_4^+-N/L (Crites and Tchobanoglous 1998; Kadlec and Wallace 2009; Fuchs et al. 2011) (see **Appendix 6.3**). A six-cell wetland was sized according to the peak flow. We considered that the system remains in place after decommissioning. CH_4, CO_2, and N_2O emissions were adapted from Fuchs et al. (2011) and Søvik et al. (2006). Carbon sequestration was estimated considering the dry weight biomass production and carbon content of *Phragmites australis* (de Klein and van der Werf 2014).

Local and neighboring WWTP (SC2, SC3 and SC5): WWTP data on energy and chemicals were provided by facility managers. Waste and sludge production was estimated (Lorenzo-Toja et al. 2015), and although sludge was later used in agriculture, we did not consider the impacts of its application due to limited data availability. In SC2 and SC3, we measured the length of sewer using ArcGIS (ESRI, 2016), but in SC5 we knew the length projected by local authorities in their quest to improve the sanitation system. One of Ciutadella's WWTPs is located at a distance of 13 km from CM. 10 km of local sewer, 8 pump stations, and 6.4 km of intercity sewer were needed to connect CM with the WWTP. The intercity sewer connects CM's outlet with Ciutadella's first sewer stretch (**Figure 9.1**). The construction and demolition of the pavement was included in all scenarios with sewers. In SC4 and SC5, we assumed that existing septic tanks would be removed. Ciutadella's internal network was not included, as it already existed. The construction of CM's and Ciutadella's WWTP was not considered because it is usually negligible when compared to the operation (Termes-Rifé et al. 2013; Lorenzo-Toja et al. 2015). We assumed that treating the additional wastewater input from CM would not increase the resource needs of the WWTP due to economies of scale.

Life cycle impact assessment

We used Simapro 8 (PRé Consultants 2014) and the ReCiPe (H) method (Goedkoop et al. 2009) up to the characterization step. We present the 18 ReCiPe midpoints and the cumulative energy demand (CED) in **Appendix 6.4**, but the Results section specially focuses on global warming (GW, kg CO_2 eq), freshwater eutrophication (FE, kg P eq), marine eutrophication (ME, kg N eq), and CED (MJ). A single score was also calculated to obtain a single value (in millipoints, mPt) that weights the damage generated by each alternative (Goedkoop et al. 2009). FE and ME are relevant in this analysis due to the effluent emissions. To get a better insight of these two indicators and obtain comparable results for each scenario, we calculated the Eutrophication Net Environmental Impact (ENEI) elaborated by Lorenzo-Toja et al. (2016). This approach estimates the net impacts of the system by accounting for the discharge of untreated water, the indirect and direct impacts of the system, and the effluent limits set by water regulations (see **Equation 9.1**). For the latter, we considered the regulation of the Balearic Islands (Government of the Balearic Islands 2015), which limits the discharge of medium-sized WWTPs to 15 mg TN/L and 2 mg TP/L. Because the constructed wetland was in a low-populated area, we considered a permitted maximum discharge of 50 mg TN/L, according to the regulation.

$$\text{ENEI} = \frac{\text{EP}_{\text{DD}} - \text{EP}_{\text{I+R}}}{\text{EP}_{\text{DD}} - \text{EP}_{\text{L}}} \qquad \textbf{Equation 9.1}$$

Where ENEI: Eutrophication Net Environmental Impact; EP_{DD}: eutrophication of the direct wastewater discharge without treatment; EP_{I+R}: indirect process eutrophication plus the real effluent discharge; EP_{L}: eutrophication of the legal effluent discharge limits.

5.2.3 *Monte Carlo simulation*

The results of this analysis will vary depending on several modeling assumptions, as well as the location of the case study. We believe that the area we chose is representative of several islands with tourist hubs and the results might be generalized to other areas with some assumptions. For this reason, we conducted a Monte Carlo simulation through 15,000 iterations to account for a margin of safety using the @RISK software. We sought to identify the potential impact variability depending on a set of parameters. In a first screening, we modeled all variables included in **Table 9.1** and determined the ones with a greater influence on the results. In addition, the functional unit related to wastewater flows and tourist hubs are expected to differ in the type of tourism (i.e., long and short-term visitors) and accommodation type (e.g., hotels or houses). The distance to the WWTP might also influence the results, but this parameter was fixed in this first analysis. For these reasons, the second round of simulations was based on summer wastewater generation, duration of the tourist season, and peak population (**Table 9.2**). The values included were retrieved from studies that assessed the variability of each parameter in similar areas. A one-way ANOVA test determined whether results for each scenario were significantly different using the StatTools software. Additionally, we studied the sensitivity through tornado diagrams to identify the relationship between the results and each parameter.

Table 9.2 Parameters and data used in the Monte Carlo simulation

Parameters	Case study data	Input data		
		Probability function	Values	Sources
Summer wastewater generation (L/person/day)	254.2	Lognormal	Average: 254 Standard deviation: 183	Domene and Saurí, (2006)
Duration of the tourist season (days)	90	Triangular	Minimum: 3 Most probable: 25 Maximum: 60	Sanyé-Mengual et al. (2014)
Peak population (residents)	1,634	Triangular	Minimum: 1,500 Most probable: 3,500 Maximum: 5,500	Sanyé-Mengual et al. (2014)

9.3 Results and Discussion

In this section, we compare the environmental performance of the selected scenarios (**Section 9.3.1**) and assess the variability and sensitivity of the results based on modeling parameters (**Section 9.3.2**).

5.3.1 Environmental comparison of the treatment scenarios

The GW, CED, FE, ME and single score results of each alternative are shown in Figure 2. *A priori* it seems that SC5 scored better than scenarios SC1 to SC4, but the results are not conclusive when looking at the 18 midpoint indicators and CED (**Appendix 6.4**). On average, each alternative has the highest environmental impacts in 35% of the indicators. Based on **Figure 9.2**, one might argue that the effects of economies of scale are clear in all of the cases – including the single score - except for the CED, where SC4 doubles the impact of the best CED scenario (SC3, 37 MJ/m^3). The single score indicator suggests that the economies of scale of the large WWTP reduce the environmental burdens by 12% with respect to total decentralization through septic systems, or by 36% with respect to the constructed wetland.

Some parallelisms were found when compared to previous studies on treatment scales. For instance, Cornejo et al. (2016) reported that centralization can be beneficial in terms of GW, but that community-scale systems outperform in the eutrophication indicators. Although the former was also true in our analysis, with SC5 having half the impacts of the worst scenario (SC1, 5.1 kg CO_2 eq/m^3), decentralized systems were not the best in eutrophication. However, SC1 did have similar FE scores than SC5, because wastewater is transported to the neighboring WWTP. In this case, the treated effluent is discharged into seawater, which has a null FE potential (Goedkoop et al. 2009). The effects of the local WWTP on SC2 and SC3 are not irrelevant, as it generated up to 80% of the eutrophication. This was due to the groundwater emissions that resulted from a poor treatment capacity of the infiltration trench. The complete decentralization of the settlement might benefit from treating both the permanent and seasonal wastewater flows in the constructed wetland in terms of ME (i.e., SC4 instead of SC3), although the wetland might be bigger to accommodate this new demand. SC4 presented potential improvements in comparison with SC2 and SC3. In FE and ME, the impacts were reduced by 50% with respect to SC3.

GW and CED also presented interesting results. In scenarios SC1 to SC3, septic tanks have a large contribution to GW (up to 50%) because of the intensive use of concrete and the methane emissions generated during the operation phase. The impacts of the tanker truck are also notable (40% of GW and 70% of CED), given the volume of water that must be transported to the WWTP. Similar results were found when analyzing the supply of decentralized services through road transportation (Oliver-Solà et al. 2009; Righi et al. 2013). The role of sewers is apparent in SC4 and SC5 in the CED indicator, where they represent around 50% of the impacts. In SC5, additional impacts should be considered, as the reduced wastewater flow might result in sludge accumulation and pressure loss, which might increase the maintenance needs or replacement of the infrastructure.

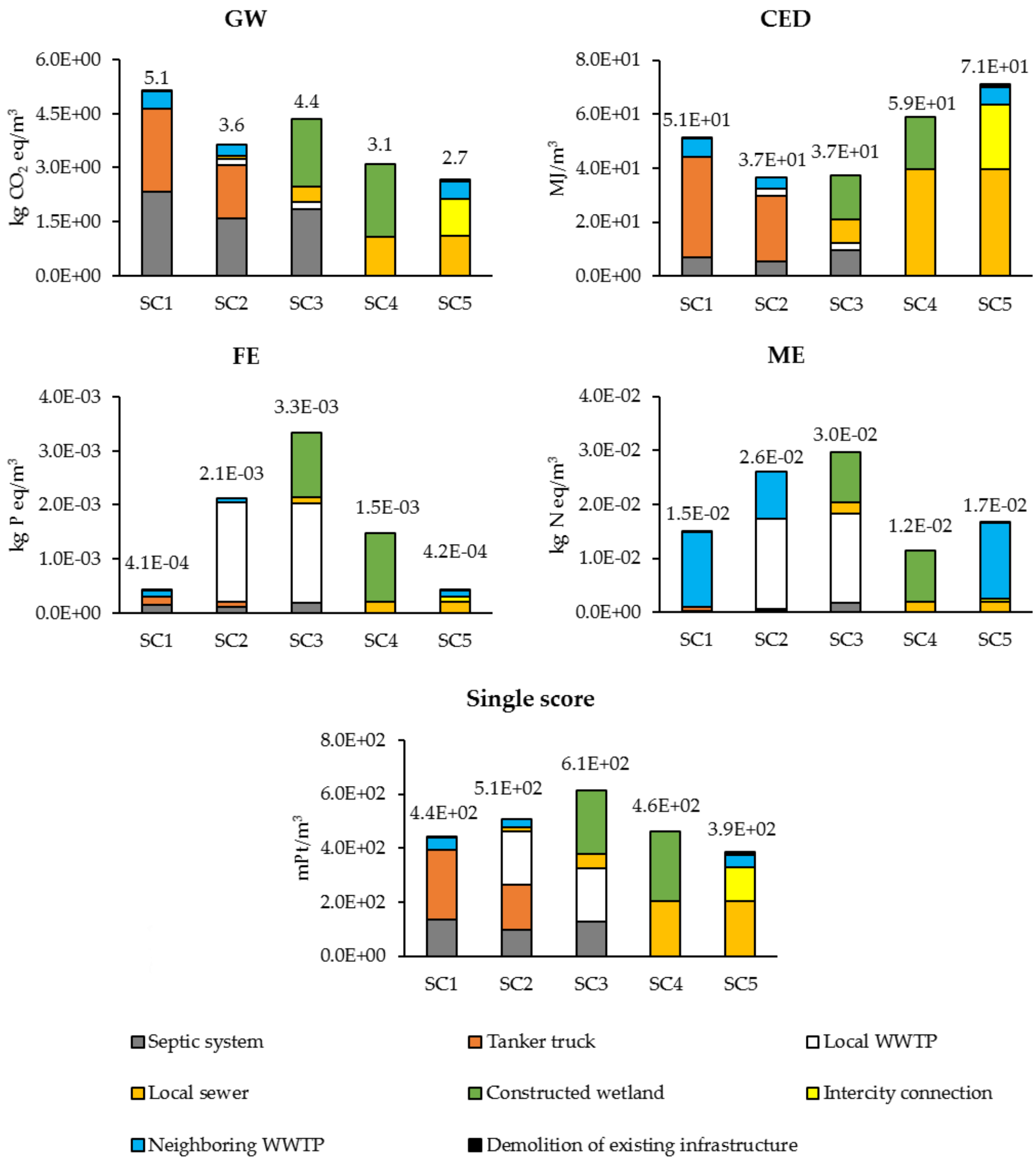


Figure 9.2 Comparison of the treatment scenarios in the CM context and contribution of each of the stages involved. Data refer to the functional unit (one cubic meter).

Because of the relevance of the eutrophication indicators in wastewater treatment (Gallego et al. 2008; Hospido et al. 2004), **Table 9.3** provides an estimation of the ENEI indicator for a better scenario comparison. For a constant TP discharge limit, larger ENEI values indicate an improved environmental performance with respect to a do-nothing scenario. For this reason, SC1 and SC5 have a greater ENEI (1.1 kg P eq/m^3) as opposed to SC2-SC5, as the latter have greater indirect life cycle impacts. In the case of TN, we differentiated the location of the wetland, which enables the discharge of larger TN concentrations (50 mg/L). In SC2 we accounted for the most restrictive scenario (15 mg TN/L), as part of the wastewater is treated at the neighboring WWTP. SC4 appeared to have the best ENEI, but because the discharge limit is not constant in all the cases, it is complex to compare the scenarios with one another. In the case of TP, ENEI validates the eutrophication comparison obtained through the LCA (Figure 2). In any case, we did not obtain ENEI < 0, which indicates that treating wastewater is always better than the direct discharge.

Table 9.3 Estimated ENEI indicators for each scenario

Eutrophication indicators		Wastewater management scenarios				
		SC1	SC2	SC3	SC4	SC5
Freshwater eutrophication (kg P eq/m^3)	Indirect process impacts + real effluent discharge	4.1E-04	2.1E-03	3.3E-03	1.5E-03	4.2E-04
	Direct wastewater discharge (no treatment)			5.5E-03		
	Legal effluent discharge limits			6.6E-04		
	Eutrophication Net Environmental Impact (ENEI)	1.1E+00	7.0E-01	4.4E-01	8.2E-01	1.1E+00
Marine eutrophication (kg N eq/m^3)	Indirect process impacts + real effluent discharge	1.5E-02	2.6E-02	3.0E-02	1.2E-02	1.5E-02
	Direct wastewater discharge (no treatment)			9.6E-02		
	Legal effluent discharge limits	1.2E-02	1.2E-02	3.9E-02	3.9E-02	1.2E-02
	Eutrophication Net Environmental Impact (ENEI)	9.6E-01	8.3E-01	1.3E+00	2.0E+00	1.0E+00

5.3.2 Variability and sensitivity to modeling parameters

Through the probabilistic analysis applied to peak population, summer wastewater generation, and duration of the tourist season (**Table 9.2**), we obtained the single score distributions shown in **Figure 9.3** (statistical values for all impact categories are provided in **Appendix 6.5**). Again, the worst score was related to SC3, with an average of 950 mPt. In general, there is a certain range of variability, but the highest was registered in SC5, where 90% of the values are expected to range between 250 and 1,100 mPt (77% difference). These variations implied that SC2 and SC5 are not significantly different based on the ANOVA test, although there was a 24% difference in the case study results (**Figure 9.2**). SC1 was the best scenario with 520 mPt, with a 13% distance ahead from SC2 and SC5.

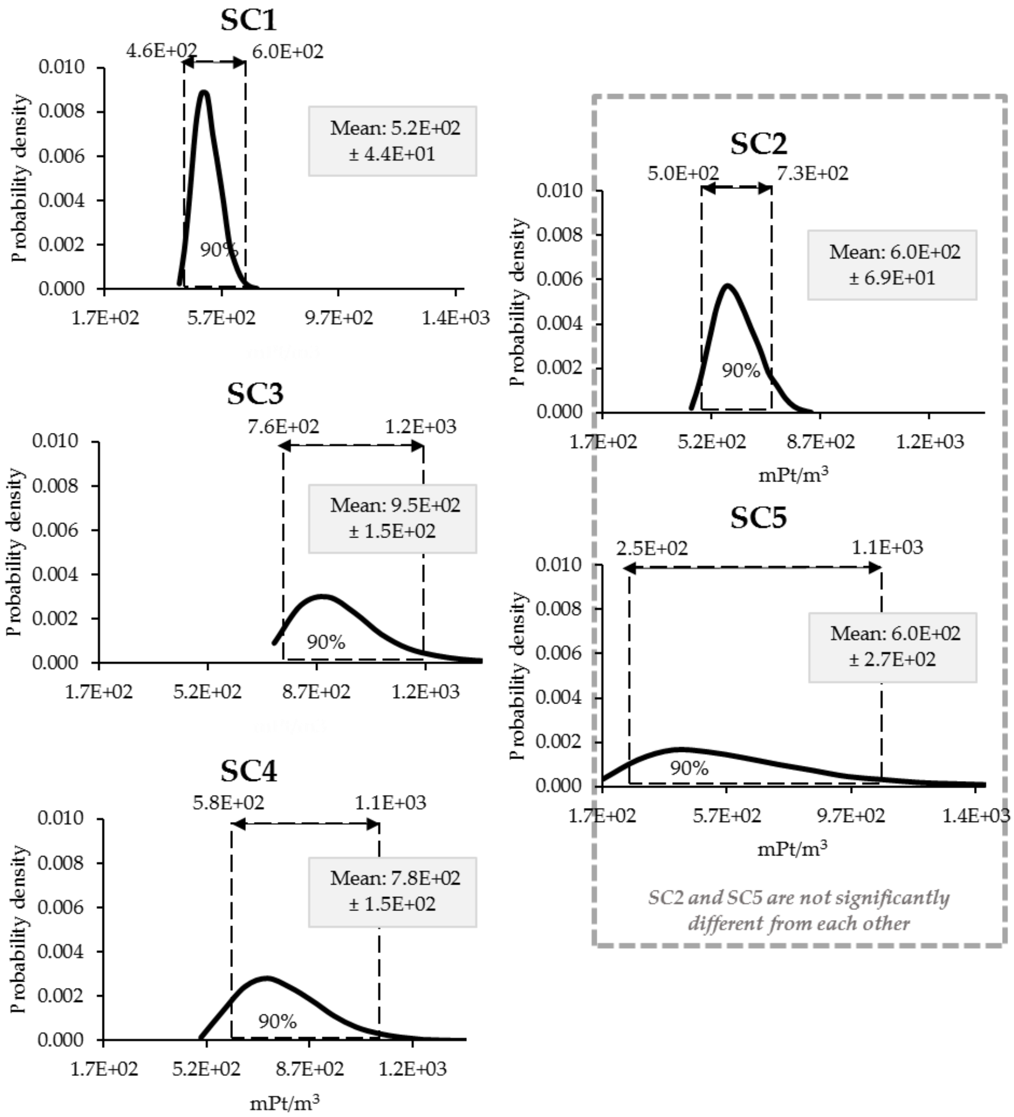


Figure 9.3 Variability in the eutrophication impacts of the scenarios' life cycle based on a Monte Carlo simulation. The probability density functions are presented for the single score indicator. All the results are significantly different at a confidence level of 95% and p-value <0.0001, except for the pair highlighted in the graph.

To understand the causes of this variability, **Figure 9.4** displays the sensitivity of the single score indicator the local parameters. In general, the results were particularly sensitive to changes in the duration of the tourist season, with the impacts being up to 23% lower and 34% higher than the average data. This might set a difference between different types of tourist areas. The highest impacts might be related to either mass tourism destinations, where hotels are the most popular accommodation option and there is a continuous tourist flow during the summer days, or second-home owners who spend the entire summer in a private property. The lowest impacts might be assigned to areas that are more rural with occasional visitors.

Wastewater generation and population seemed to have a lower effect except for SC5. Here there was a large sensitivity (up to 82% variation in the results) due to the energy required in the sewer for pumping wastewater. Depending on the number of residents, the size of the sewer might also vary, but the energy consumption will be different based on the urban form and distance to the neighboring WWTP (Petit-Boix et al. 2015a). Because these parameters were fixed in this analysis, variations in wastewater generation are the main item that increases the impacts.

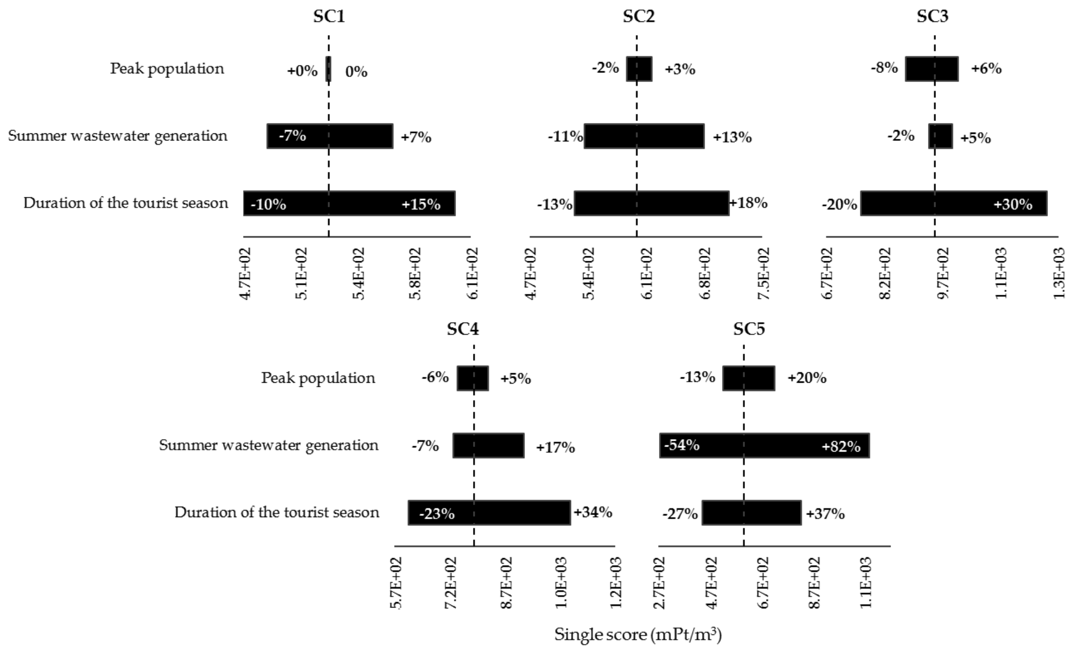


Figure 9.4 Effect on the single score outputs of each parameter modeled displayed using tornado diagrams. The broken line represents the average value of each scenario and indicator.

9.4 Implications of this analysis

In the framework of sustainable tourism goals, reducing the environmental impacts of wastewater sanitation in islands should be a priority. Previous analyses focused on sustainable transportation and user behavior in islands (Sanyé-Mengual et al. 2014), but water infrastructure still needs to be assessed in more detail. In this analysis, we integrated the environmental dimension into the assessment of the centralization versus decentralization debate. Eggimann et al. (2015) pointed the lack of geographically explicit analyses when assessing these alternatives and presented and optimization model based on costs. In our case, we discuss the application of alternatives from an environmental standpoint. This approach could be applied to many other suburban settlements, not only in the Mediterranean, but also in other islands around the world that face water issues. The type of tourism and duration of the seasonal period might condition the results, as well as the wastewater generation and consumption.

Our weighted case-study results point to the benefits of centralizing wastewater treatment in seasonal suburban settlements. However, when generalizing to other insular contexts, results are not conclusive, although it seems that decentralization might be beneficial in some cases. However, eutrophication indicators are essential in these assessments, and it is a fact that Minorca's aquifers are highly polluted and discharge P and N to the sea (OBSAM 2015; Garcia-Solsona et al. 2010). An optimized wetland system that treats the entire demand without the support of a local WWTP can be a promising decentralized alternative, but groundwater discharge is still an issue that should be monitored. In this sense, partial decentralization through septic tanks and treatment at a neighboring WWTP might be another solution that produces the same ENEI as complete decentralization. However, this might compromise other environmental indicators, and the single score depends on the importance given to each damage category. In this sense, the relevance of each indicator in islands might be a field to explore.

Part V

Environmental and economic flood damage assessment in water-sensitive urban areas

Chapter 10

Picture: Flooding alert sign in Cambrils' ephemeral stream (Tarragona, Spain)

Are we preventing flood damage eco-efficiently? An integrated method applied to post-disaster emergency actions in the Mediterranean

Chapter 10 Are we preventing flood damage eco-efficiently? An integrated method applied to post-disaster emergency actions in the Mediterranean

This chapter is based on the following paper:

Petit-Boix, A., Arahuetes, A., Josa, A., Rieradevall, J., Gabarrell, X., 2017. Are we preventing flood damage eco-efficiently? An integrated method applied to post-disaster emergency actions. Science of the Total Environment 580, 873-881.

doi: 10.1016/j.scitotenv.2016.12.034

DDD link: http://ddd.uab.cat/record/169388

Abstract

Flood damage results in economic and environmental losses in the society, but flood prevention also entails an initial investment in infrastructure. This study presents an integrated eco-efficiency approach for assessing flood prevention and avoided damage. We focused on ephemeral streams in the Maresme region (Catalonia, Spain), which is an urbanized area affected by damaging torrential events. Our goal was to determine the feasibility of post-disaster emergency actions implemented after a major event through an integrated hydrologic, environmental and economic approach. Life cycle assessment (LCA) and costing (LCC) were used to determine the eco-efficiency of these actions, and their net impact and payback were calculated by integrating avoided flood damage. Results showed that the actions effectively reduced damage generation when compared to the water flows and rainfall intensities registered. The eco-efficiency of the emergency actions resulted in 1.2 kg CO_2 eq per invested euro. When integrating the avoided damage into the initial investment, negative net impacts were obtained (e.g., -5.2E+05 € and -2.9E+04 kg CO_2 eq per event), which suggests that these interventions contributed with environmental and economic benefits to the society. The economic investment was recovered in two years, whereas the design could be improved to reduce their environmental footprint, which is recovered in 25 years. Our method and results highlight the effects of integrating the environmental and economic consequences of decisions at an urban scale and might help the administration and insurance companies in the design of prevention plans and climate change adaptation.

Keywords: risk management, life cycle assessment, life cycle costing, damage prevention, climate change

Chapter 11

Picture: Carolyn Edwards Memorial Rain Garden, The University of Toledo (Ohio, USA)

Environmental and economic assessment of a stormwater infiltration system for flood prevention in Brazil

Chapter 11 Environmental and economic assessment of a stormwater infiltration system for flood prevention in Brazil

This chapter is based on the following paper:

Petit-Boix, A., Sevigné-Itoiz, E., Rojas-Gutierrez, L.A., Barbassa, A.P., Josa, A., Rieradevall, J., Gabarrell, X., 2015. Environmental and economic assessment of a pilot stormwater infiltration system for flood prevention in Brazil. Ecological Engineering 84, 194-201.

doi: 10.1016/j.ecoleng.2015.09.010

DDD link: http://ddd.uab.cat/record/145553

Abstract

Green and gray stormwater management infrastructures, such as the filter, swale and infiltration trench (FST), can be used to prevent flooding events. The aim of this study was to determine the environmental and economic impacts of a pilot FST that was built in São Carlos (Brazil) using life cycle assessment (LCA) and life cycle costing (LCC). As a result, the components with the greatest contributions to the total impacts of the FST were the infiltration trench and the grass cover. The system has a carbon footprint of 0.13 kg CO_2 eq/m^3 of infiltrated stormwater and an eco-efficiency ratio of 0.35 kg CO_2 eq/USD. Moreover, the FST prevented up to 95% of the runoff in the area. Compared to a gray infrastructure, this system is a good solution with respect to PVC stormwater pipes, which require a long pipe length (1070 m) and have a shorter lifespan. In contrast, concrete pipes are a better solution, and their impacts are similar to those of the FST. Finally, a sensitivity analysis was conducted to assess the changes in the impacts with the varying lifespan of the system components. Thus, the proper management of the FST can reduce the economic and environmental impacts of the system by increasing its durability.

Keywords: life cycle assessment, life cycle costing, Best Management Practices, climate change adaptation, urban flood, filter, swale and infiltration trench

Chapter 12

Picture: Artificial swale, The University of Toledo (Ohio, USA)

Floods and consequential life cycle assessment: integrating flood damage into the environmental assessment of stormwater Best Management Practices

Chapter 12 Floods and consequential life cycle assessment: integrating flood damage into the environmental assessment of stormwater Best Management Practices

This chapter had the following collaborators:

Petit-Boix, A., Sevigné-Itoiz, E., Rojas-Gutierrez, L.A., Barbassa, A.P., Josa, A., Rieradevall, J., Gabarrell, X.

Abstract

Stormwater management is essential to reducing the occurrence of flooding events in urban areas and to adapting to climate change. The construction of stormwater Best Management Practices (BMPs) entails a series of life cycle environmental impacts but also implies avoided burdens, such as replacing urban infrastructure after flooding. The aim of this paper is to integrate flood damage prevention into the life cycle assessment (LCA) of BMPs for quantifying their net environmental impact (NEI) and environmental payback (EP) from a consequential LCA standpoint. As a case study, the application of a filter, swale and infiltration trench (FST) in a Brazilian neighborhood was assessed considering a high-intensity rainfall event. The potential avoided impacts were related to cars and sidewalks that were not destroyed due to flooding. In terms of CO_2 eq emissions, the environmental investment related to the FST was recovered when the destruction of one car or 84 m^2 of sidewalk was prevented. The NEI of the FSTs resulted in significant impact reductions (up to 700%) with respect to not accounting for the avoided products. This approach can be implemented to any type of BMP, and more accurate estimations can be made with data for different events and different types of material damage.

Keywords: flood, Best Management Practices, consequential LCA, urban infrastructure, water, city

12.1 Introduction

Flood risk management is crucial in several parts of the world. From 1960 to 2014, flooding events accounted for 34% of the natural disasters registered worldwide (17 floods/year), representing more than 2.5 billion USD/year in terms of monetary damage and 1,254 deaths/year (Guha-Sapir et al. 2009). Currently, 53% of the world's population lives in urban areas (The World Bank 2016), a figure that is expected to increase to 70% by 2020 (UN, 2012), implying an increasing demand for new construction and infrastructure, and decreasing the proportion of permeable areas and sometimes improper land-use planning (Jha et al. 2012). Combined with a forecasted increase in the precipitation intensity, this increase would result in a greater risk of potential flooding events, especially in mid and high latitudes (Meehl et al. 2007). Most of the major flooding events that have been recorded occur in highly populated urban areas (Jha et al. 2011).

Therefore, there is a need for proper stormwater management practices in cities to reduce the occurrence and consequences of these events. To do so, stormwater Best Management Practices (BMPs) are commonly used and at present are usually classified into gray and green infrastructure BMPs. Gray infrastructure BMPs are traditional drainage strategies (e.g., sewers or detention tanks), whereas green infrastructure BMPs provide ecosystem services, such as aquifer recharge or environmental restoration (European Commission 2013). Examples of green BMPs include decentralized systems, such as green roofs, permeable pavements, bio-retention basins and rainwater harvesting systems, among others.

In the context of Low Impact Development (LID), these techniques are based on the premise that stormwater management should not be envisioned as stormwater disposal. Implementing source-control BMPs might result in a reduction of the flooding risk, given that these systems can effectively reduce the runoff volume (Lee et al. 2013; Mentens et al. 2006; Zahmatkesh et al. 2015). Another benefit is the effect on the runoff quality. Especially green infrastructure BMPs have the potential to retain and filtrate runoff pollutants, such as heavy metals, hydrocarbons, nutrients or suspended solids (Deletic and Fletcher, 2006; Dierkes et al., 2002; Llopart-Mascaró et al., 2015). This performance depends on the type of drainage system, soil characteristics and slope (Czemiel Berndtsson 2010), among other factors.

However, the construction of these infrastructures also entails a series of environmental impacts, such as GHG emissions or resource depletion. To address this issue, life cycle assessment (LCA) offers a method for calculating and discussing the environmental burdens that are associated with the life cycle stages of a product or system, i.e., from raw material extraction to end-of-life (ISO 14040:2006). Several analyses have been conducted to calculate the life cycle impacts of different BMPs, such as green roofs (Saiz et al. 2006; Kosareo and Ries 2007), bio-infiltration basins (Flynn and Traver 2013), constructed wetlands (Risch et al. 2014), rainwater harvesting systems (Angrill et al. 2016; Devkota et al. 2015) or a combination of BMPs (De Sousa et al. 2012). These analyses mainly focus on determining the contribution of the materials and energy to the total environmental impacts of the system (i.e., attributional LCA; ALCA), as well as potential positive effects, such as carbon sequestration.

The construction of BMPs has broader consequences on the market and the society that could be assessed using consequential LCA (CLCA) (the relationship between ALCA and CLCA is presented in **Section 12.1**). Given that a reduction in the runoff volume results in less water being treated, some studies assessed the avoided impacts of wastewater treatment plants (WWTP) (Wang and Zimmerman 2015; Spatari et al. 2011; Catalano De Sousa et al. 2011). In another study, Wang et al. (2013) calculated the economic and environmental impacts related to reducing the eutrophication for different BMP combinations, such as bio-retention basins or separate stormwater networks. Nevertheless, the lack of studies that include CLCA in the assessment of BMPs hinders the effects of BMP implementation at an urban scale and from a life cycle perspective.

Implementing a certain type of BMP for preventing urban flooding might also prevent the destruction of goods that would otherwise need to be replaced. In such scenario, the impacts associated with the production of the affected goods would be avoided (e.g., urban infrastructure, buildings, etc.). However, given that a previous environmental investment was made to produce the BMP, the net balance between the impacts and benefits should be determined. This approach was already applied considering the net environmental impact (NEI) as the difference between the avoided and induced eutrophication potential in WWTPs (Lorenzo-Toja et al. 2015), and in the damage evolution resulting from post-disaster emergency actions implemented in ephemeral streams after flooding (Petit-Boix et al. 2017).

Additionally, the NEI might vary depending on the type of BMP and the area of application. This analysis is especially interesting in flood-sensitive countries. A study area could be Brazil, where there are great variations between social strata and urban densities, a fact that might influence the space that is available for constructing a BMP. In this country, floods account for 60% of the total natural disasters and an average of 200 million USD in terms of damage (Guha-Sapir et al. 2009). Additionally, the number of flooding events has generally increased over time (Guha-Sapir et al. 2009) and may increase in the future because of climate change. Given a certain urban density, the implementation of a BMP could be analyzed in a Brazilian neighborhood. In this paper, the BMP that was selected for a first case study analysis was a filter, swale and infiltration trench (FST) that had been previously constructed in a Brazilian city for experimental purposes. The specific design features and storage capacity of this BMP were thoroughly analyzed in previous studies (Lucas 2011; Lucas et al. 2013), and it was thus applied to this first assessment of flood damage prevention.

Our goal was to integrate flood damage into the LCA of stormwater Best Management Practices (BMP) for quantifying their net environmental impact (NEI) and environmental payback (EP) from a consequential LCA (CLCA) perspective. To do so, we based this analysis on a case study BMP. The specific objectives were (1) to define the steps involved in the calculation of the EP and NEI of a BMP; (2) to calculate the EP and NEI of an FST with respect to the material losses, considering the implementation of this system in Brazil; (3) to discuss the environmental implications of this approach in the field of flood prevention.

12.2 Materials and Methods

12.2.1 Consequential life cycle assessment (CLCA)

Consequential modeling is defined as "a system modeling approach in which activities in a product system are linked so that activities are included in the product system to the extent that they are expected to change as a consequence of a change in demand for the functional unit" (Sonnemann and Vigon 2011). In this sense, CLCA offers a causal explanation for the consequences of future actions (Weidema et al. 2009) and attempts to forecast in the short or long term the environmental impacts of decisions that are made in the present (e.g., implementing a BMP). This broader analysis is not provided by ALCA, which assesses the environmental status of a product system as an account of the history of the product (ISO 14040:2006), and does not include the processes that are outside of the product's immediate system boundaries (Earles and Halog 2011). The ALCA approach analyzes the entire life cycle of a selected system but does not account for the processes that are directly or indirectly affected by the existence of this system. In contrast, the CLCA approach integrates market mechanisms into environmental modeling, which is often associated with inconsistency because LCA practitioners deal with different assumptions in their studies, such as the definition of the system boundaries (Zamagni et al. 2012). However, CLCA might help to determine certain beneficial outcomes related to the system under analysis, and to inform policy-makers on the indirect environmental implications of a system (Vázquez-Rowe et al. 2014; Sanchez et al. 2012).

The consequences of flooding vary greatly depending on their type (e.g., flash floods, coastal floods, and urban floods), location and extent of flooding, and vulnerability and value of the natural and constructed environment. For example, in the same area flooding events might lead to different consequences over time because of the types of built infrastructure (e.g., conventional or prevention-oriented). The impacts of flooding may include damage to properties (e.g., buildings, vehicles, roads, and public utilities), the environment (e.g., livestock or crops) and the society (e.g., casualties, damage to cultural heritage, etc.). From a CLCA system perspective, modeling should include all of the processes that are expected to be affected by a decision, regardless of whether they are part of the existing supply chain or not (Wolf and Ekvall 2011). In this study, the decision of implementing BMPs for flood prevention may contribute to diminish its effects or damage. Modeling these consequences may become a very complicated and uncertain task due to the variety and unpredictability of flooding consequences and the effect of flood protection infrastructures. This is not the purpose of this analysis. This study is an attempt to highlight the potential environmental benefits related to flood prevention by including possible processes affected beyond the system evaluated (i.e., BMPs) in accordance with CLCA approaches. Our study aspires to contribute to the flood prevention and planning debate, as strategies that decrease damage might result in important environmental savings that are currently not captured.

In our study, we assessed the effects of implementing a BMP on the market by considering a decrease in the demand for two properties (i.e., cars and sidewalks) which are generally damaged during flooding events. Both products could be representative of possible damages resulting from any type of flood and region. In this case, it is interesting to determine the trade-offs between the construction of a BMP and the material losses, assuming that these would be reconstructed after an event (**Figure 12.1**).

12.2.2 *Steps for calculating the environmental payback (EP) and net environmental impact (NEI) of a BMP*

This section describes the steps considered for estimating the EP and NEI of implementing a BMP for flood prevention (**Figure 12.1**).

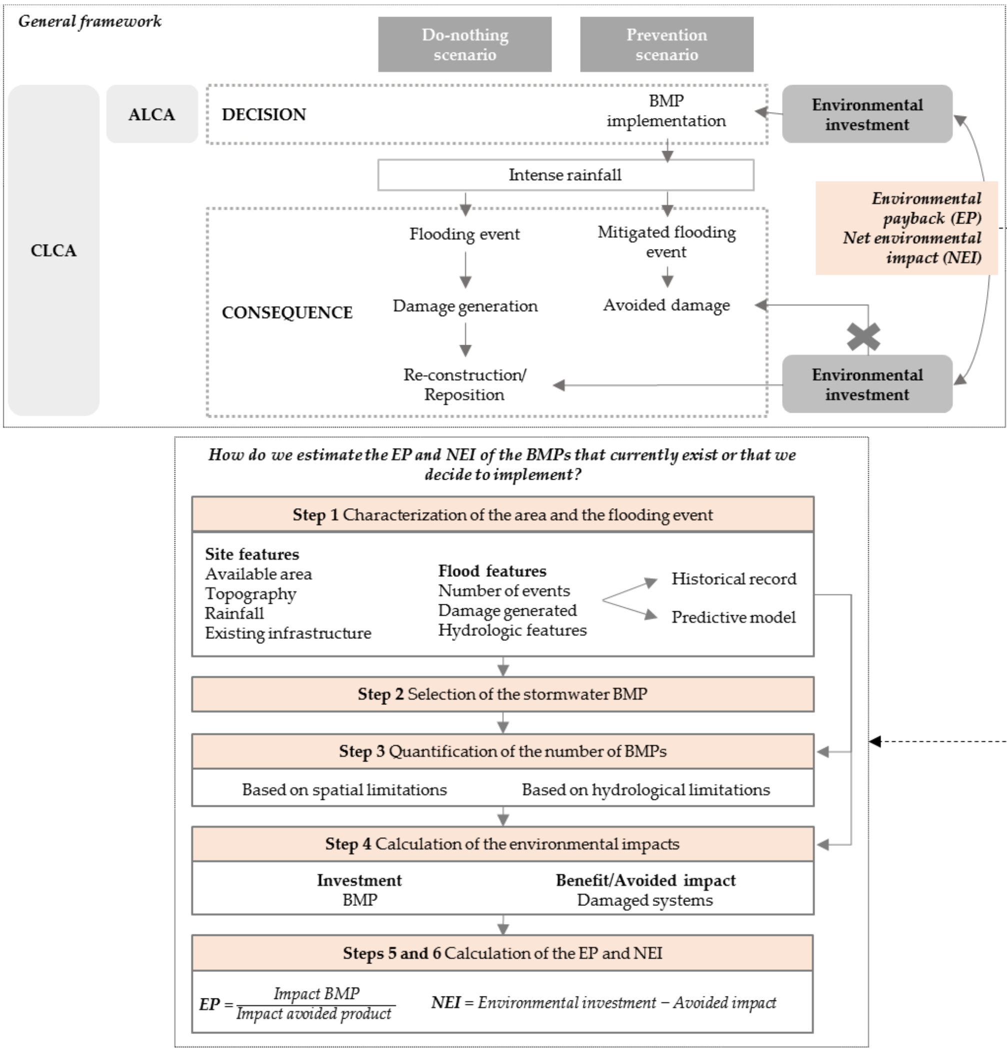


Figure 12.1 Integration of the consequences of implementing BMPs for flood prevention into the CLCA framework and steps for calculating the EP and NEI

Step 1 Characterization of the area and the flooding event: The main features of the study area must be collected, e.g., total area, topography, superficial and underground installations, building types and average rainfall patterns. Depending on the analyzed time period, data on a particular flooding event or a series of flooding events producing damage are needed. These data include hydrological indicators, such as rainfall intensity, duration and frequency; concentration time, water level height of rivers and reservoirs; and material losses, such as the number of cars, area of sidewalks or number of affected buildings (i.e., building structure and contents).

Step 2 Selection of the stormwater BMP: The selection of a suitable BMP for a specific area depends on different parameters, such as the features of the study area (Step 1), space and infiltration requirements, need for pollutant removal, land use, safety and maintenance needs (UDFCD, 2011).

Step 3 Quantification of the number of BMPs: The number of BMPs required to prevent the flooding events defined in Step 1 can be estimated as (1) the number of BMPs that could collect the total volume of stormwater, or (2) the number of BMPs that could be implemented in the area according to the space requirements. Based on hydrological parameters and a similar stormwater storage potential, the option that results in fewer BMPs is recommended. When a combination of different types of BMPs is implemented, the number of systems could be estimated through option (1), and a series of scenarios could be assessed with a varying amount of each type of BMP.

Step 4 Calculation of the environmental impacts of the BMP and avoided products: The LCA of the BMP and the avoided products is conducted based on the steps defined by ISO 14040:2006 (i.e., goal and scope definition, inventory analysis, impact assessment and interpretation). It should consider the site-specific features of the products, such as design, electricity mix, lifespan, etc.

Given that the system boundaries of this study include different elements, defining a functional unit (FU) is not straightforward (Zamagni et al. 2012). In this case, the FU could be the implementation of one or a set of BMPs, with a certain lifespan, for preventing a flooding event and its associated material damage, given a rainfall intensity of *X* mm/h. If the environmental impacts of the avoided damage are assessed from an LCA standpoint, the particular impacts of each element can be related to the production of a unit of urban infrastructure (avoided product), whose destruction can be prevented by implementing a BMP. The FU should be adapted to the period of analysis, number of flooding events and rainfall intensity assessed, because the prevented damage is not a fixed parameter, but varies depending on the event(s) and features defined in Step 1.

Step 5 Calculation of the EP: After conducting the LCA, it is interesting to know when the environmental investment made during the implementation of the BMP might be recovered. To do so, the ratio between the impacts of one BMP and one unit of avoided product can be calculated. The results show the product units that should be prevented through the lifespan of the BMP to recover the environmental investment.

Step 6 Calculation of the NEI: The net balance per flooding event can be determined applying **Equation 12.1**. For comparing different types of BMP, the total NEI per event could be expressed in terms of NEI per cubic meter of stormwater, for instance.

$$\frac{\text{NEI}}{\text{event}} = \text{A} \times \frac{\text{Impact}_{\text{BMP}}}{\text{L} \times \text{F}} - \sum_{i=1}^{n} (\text{X}_i \times \text{Impact}_i) \qquad \textbf{Equation 12.1}$$

where NEI: net environmental impact for a given impact indicator; A: number of BMPs implemented with a certain storage capacity; $\text{Impact}_{\text{BMP}}$: environmental impacts of the BMP under assessment; L: lifespan of the BMP (years); F: average annual flooding events; i: type of avoided product (e.g., car, sidewalk, building, etc.); X: units of avoided product per event; and Impact_i: environmental impact of the avoided product.

12.2.3 Case study: implementation of an FST in São Carlos (Brazil)

The steps presented in **Section 12.2.2** were applied to estimate the EP and NEI of a BMP in São Carlos (Brazil).

Step 1 Characterization of the area and flooding event

São Carlos is located in the State of São Paulo and presents a humid Subtropical climate with an annual average rainfall of 1500 mm (INMET, 2014). On average, it rained 111 days every year from 2005 to 2014, and the average daily rainfall during these events was 13.4 mm (INMET, 2014), with a maximum in 118 mm (**Appendix 9.1**). São Carlos experiences an average of three heavy rainfall events every year that result in a flood and damage to the environment (**Table 12.1**). The built environment is typically affected by the destruction of roads, sidewalks, roofs, walls, public utilities and vehicles. There are also records of injuries, deaths and destruction of homes after flooding (IPMet, 2014).

Table 12.1 Heavy rainfall events registered in São Carlos and their consequences on the natural and urban environment. Data from INMET (2014) and IPMet (2014)

Year	Rainfall per event (mm)	Estimated duration (min)	Damage to:			
			Natural environment	Buildings/public infrastructures	Vehicles	People
16-Mar-2005	20-64	40-120				
28-Oct-2005						
16-Dec-2005						
21-Dec-2005						
14-Feb-2006	10	60				
09-Jan-2009	10-66	~60				
26-Oct-2009						
17-Dec-2009						
07-Oct-2010	12	nd				
31-Jan-2011	33	45				
27-Feb-2012	16-54	10-15				
05-Jun-2012						
23-Oct-2012						
09-Nov-2012						
09-Dec-2012						
22-Feb-2013	6-89	15-480				
09-Mar-2013						
28-May-2013						
22-Oct-2013						
04-Nov-2013						
30-Nov-2013						
05-Dec-2013						
14-Jan-2014	36-37	~120				
02-Sep-2014						
Number of events that caused damage			**23**	**19**	**5**	**7**

For the purpose of this assessment, we applied data from one of the flooding events presented in **Table 12.1**. To assess an extreme event with high-intensity rainfall (>30 mm/h - FLOODsite (2008)) that produced damage, data from October 22, 2013 were analyzed. During this event, 50 mm of rainfall were recorded in 15 min (200 mm/h), resulting in the destruction of six cars and 50 sidewalk stretches with an average area of 3 m^2 each (IPMet, 2014). These data were retrieved from the Natural Disaster Database of the Institute of Meteorological Research (IPMet, 2014), which is responsible for recording disaster damage in Brazil. Given the difficulties in data gathering, very little information exists on the volume of losses, although the type of damage is reported. As a result, the estimation of the NEI and EP only focused on this specific flooding event, for which data were available, with the purpose of exemplifying the application of this approach.

Step 2 Selection of the stormwater BMP

To reduce the number of floods and their consequences, a pilot green stormwater BMP was built in 2009 on the campus of the Federal University of São Carlos (UFSCar). In this case, the BMP was an FST that occupied 600 m^2 and had a maximum storage capacity of 110 m^3, considering the saturation of the subsurface soil layer (for additional information, see Lucas (2011), Lucas et al. (2013) and Petit-Boix et al. (2015)). The technical requirements of the infiltration trenches are described in Baptista et al. (2005). As technical and design data of this system exist, we evaluated the hypothetical construction of a set of FSTs in a neighborhood. São Carlos has high- and low-density neighborhoods, and in this case, we selected the residential neighborhood Damha I (RDI) because (1) it could benefit from decentralized, green stormwater management as an alternative to combined sewers, and (2) this area can host FSTs given the space requirements, as it is a low-density neighborhood. RDI covers an area of 420,000 m^2 and consists of single-house blocks with green plots that could potentially host an FST.

Step 3 Quantification of the number of BMPs

Assuming a homogeneous rainfall distribution in São Carlos, RDI would have received 21,000 m^3 of stormwater during the selected flooding event. Considering the maximum storage capacity of the FST (110 m^3), RDI would require 190 FSTs. Nevertheless, there were a total of 107,000 m^2 worth of plots that could be used to infiltrate stormwater (Google Maps 2015) considering gardens, abandoned green areas and plots. Given the minimum size of an FST (600 m^2) and the dimensions of each individual plot (plots smaller than 600 m^2 were excluded), set up resulted in the potential to construct 179 FSTs, which is more than 90% of the requirements of the area for that high-intensity rainfall event. In this case, 179 FSTs were accounted for in the analysis considering a homogeneous soil composition, and we assumed that they would prevent the material damage in the area.

Steps 4-6 Calculation of the environmental impacts, EP and NEI

This step is based on the LCA methodology and is divided into the steps described in ISO 14040:2006.

Goal and scope definition: Our specific goal was to determine the NEI and EP of implementing FSTs in RDI for flood prevention purposes. The FU considered was the implementation of 179 FSTs with a lifespan of 10 years for preventing a flooding event and its associated damage given a rainfall intensity of 200 mm/h in a low-density neighborhood in São Carlos. We calculated the environmental impacts of producing one car and one m² of sidewalk (0.2×1×1 m), whose destruction is prevented by the implementation of the FSTs. For the purpose of the assessment, six cars and 150 m² of sidewalk were considered given the data provided by INMET (2014). The system boundaries are shown in **Figure 12.2**.

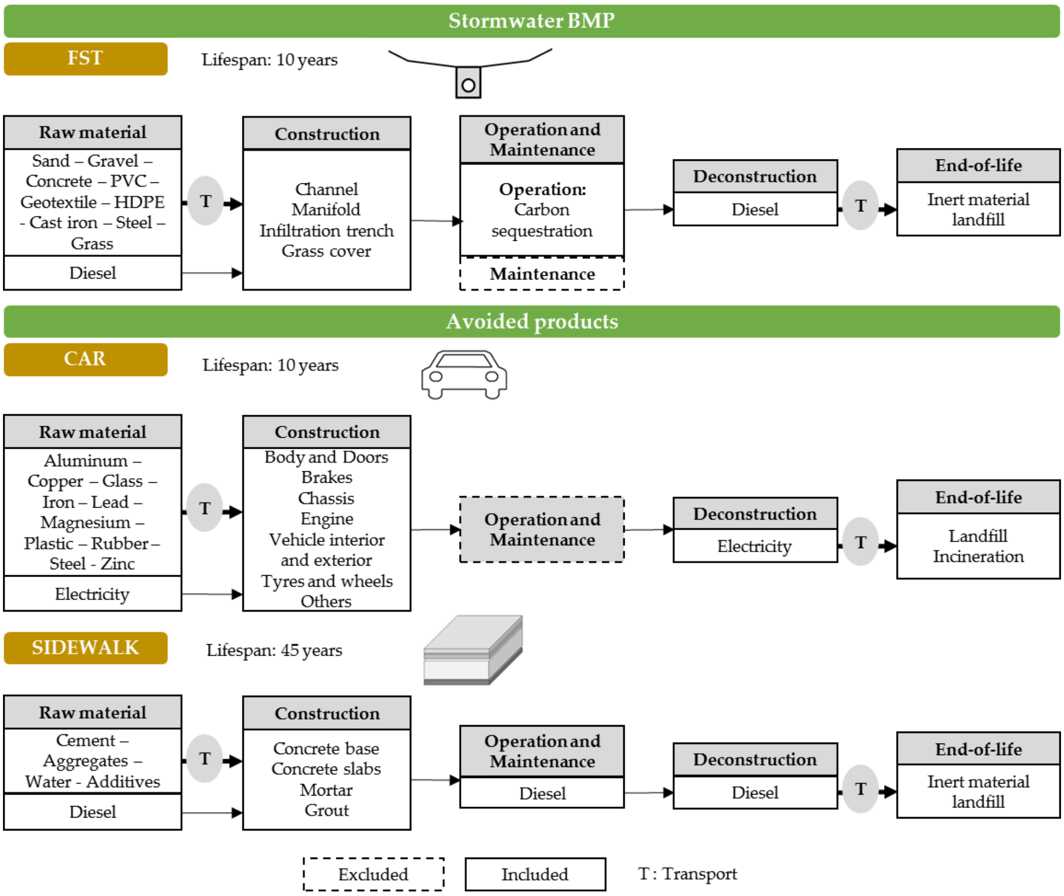


Figure 12.2 System boundaries of the systems under analysis, i.e., FST, car and sidewalk

Inventory analysis: To account for the material and energy flows related to the life cycle of a product, a life cycle inventory (LCI) was compiled. First, the FST was assessed from cradle to grave, i.e., from the raw material extraction to the end-of-life phase (**Figure 12.2**). In this case, the operation phase accounted for the carbon sequestration potential. Maintenance was excluded because it was considered negligible according to the managers and was linked to the end of service life of the FST based on a previous study (Petit-Boix et al. 2015b). We assumed that the system had an average lifespan of 10 years due to maintenance factors (USDT, 2014), although this value might vary depending on the management practices, runoff quality and precipitation intensity.

Second, LCIs were composed for the avoided products. In this particular case, cars and sidewalks were the damaged goods. For the environmental assessment of a compact car, we applied an LCI that combined data from different companies (Hawkins et al. 2013). Assuming that a car would be used regardless of being newly produced after the flood or being unharmed, the fuel consumption was not accounted for because these impacts would not be avoided. The estimated lifespan of the car was 10 years of on-road duration

(Schweimer and Levin 2000). Regarding sidewalks, we adapted the design that was presented in a previous study for one m² of concrete sidewalks with an average lifespan of 45 years (Mendoza et al. 2012). In both cases, the end-of-life phase was included, given that the damaged products must be disposed of after the flooding event. All of the production processes that were included in the LCI were adapted with the marginal electricity mix for Brazil (Schmidt and Thrane 2009), which is the most competitive technology that would be used in the production of one extra unit of product. The ecoinvent 2.2 (Frischknecht et al. 2005) database was used for retrieving background data on the life cycle of the materials and processes involved in each system. For data on the LCI of the FST and avoided products, see **Chapter 11** and **Table 12.2**, respectively. In this first analysis, the impacts on land use change were not accounted for because this was already an artificialized area, and we considered that the new system provided a greater carbon sequestration potential.

Table 12.2 Life cycle inventories of a car and concrete sidewalk

LCI of one m² of sidewalk (0.2x1x1 m) Source: Mendoza et al. (2012)			**LCI of one car** Source: Hawkins et al. (2013)		
	Amount	**Units**		**Amount**	**Units**
Construction					
			Steel	8.5E+02	kg
			Iron	1.2E+02	kg
			Aluminum	5.4E+01	kg
			Copper	2.2E+01	kg
			Lead	3.0E-01	kg
			Mg	2.0E-01	kg
			Zn	1.0E-01	kg
			Glass	6.1E+01	kg
Portland cement	6.3E+01	kg	Plastic	1.5E+02	kg
Gravel	2.4E+02	kg	Rubber	2.5E+01	kg
Water	3.5E+01	kg	Paint	1.4E+01	kg
Silica sand	1.9E+02	kg	Oxygen	8.2E-02	kg
Diesel	7.3E+00	MJ	Acetylene	6.5E-03	kg
Transport	7.1E+01	tkm	Nitrogen	1.2E-01	kg
			Carbon dioxide	5.5E-01	kg
			Drinking water	3.2E+00	m³
			Operating water	3.1E+01	m³
			Heat, natural gas, <100kW	3.4E+03	MJ
			Heat, natural gas >100kW	1.9E+02	MJ
			Compressed Air (6 bar)	7.7E+02	Nm³
			Compressed Air (12 bar)	2.1E+02	Nm³
			Natural gas	3.2E+01	kg
			Electricity	1.3E+03	kWh
Operation and Maintenance					
Diesel	8.8E+00	MJ			
Demolition					
Diesel	1.7E+01	MJ	Electricity	4.1E+02	kWh
End-of-life					
Transport to landfill	3.7E+02	tkm	Transport to landfill	3.3E+01	tkm
Landfill	3.7E+02	kg	Landfill	2.4E+02	kg

Impact assessment: Of all of the stages that are included in the impact assessment stage (ISO 14040:2006), only the classification and characterization were considered. Using the SimaPro 8.0.4 (PRé Consultants 2010) software, the method that was applied was the hierarchical approach of ReCiPe 2008 (Goedkoop et al. 2009). The selected midpoint indicators were global warming (GW, kg CO_2 eq), ozone depletion (OD, kg CFC-11 eq), terrestrial acidification (TA, kg SO_2 eq), freshwater eutrophication (FE, kg P eq), marine eutrophication (ME, kg N eq), human toxicity (HTP, kg 1,4-DB eq), photochemical oxidant formation (POF, kg NMVOC), water depletion (WD, m^3), metal depletion (MD, kg Fe eq) and fossil depletion (FD, kg oil eq). The cumulative energy demand V1.08 (CED, MJ) was also included to evaluate energy issues (Hischier et al. 2010).

Interpretation: The main contributors to the impacts of the FST, car and sidewalk were analyzed. The overall results were assessed using the EP and NEI to determine the balance between the investment and avoided impacts.

12.3 Results

Table 12.3 shows the environmental impacts of the FST and damaged products (i.e., car and sidewalk) for their respective lifespan. There were different processes that especially contributed to the burdens of each of these elements. In the case of the FST, the diesel that was consumed by the machinery accounted for 20-60% of the impacts in most of the impact categories, and the contribution of the infiltration trench was significant in 5 of 11 indicators (up to 80% of the life cycle impacts) (¡Error! No se encuentra el origen de la referencia.). With respect to the car, steel accounted for a great share of the impacts, given that it represented 65% of the production materials in terms of mass. Copper also had an important contribution to HT (>70%) because of its manufacturing process. Regarding the selected sidewalk design, the concrete base and transport were the major concerns, each accounting for between 20 and 60% of the impacts (**Appendix 9.2**).

Table 12.3 Unitary impacts of the FST, car and sidewalk and environmental payback of the FST

Impact category	Total unitary impacts			Environmental payback (EP)	
	FST	Car	m^2 sidewalk	Ratio FST-car	Ratio FST-sidewalk
Global warming (kg CO_2 eq)	6.4E+03	5.9E+03	7.6E+01	1.1	84
Ozone depletion (kg CFC-11 eq)	9.0E-04	5.7E-04	6.4E-06	1.6	142
Terrestrial acidification (kg SO_2 eq)	1.9E+02	2.9E+01	2.3E-01	6.5	824
Freshwater eutrophication (kg P eq)	1.7E+00	6.0E+00	4.4E-03	0.3	385
Marine eutrophication (kg N eq)	2.2E+01	1.5E+00	1.3E-02	15	1,761
Human toxicity (kg 1,4-DB eq)	1.6E+03	1.1E+04	6.2E+00	0.1	259
Photochemical oxidant formation (kg NMVOC)	5.7E+01	1.7E+01	3.6E-01	3.3	160
Water depletion (m^3)	1.7E+02	6.7E+01	9.5E-01	2.5	176
Metal depletion (kg Fe eq)	1.8E+03	5.0E+03	1.8E+00	0.4	1,006
Fossil depletion (kg oil eq)	2.4E+03	1.7E+03	1.5E+01	1.4	159
Cumulative energy demand (MJ)	3.2E+05	1.0E+05	7.5E+02	3.2	427

Applying the calculation procedure that is presented in Step 5 (**Section 12.2.2**), the EP of an FST was estimated. In terms of kg of CO_2 eq, the environmental investment related to the FST was recovered when the destruction of one car was prevented in 10 years. Considering that an average of three events occur every year, the potential emission of 6.4E+03 kg of CO_2 eq can be quickly compensated. The equivalent approach for sidewalks resulted in 84 m^2. The impact category with the largest EP is ME, resulting from the fertilizers required to produce the grass sod that was planted in the FSTs. In this case, the impacts were recovered when 15 cars and 1,761 m^2 of sidewalk were not destroyed.

Furthermore, results related to the NEI of the FSTs implemented to the case study neighborhood are shown in **Table 12.4**, as calculated according to Step 6. The impacts of the FSTs without accounting for the avoided burdens (A) and applying the NEI (B) were compared. The percentage reduction of the impact of the BMP per event varied in each impact category depending on their EP. Nonetheless, there were remarkable changes when calculating the NEI. For instance, the HT could be reduced by almost 700% with respect to A given that the production of the metal parts of the car would be avoided. In contrast, there was an 8% reduction in ME because this indicator had the lowest EP.

Table 12.4 NEI per event of the implementation of FSTs in RDI, considering 179 FSTs, six cars, 150 m^2 of sidewalk and an average of three intense flooding events per year

Impact categories	A Impacts of FSTs per event	B NEI of FSTs per event	Reduction
Global warming (kg CO_2 eq)	3.8E+04	-8.8E+03	123%
Ozone depletion (kg CFC-11 eq)	5.4E-03	9.9E-04	82%
Terrestrial acidification (kg SO_2 eq)	1.1E+03	9.2E+02	19%
Freshwater eutrophication (kg P eq)	1.0E+01	-2.7E+01	365%
Marine eutrophication (kg N eq)	1.3E+02	1.2E+02	8%
Human toxicity (kg 1,4-DB eq)	9.5E+03	-5.7E+04	697%
Photochemical oxidant formation (kg NMVOC)	3.4E+02	1.8E+02	46%
Water depletion (m^3)	1.0E+03	4.5E+02	55%
Metal depletion (kg Fe eq)	1.1E+04	-2.0E+04	283%
Fossil depletion (kg oil eq)	1.4E+04	1.5E+03	90%
Cumulative energy demand (MJ)	1.9E+06	1.2E+06	38%

12.4 Discussion

This first analysis showed an example of integrating damage prevention into the CLCA of a BMP. Uncertainty is relevant in CLCA and the number of effects that result from a decision might be incommensurable. In this case, we focused on the positive side of BMPs in the framework of flood prevention. However, there might be associated negative impacts, such as the creation of marginal demand for energy, fertilizers, mining, etc., related to the added pressure of producing FSTs. No further effects were accounted for, except for the marginal electricity generation, which resulted in very small variations in the environmental impacts (less than 1%).

Only data related to a single event could be applied as a first step towards more complex studies at the watershed or city scale. Obtaining this type of data presents great difficulties, especially in the case of private property damage, as only when there is an insurance coverage, for instance, will the damage be reported. This means that the number of damaged products might be typically greater than that found in databases or official reports. Damage to buildings would be expected, especially in basements and electric and gas connections (U.S. Department of Commerce Economics and Statistics Administration Economist, 2013), as shown in **Chapter 10**. However, there is still a lack of studies that assess the environmental effects of material damage prevention using and our results could not be validated. Recent models integrated hydraulic features and damage functions (Chen et al. 2016), and the difficulties in damage modeling were presented. The comparison between damage estimates and national expenditures also highlights great errors related to inaccurate estimations (Downton and Pielke 2005). Therefore, this first approach should be combined with robust predictive models in order to provide more information about the NEI of these systems.

Nevertheless, modeling these consequences was out of the scope of this study. We illustrate how the environmental assessment of damage might provide more information about the benefits of green BMPs at a local scale. So far, the European Flood Directive (European Council 2007) has resulted in policies that tend to promote non-structural strategies, such as land use regulations. In this sense, investing in BMPs might not be attractive in certain areas, but decisions might change when further effects are considered. Here, we did not include an economic assessment because our goal was to highlight the environmental relevance of these decisions. However, the economic and environmental payback of these actions should be considered in future analyses.

The outcome of this approach might be of interest in the field of communication and policy-making. We believe that urban planners, local authorities and insurance companies might benefit from these results because they provide information about indirect consequences that might have a large impact on the society. For instance, managing green areas in low-density neighborhoods might be key to approaching a more sustainable urban model. The design of the FST could vary depending on the plot dimensions so as not to affect aesthetics and other functionalities, such as leisure or private use. Moreover, the design should comply with the technical requirements to prevent soil compaction and ensure the proper infiltration of stormwater. In addition, this multi-functional approach would also foster biodiversity and prevent a reduction in the ecological connectivity of the area.

12.5 Conclusions

This analysis provided insight into the environmental effects of integrating flood damage prevention into the LCA of BMPs. Our first example was based on a Brazilian residential neighborhood. The environmental payback of an FST was related to the prevention of at least one car or 84 m^2 of sidewalk during 10 years, which is a short payback time considering the frequency of flooding events in the area. Subtracting the material damage of a historic flooding event to the potential environmental investment made for implementing a set of FSTs resulted in a net positive performance of these systems. The most favorable impact category was the HT, as a gross estimate showed an impact reduction of almost 700%.

Nevertheless, there are limitations in this analysis. When addressing this hydrological phenomenon, a risk assessment is needed to determine the frequency of this type of flooding event. Return periods, maximum water flows and population exposure, among other parameters, should be determined in a regional case study and included in the proposed methodology. In this way, the potential effect of the BMP on the water flow can be identified. Additional data are required regarding the material losses of the events. In this case, only a hypothetical analysis could be presented with data from a single flooding event with damage, but a statistical analysis would provide a more accurate idea in an attempt to generalize. However, it is currently very difficult to find registers for damage in specific locations and to predict future damage.

Future research should couple the economic value of the predicted material and ecological damage, risk assessment models and environmental impacts of the BMPs. Hence, this approach would be of great interest to insurance companies and governments for the planning and reduction of their financial budget. Combined with the environmental implications of these events, decision-makers and urban planners would be provided with more data for managing flooding risks.

Part VI

Final remarks and future research

Chapter 13

Picture: Brainstorming with middle-school students about water sustainability

Discussion of the main contributions

Chapter 13 Discussion of the main contributions

This chapter aims to connect the research conducted in this dissertation to understand its contribution to the global picture of water sustainability.

13.1 An integrated assessment of sanitation and drainage systems

The studies developed in the context of this dissertation are framed in urban environments and seek to provide eco-efficient sanitation and drainage solutions to different types of regions. As pointed out in **Chapter 1**, cities need to adapt to an increasing sanitation demand, climate change and the obsolescence of existing infrastructure. This research covers different areas of interest and offers innovative and integrative approaches that might help manage these issues. **Figure 13.1** illustrates the main contributions.

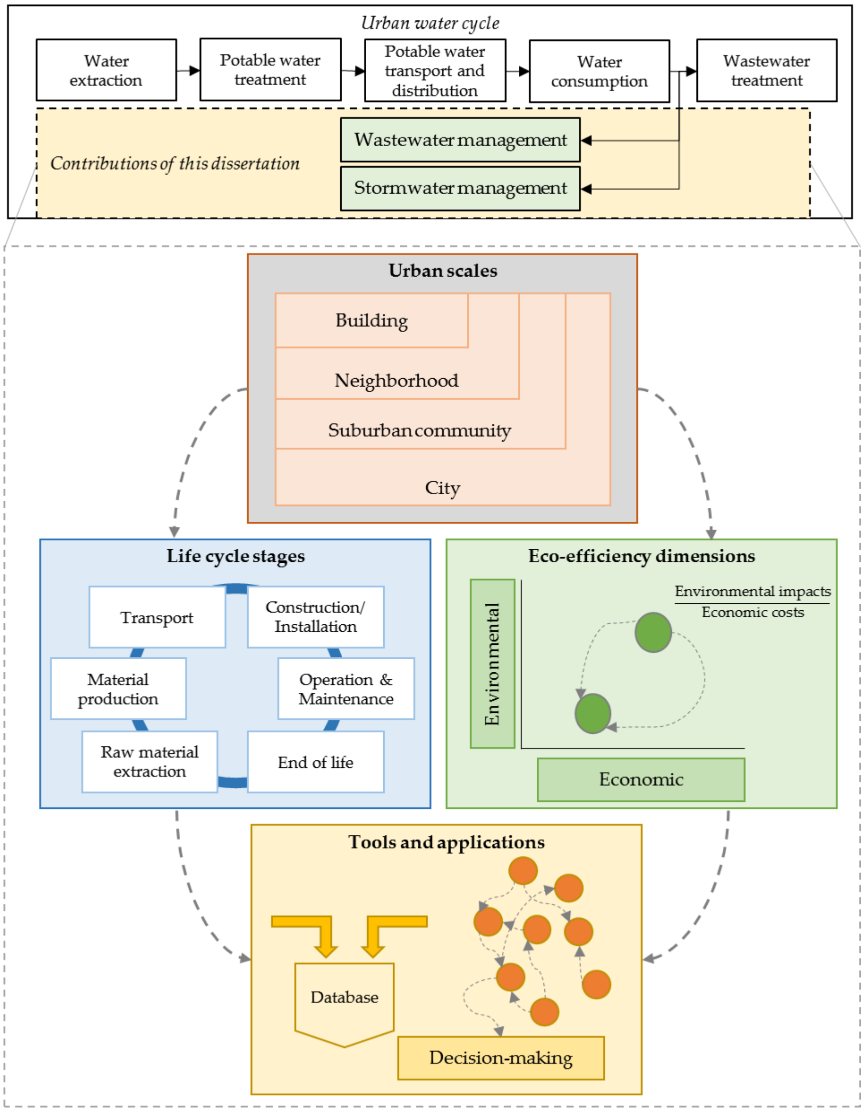


Figure 13.1 Contributions of this dissertation in the field of sanitation and drainage systems

13.1.1 *Analysis of different urban scales*

Although the concept of "city" is always present in the definition of all the assessments, different scales were considered to adapt the analysis to various needs. On the one hand, the research on ***sewer networks focused on the city as an entity***, as illustrated through the cases of Calalell and Betanzos in different climatic regions (Mediterranean versus Atlantic), urban forms (coastal versus inland configurations) and populations (seasonal versus permanent). This was indeed helpful to provide a great level of detail on the life cycle of sewers and to stablish potential parameters that might affect their performance. However, the sanitation alternatives of suburban settlements were also considered because these have a different sanitation demand than a larger urban area. In addition, they are typically located far away from existing treatment systems and need to be provided with environmentally sound solutions. This assessment is believed to be ***the first decentralization/centralization analysis developed in an island*** (i.e., Minorca), which is especially relevant given the treatment difficulties faced by seasonal areas.

On the other hand, ***drainage systems were addressed at three different scales*** because of the complexity of this infrastructure. In order from smallest to largest urban scale, the FST (i.e., green BMP) was designed based on a single building and later assessed in the context of a low-density neighborhood in Brazil; in contrast, ephemeral streams covered three different cities in a Mediterranean watershed. However, the studies had a strong focus on the environmental effects associated with urban runoff, without accounting for alternative uses of rainwater in buildings or neighborhoods. These aspects were already covered in previous analyses of rainwater harvesting systems (Vargas-Parra et al. 2013, 2014; Vargas-Parra 2015; Angrill et al. 2012, 2016), among others. The flood prevention studies included in this dissertation are believed to be the first step towards the implementation of eco-efficient measures at the urban scale. Nevertheless, their complexity requires that future analyses broaden their scope and methods as explained in **Chapter 15**.

13.1.2 *A complete life cycle perspective with an integrated essence*

Another particularity of this research is the life cycle approach. Each of the analyses covered one or more life cycle stages related to the system of interest. Especially in the case of sewer networks, there is a lack of in-depth studies that assess their variations and requirements under different circumstances. This research gap was addressed through different tools in this dissertation:

- At the ***construction stage***, a parametric analysis was combined with LCA to determine the effects of traffic loads and pavement degradation on the impacts of sewers. In addition, the designs that were proposed were technically feasible in compliance with regulations and conducted thanks to the collaboration of civil engineers.
- The ***operation and maintenance (O&M)*** presented several challenges because it is extremely heterogeneous depending on the city. On the one hand, energy issues vary by site based on climate and urban form and they were evaluated through a statistical analysis of a sample of cities covering typical scenarios. This modeling effort offered some guidance for understanding how urban sanitation should be envisioned in the context of urban complexity. On the other hand, the direct GHG emissions that result from wastewater degradation are still a field to explore in

more detail. However, this first onsite experimental process was of use to shed light on the potential emissions of sewers.

- Based on the construction and O&M modeling results, ***case studies*** served to validate these data and identify specific features that might indeed affect the environmental and economic effects of sewers. In these regions, the location of the WWTP was of great relevance in environmental terms. This approach was essential to providing data at different levels, i.e., ***from both a general and specific perspective***. Addressing sewers from this point of view is particularly relevant for decision-making, as different types of stakeholders are responsible for one or more life cycle stages. This means that disaggregated data are needed if the ones interested in improving the construction do not deal with O&M, which is therefore unimportant for them and vice versa.

In the case of stormwater management, life cycle tools were also included in the studies, but their main contribution was the methodological approach. When dealing with flood management, the main objective is to reduce damage, be it human, ecological or economic. For this reason, a series of steps were presented to ***design and calculate the environmental impacts of prevention infrastructure from an adapted CLCA perspective***. This approach was interesting to find out whether the selected systems are easily paid off when a number of urban elements is not destroyed in the short term.

13.1.3 The role of eco-efficiency in a system's net balance

The concept of "eco-efficiency" is not new, but this dissertation is probably one of the first studies that followed the guidelines stablished by ISO 14045:2012. As stated in **Chapter 3**, the sewer eco-efficiency results were validated by AENOR (http://www.en.aenor.es/aenor/aenor/perfil/perfil.asp), which adds up to the quality of the assessment. The integration of environmental and economic results played different roles depending on the system of interest and the selected approach:

- In sewers, eco-efficiency was used to determine the status of two different case studies based on their life cycle stages. Product system benchmarking is appealing when comparing and finding a starting point that depicts the way forward, the way of improving the eco-efficiency. However, we first need to understand what happens within the system boundaries of a product system. For this reason, ***the eco-efficiency studies of sewers not only compared two cities, but the life cycle stages of these systems***. This approach was key to identifying the main improvement routes. In this case, improvements were required in the pumping energy to reduce the environmental impacts, whereas labor contributed to the economic costs of the installation.
- In general, the applications in stormwater management research had a different goal. Flood studies balanced the investment in prevention infrastructure and the resulting avoided damage from an environmental and economic viewpoint. Although eco-efficiency ratios were presented, the main contributions were the payback period (PP) and net impacts. The PP was especially useful, and in the case of ephemeral streams, the ***differences between economic and environmental PP helped to determine whether money and impacts are paid off at the same time***.
- A challenging contribution of this analysis was the translation of the economic damage provided by insurances into environmental impacts. The nature of each

damaged item (e.g., private goods or services) increases the complexity of determining their exact composition and impacts. The ***use of national input-output tables*** was a first attempt at finding out their potential environmental effects.

13.2 Application in the context of the urban water cycle

As shown in **Figure 13.1**, the urban water cycle includes additional stages that were not accounted for in this dissertation. Existing research already dedicated large efforts in determining the environmental and/or economic effects of PWTPs (Amores et al. 2013; Muñoz et al. 2010), potable water transport and distribution networks (Sanjuan-Delmás et al. 2015a, 2014, 2015b), and WWTPs (Corominas et al. 2013; Hospido et al. 2004; Gallego et al. 2008; Lorenzo-Toja et al. 2015, 2016b; Rodriguez-Garcia et al. 2011). However, the sewer studies included in this dissertation belonged to a larger water cycle project (i.e., LIFE+ Aquaenvec). Through this collaboration with other research groups, the eco-efficiency of sewers in Calafell and Betanzos could be compared to that of the remaining water cycle stages.

Table 13.1 summarizes the global warming and economic costs of each life cycle stage in Calafell and Betanzos (Aquaenvec 2015), including the ones obtained in **Chapter 8**. The relevance of sewers is especially notable in the life cycle costs of the system, with a 40-45% contribution in both cities. In terms of global warming, WWTPs represent up to 50% of the cycle's impacts, but these values include the direct GHG emissions generated during the treatment process, which might account for 60% of this indicator (Lorenzo-Toja et al. 2016a). Due to the lack of precise models, sewer results do not consider the direct GHG emissions, meaning that they are expected to have larger impacts and the relevance of sewers is lower.

Table 13.1 Environmental and economic impacts of the urban water cycle in Calafell and Betanzos based on Chapter 8 and Aquaenvec (2015)

Stage	Betanzos		Calafell	
	Global warming (kg CO_2 eq/m^3)	Economic cost (€/m^3)	Global warming (kg CO_2 eq/m^3)	Economic cost (€/m^3)
PWTP	0.48 (*35%*)	0.30 (*14%*)	0.14 (*12%*)	0.36 (*15%*)
Potable water transport and distribution	0.09 (*6%*)	0.42 (*19%*)	0.09 (*8%*)	0.79 (*31%*)
Sewer network	**0.19 (*14%*)**	**0.98 (*45%*)**	**0.33 (*30%*)**	**1.02 (*40%*)**
WWTP	0.63 (*45%*)	0.49 (22%)	0.57 (*50%*)	0.36 (*14%*)
Total	**1.39**	**2.19**	**1.13**	**2.53**

When putting sewers in context, it becomes clear that their performance had to be better understood in order not to underestimate the transport of wastewater. In general, it seems that sanitation systems account for the largest global warming, but depending on the environmental indicators, results might again vary. For instance, ozone depletion is basically affected by PWTPs due to the chlorination processes, whereas WWTPs are responsible for the eutrophication impacts (Aquaenvec 2015). For this reason, science should provide decision-makers with a complete set of tools and indicators that help invest in the most eco-efficient alternatives. An example is shown in **Section 13.3**.

13.3 Tools and open data for decision-making

The results of this research constituted the framework of some tools and data that can be easily consulted for decision-making purposes. One of the products of the LIFE+ Aquaenvec project was an open web-based software (http://tool.life-aquaenvec.eu/en). This tool enables the calculation of a city's eco-efficiency based on the complete or partial stages of the water cycle. Some city council's and water managers can be assisted by this tool to understand the performance of their water and wastewater systems. A specific subsection is dedicated to sewer networks. The design and creation of this subsection was based on the structure, models and results of the sewer studies included in this dissertation. **Figure 13.2** shows the data input requirements of the construction phase, where the type of pipe materials, diameter, length, and trench design must be indicated.

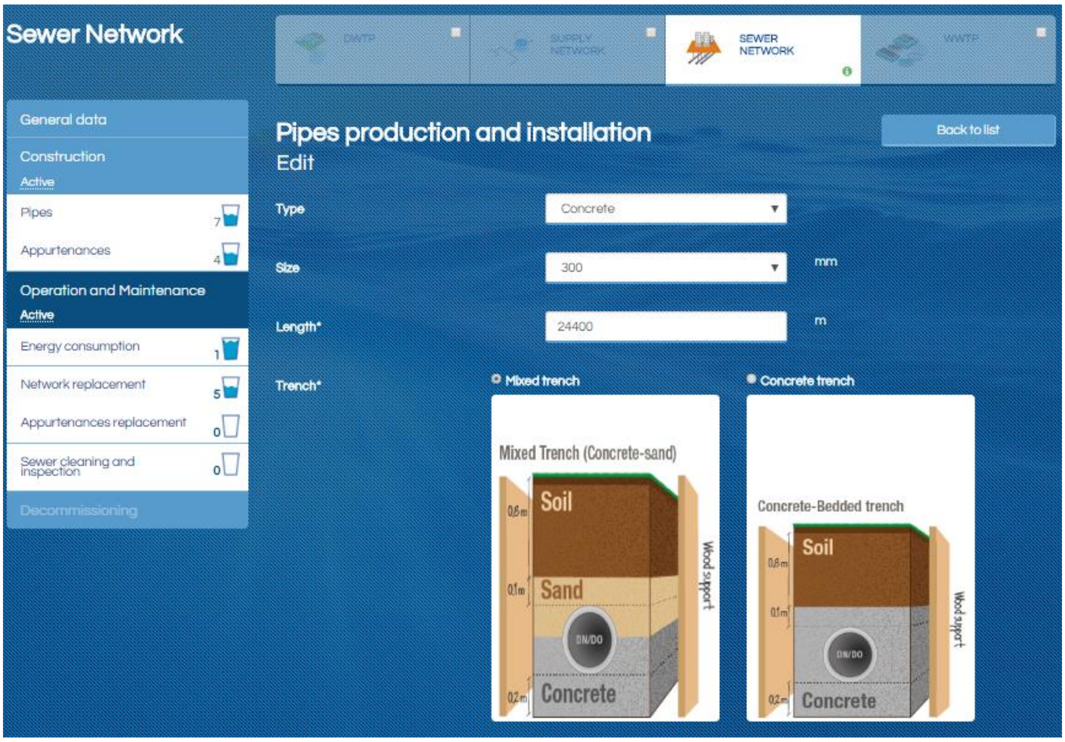


Figure 13.2 Sewer sub-section of the Aquaenvec tool available at http://tool.life-aquaenvec.eu/en

This software provides the estimates of four environmental indicators (i.e., global warming, ozone depletion, eutrophication and cumulative energy demand), and the economic cost. However, LCA practitioners interested in conducting specific eco-efficiency assessments have specialized data available for use. Most of the life cycle inventories (LCI) elaborated during this research were submitted, validated and published at the LCADB.sudoe database (http://lcadb.sudoe.ecotech.cat). This database focuses on the southwest of Europe to offer representative LCIs than can be used in the studies conducted in this region. **Figure 13.3** presents an example of the LCI of one meter of sewer, which results from **Chapter 4**. Some of the LCIs associated with this dissertation are available for further case studies and contributes to the pool of LCA-related data demanded by research and companies. These are meant to reduce the complexity and workload of modeling processes by providing background data.

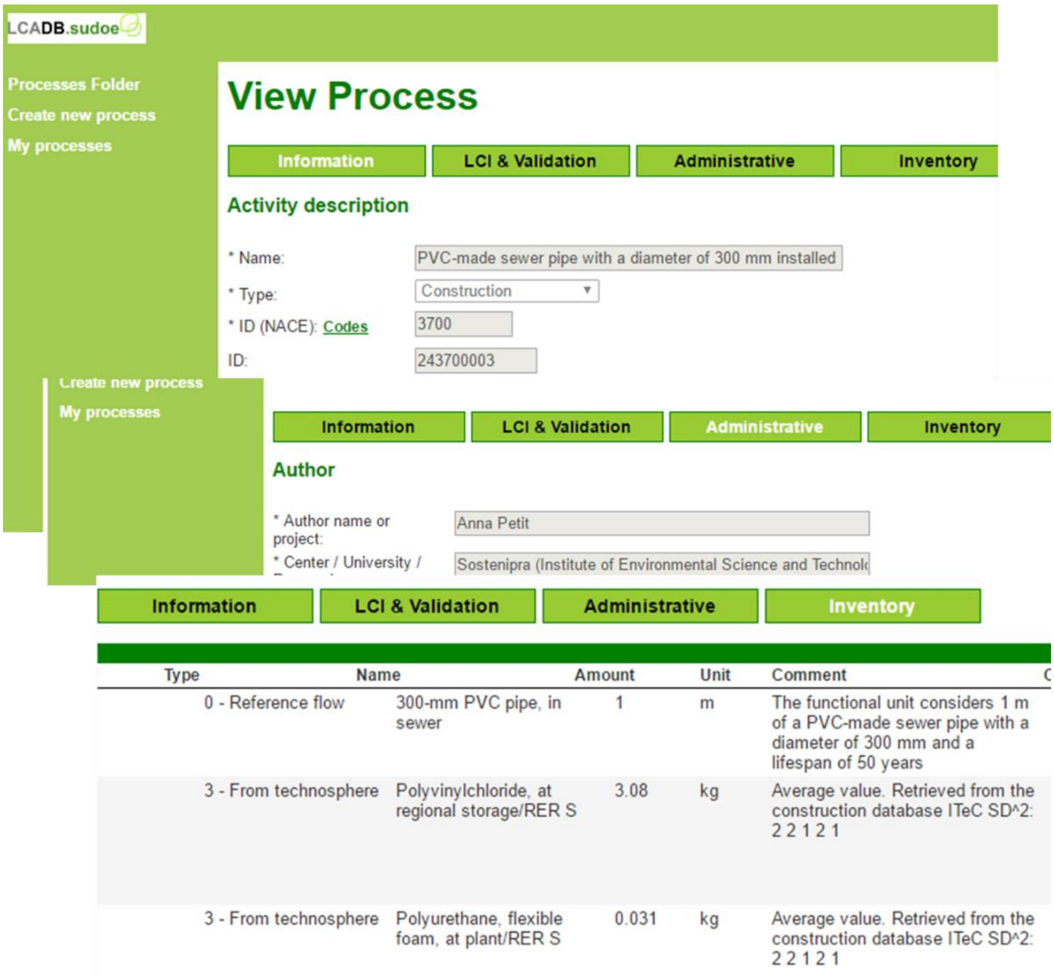
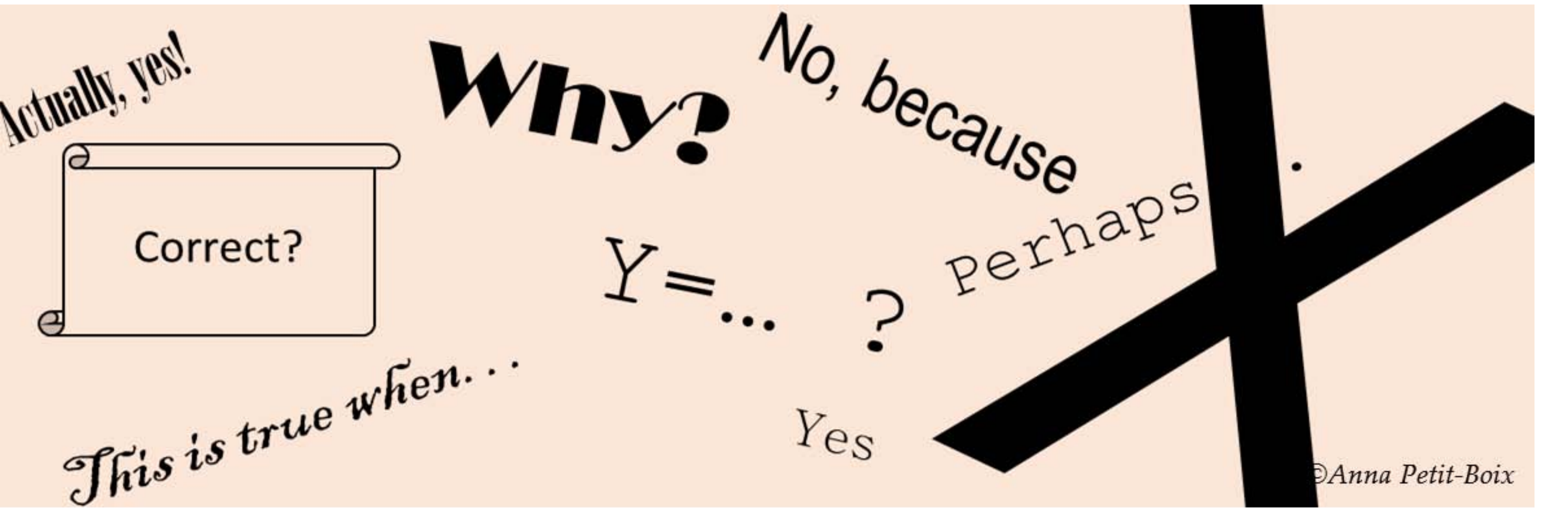


Figure 13.3 Sewer LCI available at the LCADB.sudoe database

Chapter 14



Picture: Answering and justifying our questions

Conclusions

Chapter 14 Conclusions

After discussing the potential contributions of this dissertation to the current sanitation and drainage knowledge, this chapter summarizes the conclusions based on the research questions presented in **Chapter 2**.

QUESTION 1 What are the environmental and economic hotspots of sewer networks and the parameters that affect this outcome in medium-sized cities?

Each of the life cycle stages of sewers was assessed in detail and under different circumstances in **Chapters 4 to 8**.

ANSWER 1a The environmental impacts of sewer construction are highly dependent on the pipe materials, trench designs and external disturbances

The first stage dealt with the constructive solutions applied in sewers. The main quantitative results obtained in **Chapters 4 and 5** can be summarized as presented in **Table 14.1**. The first LCA values (**Chapter 4**) pointed to the **benefits of concrete pipes** when compared to plastic pipes due to their potential **longer service life** and material components. One of the main findings of this research was the **large contribution of the installation phase** to the overall impacts of the construction. In some cases, it might represent **up to the 80% of the environmental impacts**, which can be very relevant in terms of management. Applying general trench designs, it was found that plastic pipes should preferably be embedded in soil, whereas concrete pipes might benefit from trenches that do not apply a large concrete base. The global warming results for one meter of pipe with Ø 300 mm ranged between 120, when using concrete pipes, and 190 kg CO_2 eq., when selecting HDPE.

Table 14.1 Environmental results of the sewer constructive solutions and potential recommendations

Pipe material (Ø 300 mm)	Chapter 4			Chapter 5		
	Best trench design	Lifespan	kg CO_2 eq/m in 100 years	Effects of pavement degradation	Traffic load effects	Recommendation
Concrete	Concrete + soil	100	120	Medium	Medium	Embed in soil-based trench with an equivalent structural function
HDPE	Soil	50	190	High	High	If traffic or pavement disturbances are expected, thinner pipes might need to be replaced. Embedding in concrete might be a better solution
PVC	Soil	50	150	High	High	

However, because these results are particularly dependent on the technical feasibility of the designs, a parametric analysis shed light on potential differences among materials when equivalent designs are studied (**Chapter 5**). Here, we found that the functions of a soil-based and a concrete-based trench are the same when protecting concrete pipes. However, **up to 56% of the environmental impacts can be reduced if soil-based trenches are applied**. In addition, pavement degradation and traffic loads generated some effects on the environmental performance of this design, but the case of plastic pipes was comparatively more relevant. Plastic pipes embedded in soil required varying plastic thicknesses with increasing traffic loads and pavement degradation. For this reason, when superficial disturbances are not expected at the time of construction, the selected pipes might be thinner than the future requirements. This ultimately results in recurrent repositions to implement the right pipe design. An alternative is to include **a concrete bedding that becomes environmentally attractive when the application of soil-based trenches is unfeasible** (for instance, at traffic loads of 600 kN).

ANSWER 1b Urban form and topography are the main factors that influence the operation of sewers, as well as other variables such as climate

The analysis of the operation phase was challenging due to the number of variables involved (**Figure 14.1**). The statistical study performed through a sample of Spanish cities (**Chapter 6**) suggested that the **location and topography of the city** (i.e., inland or close to the sea, and height of the different elements of the system) and **climate** affect the energy consumption in sewers. On average, Atlantic cities consumed five times as much energy (19.8 kWh/TEP) as Mediterranean and Subtropical cities. The former usually transport larger stormwater flows in combined sewers. Similarly, coastal areas need three times as much electricity as inland sewers due to the location of the WWTP and additional blockage problems at sea level. In addition, correlation models showed that the **length of sewer and wastewater production** were the main predictors of the energy requirements of an urban sewer. However, the topography of each city might also have a relevant effect.

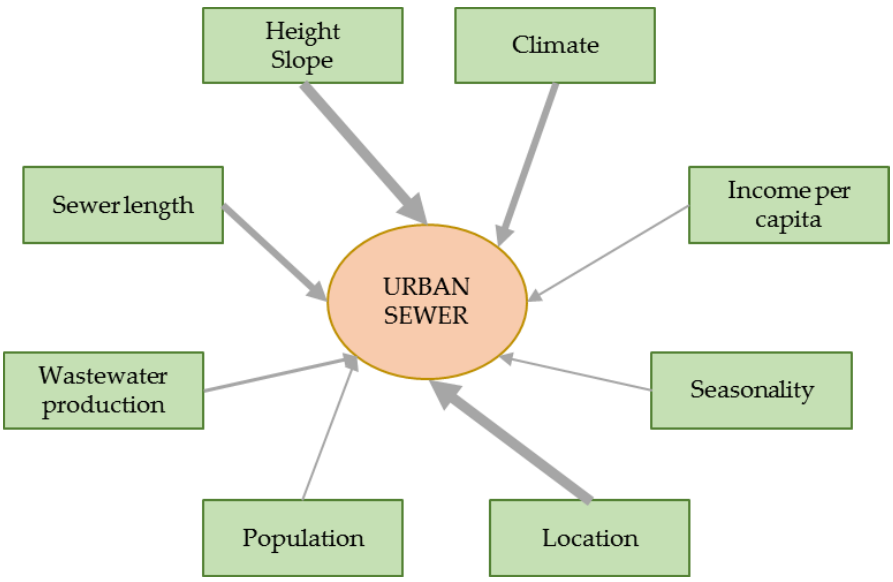


Figure 14.1 Parameters involved in the operation of sewers. The thickness of the arrows gives an idea of the effect of each parameter

This **modeling complexity was validated through the sewers of Calafell and Betanzos** in order to obtain detailed data on all possible parameters (**Chapter 8**). In this case, we found that **location was the key variable**, but it has a close relationship with other parameters because of the features of coastal cities. Calafell had the largest energy consumption (0.47 kWh/m^3) as opposed to Betanzos (0.11 kWh/m^3). The former is a coastal city with a **WWTP located at a higher altitude**, which demands a substantial energy input. The length of sewer is more important in terms of construction efforts. In this sense, the **location defines a certain urban form** that results in higher or lower environmental and economic impacts. For this reason, there is a need to envision **sewers in the context of the urban water cycle**, especially in combination with WWTPs, because their joint contribution to a city's eco-efficiency is crucial.

In addition, if the urban form demands more pump stations, larger amounts of direct GHG emissions might be expected. We found that temperature and wastewater turbulence in pump stations are two drivers towards the generation of CH_4 and N_2O (**Chapter 7**). For instance, the CH_4 emissions in Calafell ranged between 1.8 and 316.7 µg/L in the summer (29º C), and between 10 and 89.4 µg/L in the winter (15ºC). A model is still needed to predict the total direct emissions in sewers, but **this first analysis suggested that the measurements at a single sampling site might account for at least 4% of the operational impacts**. For this reason, we expect that direct emissions will have a larger environmental importance once a model predicts the GHG of the entire network.

ANSWER 1c The eco-efficiency of sewers depends on the operational impacts and installation costs

Assessing the eco-efficiency of sewers presented two different interests (**Chapter 8**). On the one hand, identifying the hotspots within the life cycle of sewers in two contrasting cities to find out whether they have common features or not. On the other hand, providing a convenient communication tool for decision-making that combined the environmental and economic implications of sewer networks.

In the first case, we found that **although Calafell and Betanzos have different features, their eco-efficiency presents similar trends** **(Figure 14.2)**. The **installation stage (i.e., trench construction) accounted for the largest economic investment** due to the labor costs (70-75% of the life cycle impacts; 0.7 €/m^3). In contrast, the **operation and maintenance (O&M) resulted in greatest environmental impacts** because it was affected by the energy consumption, with 140 mPt/€. Again, the greatest challenge in Calafell was associated with the location of the WWTP. Using average WWTP data, the global impacts of these cities resulted in seven times more life cycle costs and about twice as much environmental impacts than WWTPs. However, the environmental impacts of WWTP are larger when direct GHG emissions are considered and this relationship might be different. These variations also highlight the need for an integrated management of sewers and WWTPs that improves the eco-efficiency of sanitation systems.

In terms of communication, **eco-efficiency seems to be a robust tool** that can be easily understood by the target audience. However, it does not illustrate the social perspective of the system, although it might be intrinsically covered through some aspects such as job creation (i.e., investment in labor). One of the challenges of this analysis was **the selection of an indicator** that illustrates the environmental and economic status of the systems without disregarding potential trade-offs among environmental compartments. A weighted single score indicator was applied for an easier representation and communication, but some uncertainties still remain unclear, such as the modeling choices applied in weighted models.

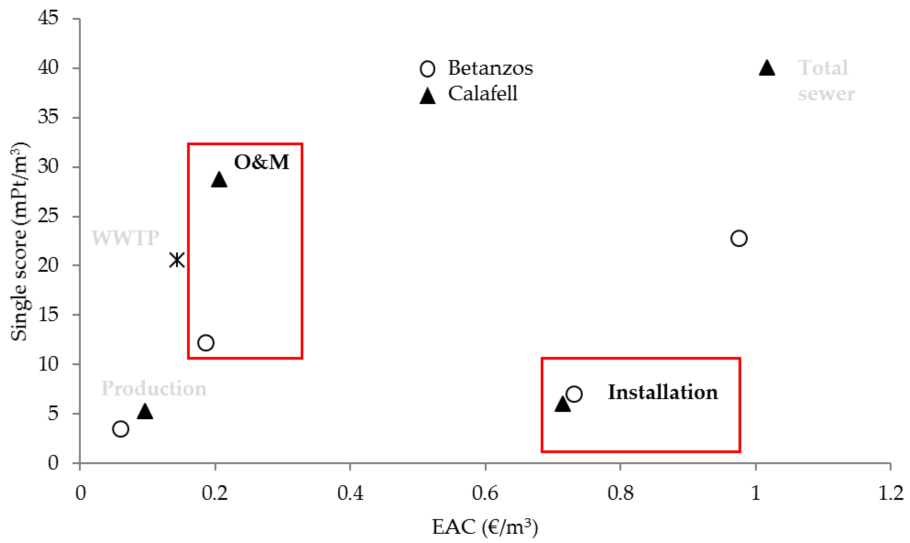


Figure 14.2 Life cycle stages with the greatest room for improvement

QUESTION 2 Is decentralized sanitation a better environmental option in small suburban areas when dealing with peak sanitation demand?

ANSWER 2 Centralization and partial decentralization offer environmental benefits depending on the indicators and conditions of suburban seasonal settlements

The previous analyses dealt with centralized sanitation in existing cities. But, what if suburban areas are in need of new infrastructure, or urban expansions are projected? Should it have the same structure as in a centralized city? **From an environmental perspective, answering to this question was not straightforward**. The study was based on an area that faces this problem (**Chapter 9**). In this case, a suburban settlement in Minorca was selected because (1) there are significant population fluctuations during seasonal periods, and (2) aquifer pollution and eutrophication threaten the aquatic ecosystems of the island due to the lack of organized treatment plans. Five treatment scenarios were proposed with varying degrees of centralization.

In this specific location, it seems that connecting the area to an existing neighboring WWTP with a larger treatment capacity offers the best environmental results. A single score indicator suggested that the **economies of scale of the large WWTP** (390 mPt/m³) **reduce the environmental burdens** by 12% with respect to partial decentralization through septic systems, or by 36% with respect to a constructed wetland. Centralization also offered some **benefits in terms of reduced eutrophication impacts**, but the same results were obtained when treating wastewater in septic tanks and transporting the effluent to the WWTP via tanker truck. However, the **sewer connection** accounted for 50% of the cumulative energy demand and this indicator did not benefit from this alternative. In the other cases, **septic**

tanks, road transportation, and the treatment efficiency of the local WWTP were the main hotspots.

Because this analysis was based on theoretical modeling, a **probabilistic assessment** was conducted to determine if these results could be extrapolated to other suburban settlements with seasonal population. When varying peak population, summer wastewater generation, and duration of the tourist season, **septic tanks with road transportation of the effluent obtained the best average scores** (520 mPt/m^3). In general, it seems that results are **sensitive to the duration of the tourist season,** which might differentiate mass tourism destination from second-home tourism. The **centralized scenario presented the largest variability** (77%), but it was associated with the **pumping energy**, which again depends on the urban form. This highlights the need for studying permanent and seasonal urban settlements in different types of analyses that cover the specific variables affecting each system.

This analysis contributed to the **centralization versus decentralization debate**, but it becomes clear that sometimes a single solution does not always offer the best results. This was a **first application in an insular context** that should be incorporated into this debate in order to come up with the best alternatives. What becomes clear is that permanent and seasonal areas have different behaviors and need to be addressed independently.

QUESTION 3 Do flood prevention strategies result in a positive environmental and economic performance when avoided damage is accounted for? If so, are green BMPs better than gray systems?

ANSWER 3a Prevention strategies and new infrastructure can be justified taking into account flood damage

In the field of stormwater drainage, studies focused on flood prevention systems and their environmental and/or economic impacts in different climatic and urban conditions (**Chapters 10 to 12**). The main result of these assessments was the **proposal of a methodological approach for expanding the system boundaries beyond the life cycle of the prevention strategies** to integrate the damage that they potentially avoid in an urban context (**Figure 14.3**). This idea was applied in two different contexts and required specific adaptations with the aim of justifying the need for prevention infrastructure.

On the one hand, **post-disaster emergency actions were assessed in ephemeral streams** after a major flooding event that affected three Mediterranean cities (**Chapter 10**). The environmental and economic investment in these actions was evaluated from a **historical perspective** to account for the damage and stormwater flows generated before and after their implementation. One of the challenges of this analysis was to **translate the economic damage** provided by insurance companies into environmental impacts and national input-output tables were used as a first solution.

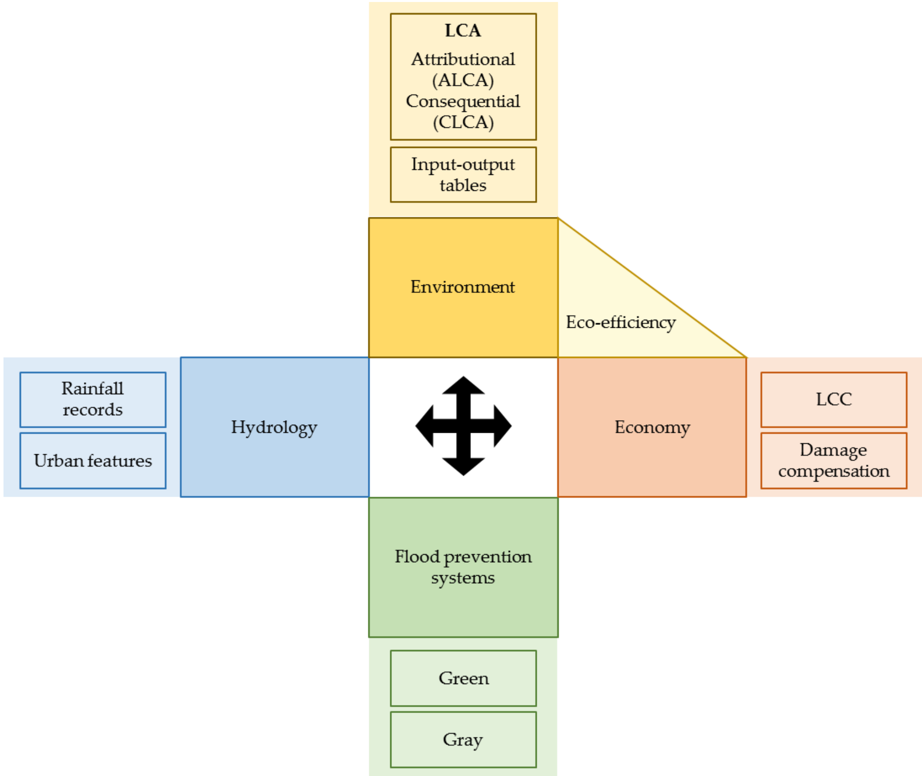


Figure 14.3 Integrated approach for assessing stormwater drainage and flood prevention

On the other hand, a green prevention system was studied in Brazil (**Chapter 11**) and a methodology was proposed in **Chapter 12** to again integrate damage prevention into the environmental assessment. This approach was based on the **concept of consequential LCA** (CLCA), as the system was expanded to include avoided damage in the analysis. This study was similar to the one of ephemeral streams, but it provides some additional hints. Here, **specific urban elements (i.e., a car and a sidewalk) were analyzed** in order to determine the balance of the prevention system with respect to a unit of product. The novelty of this evaluation can be useful when balancing to which extent we are willing to **compromise sustainability for safeguarding our personal properties**. Although this field should be studied in more detail, this is the first step towards understanding flooding in the context of urban sustainability.

ANSWER 3b The investment in green BMPs might be paid off more quickly than that of gray systems because of the material intensity

Comparing the green and gray systems that were studied in this dissertation is not straightforward because the scopes of the assessments were different. The FST and post-disaster emergency actions are not equivalent systems, but their function is to reduce the flooding risk. The former emulates a natural system, whereas the latter is an artificial system. Ephemeral streams can also be more natural when they are not in urban areas. However, comparing the eco-efficiency ratios that resulted from each eco-efficiency assessment might be an option (**Figure 14.4**). It seems that **every euro invested in restoring the ephemeral streams triplicated the emissions of the FST**.

	Filter, swale and infiltration trench (FST)	Post-disaster emergency actions in ephemeral streams
ECO-EFFICIENCY RATIO	0.38 kg CO_2 eq / € (0.35 kg CO_2 eq / USD)	1.2 kg CO_2 eq / €

Figure 14.4 Eco-efficiency ratios for two flood prevention strategies

Because the systems and climatic conditions (i.e., Spanish and Brazilian cities) are not the same, one might argue that perhaps the FST is comparatively more expensive than the restoration actions and this comparison might be inconclusive. However, the environmental impacts and economic costs of these actions were balanced with the potential damage that they prevented. Here, we found that they have an **economic payback of two years, whereas 25 years are needed to offset the environmental** investment. One of the reasons of these environmental results is the material intensity of the infrastructure, which implies the reconstruction of walls, or restoration of the streams, and is meant to be durable. In contrast, the FST enabled the sequestration of carbon and was not concrete intensive, which might have reduced the global warming scores obtained in the LCA. In fact, the **global warming of an FST is** expected to be paid off once the destruction of one car is avoided during a flooding event. In the short term, this might happen quickly due to the frequency of flooding in the area. Furthermore, the FST provided additional **benefits in terms of stormwater infiltration and storage**, and resulted in a more convenient drainage solution when compared to storing and transporting stormwater through a pipeline.

Again, this field still requires more analyses that define the actual comparison of green and gray systems in urban areas. Some ideas were provided in this analysis to enhance the debate and ease the decision-making process through comparisons that are close to public concerns.

Chapter 15

Picture: Wetland at Maumee Bay State Park, Oregon (Ohio, USA)

Future research

Chapter 15 Future research

This research might inspire the elaboration of future analyses in the field of urban sanitation and drainage that help to improve local sustainability and be a step closer to sustainable cities.

Potential future research lines might revolve around the following topics:

(1) Sustainability assessment of sewer networks in big cities
(2) Analysis of decentralized systems at different scales
(3) Predictive modeling of flooding consequences
(4) Integration of the ecosystem services approach into the LCA of flood management

15.1 Sustainability assessment of sewer networks in big cities

The results of this dissertation refer to the behavior of small to medium sized cities, which dominate in most areas of the world. However, the proliferation of big metropolis implies the creation of large and complex sewer networks that might not follow the same trends that smaller cities do. For this reason, specific analyses should focus on a sample of big cities to study:

a) their urban structure and network composition to identify the main materials and diameters applied.
b) the creation of a model that builds upon the features of each neighborhood (e.g., urban form, water consumption, etc.), which are possibly similar to those of a small city.
c) the environmental, economic and social impacts of these networks, considering that direct GHG emissions might affect a greater share of the population and social effects might be more relevant.

With these ideas, the sustainability of sewers might be the result of modeling layers that combine the social structure of the city, physical properties of the area and the investment in sanitation infrastructure. To do so, different types of tools are needed. GIS-LCA/LCC based models might be helpful to associate the environmental impacts with certain sites of the city. Direct GHG emissions should also be modelled in more detail to predict the emissions produced along the network, not only at specific sites.

The social dimension can also be monitored through odor maps or sections with recurrent leakages and maintenance needs that generated traffic delays, noise or dust. In addition, the features of the stakeholders involved in all life cycle stages might be interesting, such as working conditions, equality, etc. Life cycle sustainability assessment (LCSA) might be a feasible method for connecting all data layers and coming up with a piece of information that is easily understood by citizens and local authorities. As a result, the relationship between the optimum sewer design and sustainability should lead to improvements at the city and neighborhood scale and within each life cycle stage.

15.2 Analysis of decentralized systems at different scales

Because the centralization debate is usually inconclusive, real-life systems can provide accurate data for calculating the actual service demand, material use, treatment efficiency, etc. Still, modeling efforts are highly valuable for addressing variations and potential applications of a given system, as exemplified in this dissertation. However, scanning a number of urban, suburban and rural areas might lead to different results under different conditions.

For example, sprawled suburban areas are on the rise, but they only host permanent residents, which means that there are no variations in the wastewater generation in the summer – it might even decrease. In contrast, all-inclusive resorts might also be far away from urban areas and produce the largest sanitation demand during seasonal periods. Rural areas are also interesting because they might consist of a very small house ensemble with different consumption needs.

A precise analysis of different treatment and population scales would provide more insightful information about potential environmental and economic trends. Different treatment systems should be assessed in buildings, rural communities, suburban settlements, and urban neighborhoods. The distance to existing sanitation systems will condition the results, but they should be considered in a complete set of variables that includes the population type, density, water consumption patterns and status of the water bodies.

In this line, parallel research on decentralized systems should also address the provision of water in these areas. Stormwater was not included in the decentralization debate of this dissertation, but the variety of options that it offers to users should be assessed from a technical and sustainability point of view. For instance, suburban areas might benefit from existing green areas that act as retention systems and provide water during the driest seasons. This symbiosis is also interesting in the field of food production, as rainwater or reclaimed water can be used in urban agriculture or other production systems and reduce the demand for centralized potable water supply. In this sense, different scenarios should be analyzed to define the optimum systems that provide water for food production or services at the lowest costs.

15.3 Predictive modeling of flooding consequences

The exact consequences of a flooding event are very difficult to predict. However, hydrologic models are being adapted to this application in order to identify the areas of a city that might be most affected by urban runoff. For instance, Ciervo et al. (2015) simulated flash floods in a small city through numerical modeling and surveys. Still, the environmental dimension is not present in these assessments.

For this reason, future research should combine sustainability assessments with predictive risk models that map the specific damage and areas affected by flooding events. This analysis should also account for climate change patterns in the short and long terms to obtain more information about the frequency and intensity of future events. This should enable the quantification and comparison of past, present and future flooding events.

Prevention systems should be sized accordingly to determine the environmental and economic investment needed under each of these conditions. In this context, the methodologies proposed in this dissertation could be applied, modified and expanded to include more items and compare different systems with one another. CLCA is one the fields to explore, as the prospection would not only be accounted for in terms of hydrologic modeling, but in terms of impacts on sustainability. The modeling principles should be well defined, with system boundaries that include different expanded systems, such as the ones related to the materials and energy needed to produce the prevention technologies. A complete CLCA perspective is complex due to the number of processes involved in the products and systems, but each assessment should aim to move a step forward to give a clear picture of the consequences of our decisions.

15.4 Integration of the ecosystem services approach into the LCA of flood management

The consequences of flooding go beyond the effects on urban infrastructure. Prevention strategies can be provide additional benefits to humans, such as clean air, water availability or biodiversity. These benefits should be further addressed and integrated into the current modeling framework to understand the general effects of these systems. In this context, the incorporation of the ecosystem services approach into LCA is a field under development (Zhang et al. 2010a, 2010b).

This type of analysis would be helpful to identify the prevention systems, locations and conditions that increase the benefits of each alternative. When combined with LCA it should aim at providing more data for the decision-making process, although quantifying cultural services, for instance, might complicate this task. LCA models should be adapted to accommodate these parameters, which might ultimately cover all of the social, economic and environmental implications of prevention strategies in a comprehensive sustainability approach.

In fact, damage prevention might as well be considered an ecosystem service, because humans benefit from not having to replace their cars, furniture, etc. Including this aspect was complex, which means that ecosystem service analyses should first identify a set of aspects that stand out as the main problems identified by the society, for instance. Understanding how each of these services is created is of paramount importance if we seek to explain why cities must adapt to climate change and become more resilient.

References

©Suez services company. 2012. CONTEC (Control Técnico del Ciclo Integral del Agua) and SGO (Sistema de Gestión Operativa). ©Aqualogy services company.

Abril, P. and R. Argemí. 2009. *Application of an energy saving and efficiency programme to wastewater treatment plants: methodology and first results (Aplicació del programa d'estalvi i eficiència energética a les estacions de depuració d'aigües residuals: metodologia i primers resulta. IV Jornades Tècniques de Gestió de Sistemes de Sanejament D'aigües Residuals.*

AEMET. Spanish National Meteorological Agency. 2013. Standard Climate Values. http://www.aemet.es/en/serviciosclimaticos/datosclimatologicos/valoresclimatologicosAccessed November 2013.

AG. GEBERIT INTERNATIONAL. 1998. Environmental comparison of domestic sewage pipes of different materials (Ökobilanz von Rohren zur Hausentwässerung). *Geberit International AG, Switzerland*.

Ahn, Y.-H. 2006. Sustainable nitrogen elimination biotechnologies: A review. *Process Biochemistry* 41(8): 1709–1721. http://linkinghub.elsevier.com/retrieve/pii/S1359511306001280. Accessed July 30, 2014.

Akhtar, S., B. Reza, K. Hewage, A. Shahriar, A. Zargar, and R. Sadiq. 2014. Life cycle sustainability assessment (LCSA) for selection of sewer pipe materials. *Clean Technologies and Environmental Policy* 17(4): 973–992. http://link.springer.com/10.1007/s10098-014-0849-x. Accessed March 13, 2015.

Alberti, M. 1996. Measuring urban sustainability. *Environmental Impact Assessment Review* 16(4–6): 381–424. http://linkinghub.elsevier.com/retrieve/pii/S0195925596000832. Accessed January 30, 2017.

Alberti, M. 2010. Maintaining ecological integrity and sustaining ecosystem function in urban areas. *Current Opinion in Environmental Sustainability*.

Alberti, M. and L. Susskind. 1996. Managing urban sustainability: An introduction to the special issue. *Environmental Impact Assessment Review* 16(4–6): 213–221. http://linkinghub.elsevier.com/retrieve/pii/S0195925596000704. Accessed January 30, 2017.

Allenby, B.R. 1992. Achieving sustainable development through industrial ecology. *International Environmental Affairs* 4(1): 56–68.

Amores, M.J., M. Meneses, J. Pasqualino, A. Antón, and F. Castells. 2013. Environmental assessment of urban water cycle on Mediterranean conditions by LCA approach. *Journal of Cleaner Production* 43: 84–92. http://dx.doi.org/10.1016/j.jclepro.2012.12.033.

Anders, K. and A. Anders. 1997. Environmental assessment of sewage pipes in PVC, PE, PP and concrete (Miljøvurdering af afløbsrør i PVC, PE, PP og beton). *DOR/Nordiska Plaströrgruppen, Stockholm, Sweden*.

Angrill, S., R. Farreny, C.M. Gasol, X. Gabarrell, B. Viñolas, A. Josa, and J. Rieradevall. 2012. Environmental analysis of rainwater harvesting infrastructures in diffuse and compact urban models of Mediterranean climate. *The International Journal of Life Cycle Assessment* 17(1): 25–42.

http://link.springer.com/10.1007/s11367-011-0330-6. Accessed July 14, 2014.

Angrill, S., L. Segura-Castillo, A. Petit-Boix, J. Rieradevall, X. Gabarrell, and A. Josa. 2016. Environmental performance of rainwater harvesting strategies in Mediterranean buildings. *The International Journal of Life Cycle Assessment* in press: 1–12. http://link.springer.com/10.1007/s11367-016-1174-x. Accessed August 25, 2016.

Aparicio, F. 1999. *Fundamentos de hidrología de superfície. Limusa, México*.

Aquaenvec. 2015. Assessment and improvement of the urban water cycle eco-efficiency using LCA and LCC. Layman's report 2012-2015. http://www.life-aquaenvec.eu/wp-content/uploads/2016/05/AF-LAYMANS-AQUAENVEC-LR.pdf.

Arpke, A. and N. Hutzler. 2006. Domestic Water Use in the United States. A Life-Cycle Approach. *Journal of Industrial Ecology* 10(1): 169–184.

ASTM Standard C118. 2011. *Standard Specification for Concrete Pipe for Irrigation or Drainage*. DOI: 10.1520/C0118-11, www.astm.org.

ASTM Standard D2321. 2014. *Standard Practice for Underground Installation of Thermoplastic Pipe for Sewers and Other Gravity-Flow Applications*. DOI: 10.1520/D2321-14, www.astm.org.

Augas de Galicia. Xunta de Galicia. 2011. Annex III: Water use and demand. Galicia-Costa Hydrological Demarcation (Anejo III: Usos y demandas de agua. Demarcación Hidrográfica de Galicia-Costa). http://augasdegalicia.xunta.es/PHGC/PHGC-ES/PHGC_Anexo_3_Usos_Demandas.pdf Accessed January 2014.

Balkema, A.J., H.A. Preisig, R. Otterpohl, and F.J.. Lambert. 2002. Indicators for the sustainability assessment of wastewater treatment systems. *Urban Water* 4(2): 153–161.

Baptista, M., N. Nascimento, and S. Barraud. 2005. *Compensatory techniques in urban drainage (Técnicas compensatórias em Drenagem Urbana)*. *Brazilian Association of Water Resources*. Porto Alegre.

Barbosa, A.E., J.N. Fernandes, and L.M. David. 2012. Key issues for sustainable urban stormwater management. *Water Research* 46(20): 6787–6798.

Bare, J.C., P. Hofstetter, D.W. Pennington, and H.A.U. de Haes. 2000. Midpoints versus endpoints: The sacrifices and benefits. *The International Journal of Life Cycle Assessment* 5(6): 319–326. http://link.springer.com/10.1007/BF02978665. Accessed August 18, 2016.

Barjoveanu, G., I.M. Comandaru, G. Rodriguez-Garcia, A. Hospido, and C. Teodosiu. 2014. Evaluation of water services system through LCA. A case study for Iasi City, Romania. *The International Journal of Life Cycle Assessment* 19(2): 449–462. http://link.springer.com/10.1007/s11367-013-0635-8. Accessed July 30, 2014.

Barrera, A., M.C. Llasat, and M. Barriendos. 2006. Estimation of extreme flash flood evolution in Barcelona County from 1351 to 2005. *Natural Hazards and Earth System Sciences* 6(4): 505–518.

Beale, D.J., D.R. Marlow, and S. Cook. 2013. Estimating the Cost and Carbon Impact of a Long Term Water Main Rehabilitation Strategy. *Water Resources Management* 27(11): 3899–3910. http://link.springer.com/10.1007/s11269-013-0386-0. Accessed July 30, 2014.

Bizier, P. 2007. *Gravity Sanitary Sewer Design and Construction*. ASCE 2nd E. ASCE Publications: ASCE

Publications. http://books.google.com/books?id=Ya3vPw0guGIC&pgis=1. Accessed August 25, 2014.

Blanco, A., P. Pujadas, S.H.P. Cavalaro, and A. Aguado. 2014. Methodology for the design of controlled low-strength materials. Application to the backfill of narrow trenches. *Construction and Building Materials* 72: 23–30. http://www.sciencedirect.com/science/article/pii/S095006181401023X. Accessed December 10, 2015.

Bobylev, N. 2011. Comparative analysis of environmental impacts of selected underground construction technologies using the analytic network process. *Automation in Construction* 20(8): 1030–1040. http://linkinghub.elsevier.com/retrieve/pii/S0926580511000537. Accessed August 28, 2014.

BOE. 2001a. Real Decreto 1098/2001 de 12 de octubre, por el que se aprueba el Reglamento general de la Ley de Contratos de las Administraciones Públicas 257.

BOE. 2001b. Real Decreto Legislativo 1/2001, de 20 de julio, por el que se aprueba el texto refundido de la Ley de Aguas (BOE-A-2001-14276) 176.

Boon, A. 1995. Septicity in sewers - Causes, consequences and containment. *Water Sci. Technol* 31(7): 237–253.

Boon, A., A. Vincent, and K. Boon. 1998. Avoiding the problems of septic sewage. *Water Sci. Technol* 37(1): 223–231.

Bottero, M. and D. Peila. 2005. The use of the Analytic Hierarchy Process for the comparison between microtunnelling and trench excavation. *Tunnelling and Underground Space Technology* 20(6): 501–513. http://linkinghub.elsevier.com/retrieve/pii/S0886779805000210. Accessed August 28, 2014.

Bouchard, N.R., D.L. Osmond, R.J. Winston, and W.F. Hunt. 2013. The capacity of roadside vegetated filter strips and swales to sequester carbon. *Ecological Engineering* 54: 227–232. http://linkinghub.elsevier.com/retrieve/pii/S0925857413000335. Accessed July 30, 2014.

Brattebø, H. 2005. Toward a Methods Framework for Eco-efficiency Analysis? *Journal of Industrial Ecology* 9(4): 9–11. http://doi.wiley.com/10.1162/108819805775247837.

Brouwer, R. and R. van Ek. 2004. Integrated ecological, economic and social impact assessment of alternative flood control policies in the Netherlands. *Ecological Economics* 50(1): 1–21.

Bueno, S.M. 2010. What lies underground? Trenchless technology polls Municipalities on sewer pipe usage. 2010. *Pipe Materials Guide, Trenchless Technology Special Supplement*: 8–11.

Butler, D. and J.W. Davies. 2000. *Urban Drainage*. 3rd Ed. Spon Press, an imprint of Taylor & Francis. London and New York.

Camarasa Belmonte, A.M. and F. Segura Beltrán. 2001. Flood events in Mediterranean ephemeral streams (ramblas) in Valencia region, Spain. *CATENA* 45(3): 229–249. http://www.sciencedirect.com/science/article/pii/S0341816201001461. Accessed March 23, 2015.

Catalan Water Agency. 2003. *Technical recommendations for flooding studies at a local level (Recomanacions tècniques per als estudis dínundabilitat d'àmbit local)*. Barcelona, Spain.

Catalan Water Agency. 2015. Official project documents provided by the archive of the Catalan Water Agency, Barcelona.

Catalan Water Agency. 2016. Data consultation. http://aca-web.gencat.cat/aca/appmanager/aca/aca?_nfpb=true&_pageLabel=P3800245291211883042687.

Catalano De Sousa, M., A. Erispaha, S. Spatari, and F. Montalto. 2011. LCA as a Tool to Evaluate Green Infrastructure's Environmental Performance. *American Geophysical Union*. http://adsabs.harvard.edu/abs/2011AGUFMGC43B0896C. Accessed September 23, 2015.

CEDEX. Centre for Hydrographic Studies. 2009. *Technical guide on sewer networks and urban drain (Guía técnica sobre redes de saneamiento y drenaje urbano)*. 3rd ed. Centro de Publicaciones, Madrid: Ministerio de Fomento. Secretaría General Técnica.

Chan, A.L.S. and T.T. Chow. 2013. Energy and economic performance of green roof system under future climatic conditions in Hong Kong. *Energy and Buildings* 64: 182–198. http://linkinghub.elsevier.com/retrieve/pii/S0378778813002910. Accessed July 15, 2014.

Chen, A.S., M.J. Hammond, S. Djordjević, D. Butler, D.M. Khan, and W. Veerbeek. 2016. From hazard to impact: flood damage assessment tools for mega cities. *Natural Hazards*. http://link.springer.com/10.1007/s11069-016-2223-2. Accessed March 1, 2016.

Chen, D. and P. Szostak. 2013. Factor analysis of H2S emission at a wastewater lift station: a case study. *Environmental Monitoring and Assessment* 185(4): 3551–60. http://www.ncbi.nlm.nih.gov/pubmed/22941185. Accessed December 23, 2013.

Ciervo, F., M.N. Papa, V. Medina, and A. Bateman. 2015. Simulation of flash floods in ungauged basins using post-event surveys and numerical modelling. *Journal of Flood Risk Management* 8(4): 343–355. http://doi.wiley.com/10.1111/jfr3.12103. Accessed April 24, 2017.

CLABSA. Clavegueram de Barcelona; S.A. 2013. Personal Communication. Acer, 16, 08038 Barcelona, Spain.

Cleemput, O. Van. 1998. Subsoils: chemo- and biological denitrification, N2O and N2 emissions. *Nutrient Cycling in Agroecosystems* 52(2–3): 187–194.

Clemens, J. and B. Haas. 1997. Nitrous Oxide Emissions in Sewer Systems. *Acta Hydrochimica et Hydrobiologica* 25(2): 96–99. http://doi.wiley.com/10.1002/aheh.19970250207.

Cohen R, G. Wolff, E. Cousins, and B. Greenfield. 2004. *Energy Down the Drain. The Hidden Costs of California's Water Supply*. Natural Resources Defense Council, Pacific Institute, Oakland, California.

Cornejo, P.K., Q. Zhang, and J.R. Mihelcic. 2016. How does scale of implementation impact the environmental sustainability of wastewater treatment integrated with resource recovery? *Environmental Science & Technology*. http://dx.doi.org/10.1021/acs.est.5b05055. Accessed March 28, 2016.

Corominas, L., J. Foley, J.S. Guest, A. Hospido, H.F. Larsen, S. Morera, and A. Shaw. 2013. Life cycle assessment applied to wastewater treatment: state of the art. *Water Research* 47(15): 5480–92. http://www.sciencedirect.com/science/article/pii/S004313541300540X. Accessed October 1, 2014.

Council Directive. EC 2000/60/EC of 23 October 2000 on establishing a framework for Community action in the field of water policy.

Council of the European Union. 1991. Council Directive of 21 May 1991 concerning Urban Waste Water Treatment (91/271/EEC). *Official J. Eur. Commun. L* 34(May 1991): 1–16. http://scholar.google.com/scholar?hl=en&btnG=Search&q=intitle:COUNCIL+DIRECTIVE+of+21+May+1991+concerning+urban+waste+water+treatment+(91/271/EEC)#1.

CPSA. Concrete Pipeline Systems Association. 2010. PAS 2050- partial lifecycle assessment. Cradle-to-gate analysis for concrete pipeline. Manhole Ring and Cover Slab. Leicester.

CREM. Centro Regional de Estadística de Murcia. 2011. Evolution of the total Gross Available Income per home and per capita (Evolución de la Renta Disponible Bruta total de los hogares y per cápita). http://www.carm.es/econet/sicrem/PU_datosBasicos/sec167.html Accessed November 2013.

Crites, R.W. and G. Tchobanoglous. 1998. Small and decentralized wastewater management systems. *McGrawHill Series in Water Resources and Environmental Engineering*: 1084. http://books.google.com/books?id=yx9SAAAAMAAJ&pgis=1.

Crossland, C.J., D. Baird, J.-P. Ducrotoy, H. Lindeboom, R.W. Buddemeier, W.C. Dennison, B.A. Maxwell, S. V. Smith, and D.P. Swaney. 2005. The Coastal Zone — a Domain of Global Interactions. In *Coastal Fluxes in the Anthropocene. The Land-Ocean Interactions in the Coastal Zone Project of the International Geosphere-Biosphere. Editors: Crossland, C.J., Kremer, H.H., Lindeboom, H., Marshall Crossland, J.I., Le Tissier, M.D.A. (Eds.) Programme*.

CYPE Ingenieros. 2015. Construction costs generator [Generador de precios de la construcción] http://www.generadordeprecios.info/.

Czemiel Berndtsson, J. 2010. Green roof performance towards management of runoff water quantity and quality: A review. *Ecological Engineering* 36(4): 351–360. http://www.sciencedirect.com/science/article/pii/S0925857410000029. Accessed April 15, 2015.

Czepiel, P., P. Crill, and R. Harriss. 1995. Nitrous oxide emissions from municipal wastewater treatment. *Environmental Science & Technology* 29(9): 2352–6. http://www.ncbi.nlm.nih.gov/pubmed/22280278.

Daelman, M.R.J., B. De Baets, M.C.M. van Loosdrecht, and E.I.P. Volcke. 2013. Influence of sampling strategies on the estimated nitrous oxide emission from wastewater treatment plants. *Water Research* 47(9): 3120–30. http://www.ncbi.nlm.nih.gov/pubmed/23557698. Accessed December 23, 2013.

Davies, J., B. Clarke, J. Whiter, R. Cunningham, and A. Leidi. 2001. The structural condition of rigid sewer pipes: a statistical investigation. *Urban Water* 3(4): 277–286. http://linkinghub.elsevier.com/retrieve/pii/S146207580100036X.

Davis, J.L., N. Dana, S. Kameron, and D.. J. Roberts. 1998. Analysis of concrete from corroded sewer pipe. *International Biodeterioration & Biodegradation* 42(1): 75–84. http://linkinghub.elsevier.com/retrieve/pii/S0964830598000493.

Day, T.J. 2000. *Sewer management systems*. John Wiley & Sons.

Debruyn, W., G. Lissens, J. Rensbergen, and M. Wevers. 1994. Nitrous oxide emissions from waste water 31(1): 159–165.

Deletic, A. and T.D. Fletcher. 2006. Performance of grass filters used for stormwater treatment—a field and modelling study. *Journal of Hydrology* 317(3–4): 261–275. http://linkinghub.elsevier.com/retrieve/pii/S0022169405002726. Accessed July 30, 2014.

Dennison, F.J., A. Azapagic, R. Clift, and F.S. Coulbourne. 1999. Life Cycle Assessment: Comparing strategic options for the mains infrastructure - Part I. *Water Science and Technology* 39(10): 315–319.

Desloover, J., S.E. Vlaeminck, P. Clauwaert, W. Verstraete, and N. Boon. 2012. Strategies to mitigate N2O emissions from biological nitrogen removal systems. *Current Opinion in Biotechnology* 23(3): 474–82. http://www.ncbi.nlm.nih.gov/pubmed/22244791. Accessed December 23, 2013.

Devkota, J., H. Schlachter, and D. Apul. 2015. Life Cycle Based Evaluation of Harvested Rainwater Use in Toilets and for Irrigation. *Journal of Cleaner Production*. http://www.sciencedirect.com/science/article/pii/S0959652615001341. Accessed February 23, 2015.

Diaz-Valbuena, L.R., H.L. Leverenz, C.D. Cappa, G. Tchobanoglous, W.R. Horwath, and J.L. Darby. 2011. Methane, carbon dioxide, and nitrous oxide emissions from septic tank systems. *Environmental Science & Technology* 45(7): 2741–7. http://dx.doi.org/10.1021/es1036095. Accessed March 15, 2016.

Dierkes, C., L. Kuhlmann, J. Kandasamy, and G. Angelis. 2002. Pollution Retention Capability and Maintenance of Permeable Pavements. In *Global Solutions for Urban Drainage*, 1–13. ASCE. http://cedb.asce.org/cgi/WWWdisplay.cgi?133447. Accessed February 18, 2015.

Dincer, F. and A. Muezzinoglu. 2007. Odor Determination at Wastewater Collection Systems: Olfactometry versus H2S Analyses. *CLEAN – Soil, Air, Water* 35(6): 565–570. http://doi.wiley.com/10.1002/clen.200700057. Accessed February 14, 2013.

Doka, G. 2003. *Ecoinvent report No. 13, Part IV: Wastewater Treatment*. Ecoinvent report for v1.3.

Domene, E. and D. Saurí. 2003. Urban Models and Water Consumption. Watering of private gardens in the Metropolitan Region of Barcelona (Modelos Urbanos y Consumo de Agua. El Riego de Jardines Privados en la Región Metropolitana de Barcelona). *Investigaciones Geográficas* 32: 5–17.

Domene, E. and D. Saurí. 2006. Urbanisation and water consumption: Influencing factors in the metropolitan region of Barcelona. *Urban Studies* 43(9): 1605–1623.

Doorn, M.R.J., S. Towprayoon, S.M. Manso Vieira, W. Irving, C. Palmer, R. Pipatti, and C. Wang. 2006. *2006 IPCC Guidelines for National Greenhouse Gas Inventories, Chapter 6: Wastewater Treatment and Discharge*. Japan.

Downton, M.W. and R.A. Pielke. 2005. How Accurate are Disaster Loss Data? The Case of U.S. Flood Damage. *Natural Hazards* 35(2): 211–228. http://link.springer.com/10.1007/s11069-004-4808-4. Accessed March 3, 2016.

Drunen, M., Tukker, A., Rioolbuizen, M. Van. 2000. Life cycle of wastewater pipes made from PVC, concrete and HD-PE. *Pub: Institute for Environmental Studies (IVM) and TNO-STB, DELFT, Netherlands*.

EA. The Environment Agency. 2008. *Greenhouse gas emissions of water supply and demand management options. Science Report - SC070010*. Environment Agency, Bristol, UK.

EA. The Environment Agency. 2010. *Energy and carbon implications of rainwater harvesting and greywater recycling. Science Report SC090018*.

Earles, J.M. and A. Halog. 2011. Consequential life cycle assessment: a review. *The International Journal of Life Cycle Assessment* 16(5): 445–453. http://link.springer.com/10.1007/s11367-011-0275-9. Accessed November 19, 2014.

EC-JRC. 2010a. *International Reference Life Cycle Data System (ILCD) handbook: General guide for Life Cycle Assessment - Detailed guidance*. Luxembourg: European Commission - Joint Research Centre - Institute for Environment and Sustainability. Publications Office of the European Union.

EC-JRC. 2010b. *Framework and Requirements for Life Cycle Impact Assessment Models and Indicators. International Reference Life Cycle Data System (ILCD) Handbook*. http://eplca.jrc.ec.europa.eu/uploads/ILCD-Handbook-LCIA-Framework-Requirements-ONLINE-March-2010-ISBN-fin-v1.0-EN.pdf%5Cnhttp://eplca.jrc.ec.europa.eu/.

EEA. 1999. *Environmental indicators : Typology and overview. European Environment Agency*. Vol. 25.

Eggimann, S., B. Truffer, and M. Maurer. 2015. To connect or not to connect? Modelling the optimal degree of centralisation for wastewater infrastructures. *Water Research* 84: 218–231. http://linkinghub.elsevier.com/retrieve/pii/S004313541530107X. Accessed March 17, 2017.

Ehrenfeld, J.R. 2005. Eco-efficiency Philosophy, Theory, and Tools. *Journal of Industrial Ecology* 9(4): 6–8.

Eijo-Río, E., A. Petit-Boix, G. Villalba, M.E. Suárez-Ojeda, D. Marin, M.J. Amores, X. Aldea, J. Rieradevall, and X. Gabarrell. 2015. Municipal sewer networks as sources of nitrous oxide, methane and hydrogen sulphide emissions: A review and case studies. *Journal of Environmental Chemical Engineering* 3(3): 2084–2094. http://www.sciencedirect.com/science/article/pii/S2213343715001773. Accessed August 24, 2015.

Ekins, P. 2005. Eco-efficiency: Motives, Drivers, and Economic Implications. *Journal of Industrial Ecology* 9(4): 12–15.

El-Fadel, M. and M. Massoud. 2001. Methane emissions from wastewater management. *Environmental Pollution* 114(2): 177–85. http://www.ncbi.nlm.nih.gov/pubmed/11504340.

EN 1401-1:2009. *Plastic piping systems for non-pressure underground drainage and sewerage. Unplasticized poly(vinyl chloride) (PVC-U). Specifications for pipes, fittings and the system*. British Standards Institution.

EN 15804:2011. Sustainability of construction works - Environmental product declarations - Core rules for the product category of construction works. European Standard. Brussels, Belgium: European Committee for Standardization.

EN 1916:2002. *Concrete pipes and fitting, unreinforced, steel fibre and reinforced*. AENOR, Madrid, Spain.

ESRI (Environmental Systems Resource Institute). 2016. ArcGIS Desktop: Release 10.4. *Redlands CA.*

European Commission. 2013. *Building a Green Infrastructure for Europe*. European Union ISBN 978-92-79-33428-3.

European Commission. Joint Research Centre (JRC)/ Netherlands Environmental Assessment Agency (PBL). 2009. Emission Database for Global Atmospheric Research (EDGAR), release version 4.0. http://edgar.jrc.ec.europa.eu.

European Council. 2007. Directive 2007/60/EC of the European Parliament and of the Council of 23 October 2007 on the assessment and management of flood risks.

Eurostat. 2015. Water statistics on national level. http://ec.europa.eu/eurostat/data/database.

Farmani, R. and D. Butler. 2014. Implications of Urban Form on Water Distribution Systems Performance. *Water Resources Management* 28: 83–97.

Farreny, R., X. Gabarrell, and J. Rieradevall. 2011. Cost-efficiency of rainwater harvesting strategies in dense Mediterranean neighbourhoods. *Resources, Conservation and Recycling* 55(7): 686–694. http://linkinghub.elsevier.com/retrieve/pii/S0921344911000103. Accessed July 27, 2014.

Fenner, R.A. 2000. Approaches to sewer maintenance: a review. *Urban Water* 2: 343–356.

Feo, G. de, G. Antoniou, H.F. Fardin, F. El-Gohary, X.Y. Zheng, I. Reklaityte, D. Butler, S. Yannopoulos, and A.N. Angelakis. 2014. The historical development of sewers worldwide. *Sustainability (Switzerland)*.

Filion, Y.R. 2008. Impact of Urban Form on Energy Use in Water Distribution Systems. *J. Infrastruct. Syst.* 14(December): 337–346.

FLOODsite. 2008. Intensity of rainfall. http://www.floodsite.net/juniorfloodsite/html/en/student/thingstoknow/hydrology/rainfallintensity.html. Accessed February 19, 2015.

Flynn, K.M. and R.G. Traver. 2013. Green infrastructure life cycle assessment: A bio-infiltration case study. *Ecological Engineering* 55: 9–22. http://linkinghub.elsevier.com/retrieve/pii/S0925857413000190. Accessed July 9, 2014.

Foley, J. and P. Lant. 2009. Direct Methane and Nitrous Oxide emissions from full-scale wastewater treatment systems. *Water Services Association of Australia Ltd*(24).

Fragkou, M.C., T. Vicent, and X. Gabarrell. 2016. An ecosystemic approach for assessing the urban water self-sufficiency potential: lessons from the Mediterranean. *Urban Water Journal* 13(7): 663–675.

Friedrich, E., S. Pillay, and C.A. Buckley. 2009. Carbon footprint analysis for increasing water supply and sanitation in South Africa: a case study. *Journal of Cleaner Production* 17(1): 1–12. http://linkinghub.elsevier.com/retrieve/pii/S0959652608000486. Accessed January 31, 2013.

Frischknecht, R., N. Jungbluth, H.-J. Althaus, G. Doka, R. Dones, T. Heck, S. Hellweg, et al. 2005. The ecoinvent database: Overview and methodological framework. *International Journal of Life Cycle Assessment* 10: 3–9.

Frosch, R.A. 1992. Industrial ecology: a philosophical introduction. *Proceedings of the National Academy of Sciences of the United States of America* 89(3): 800–3.

http://www.ncbi.nlm.nih.gov/pubmed/11607255. Accessed February 1, 2017.

Fuchs, V.J., J.R. Mihelcic, and J.S. Gierke. 2011. Life cycle assessment of vertical and horizontal flow constructed wetlands for wastewater treatment considering nitrogen and carbon greenhouse gas emissions. *Water Research* 45(5): 2073–81. http://www.sciencedirect.com/science/article/pii/S0043135410008687. Accessed April 14, 2016.

Gabarrell, X., J. Rieradevall, and C. Gasol. 2013. LCADB.sudoe. Base de datos para la realización de Análisis del Ciclo de Vida para la región Sudoe. Register of Community Trade Marks Nº: 010999373.

Gallego, A., A. Hospido, M.T. Moreira, and G. Feijoo. 2008. Environmental performance of wastewater treatment plants for small populations. *Resources, Conservation and Recycling* 52(6): 931–940. http://www.sciencedirect.com/science/article/pii/S092134490800030X. Accessed April 12, 2016.

Garcia-Solsona, E., J. Garcia-Orellana, P. Masqué, E. Garcés, O. Radakovitch, A. Mayer, S. Estradé, and G. Basterretxea. 2010. An assessment of karstic submarine groundwater and associated nutrient discharge to a Mediterranean coastal area (Balearic Islands, Spain) using radium isotopes. *Biogeochemistry* 97(2–3): 211–229. http://link.springer.com/10.1007/s10533-009-9368-y. Accessed March 17, 2017.

Garner, A. and G.A. Keoleian. 1995. *Industrial ecology: an introduction. Ann Arbor, Michigan: National Pollution Prevention Center for Higher Education, University of Michigan*.

Getter, K.L., D.B. Rowe, and J. a. Andresen. 2007. Quantifying the effect of slope on extensive green roof stormwater retention. *Ecological Engineering* 31(4): 225–231. http://linkinghub.elsevier.com/retrieve/pii/S0925857407001309. Accessed July 10, 2014.

Getter, K.L., D.B. Rowe, G.P. Robertson, B.M. Cregg, and J. a Andresen. 2009. Carbon sequestration potential of extensive green roofs. *Environmental Science & Technology* 43(19): 7564–70. http://www.ncbi.nlm.nih.gov/pubmed/19848177.

Gikas, P. and G. Tchobanoglous. 2009. Sustainable use of water in the Aegean Islands. *Journal of Environmental Management* 90(8): 2601–11. http://www.sciencedirect.com/science/article/pii/S0301479709000243. Accessed March 16, 2016.

GISAgua. 2012. ©Aqualogy Services Company.

Gleick, P. 1996. Basic water requirements for human activities: meeting basic needs. *Water International* 21: 83–92.

Goedkoop, M., R. Heijungs, and R. Huijbregts M, De Schryver A, Struijs J, Van Zelm. 2009. *ReCiPe 2008, A Life Cycle Impact Assessment Method Which Comprises Harmonised Category Indicators at the Midpoint and the Endpoint Level. Report I: Characterisation*. First Ed. Available at: http://www.lcia-recipe.net.

Google Maps. 2015. Res. Damha I - Google Maps. https://www.google.es/maps/place/Res.+Damha+I,+SP,+Brazil/@-21.9642108,-47.9018604,995m/data=!3m2!1e3!4b1!4m2!3m1!1s0x94b8705d8cf52391:0xd2a66e30c1e9ecea7.9018604,995m/data=!3m2!1e3!4b1!4m2!3m1!1s0x94b8705d8cf52391:0xd2a66e30c1e9ecea. Accessed February 19, 2015.

Gössling, S., P. Peeters, C.M. Hall, J.-P. Ceron, G. Dubois, L.V. Lehmann, and D. Scott. 2012. Tourism and water use: Supply, demand, and security. An international review. *Tourism Management* 33(1): 1–15. http://www.sciencedirect.com/science/article/pii/S0261517711000793. Accessed August 14, 2015.

Goverment of the Balearic Islands. 2016. Agreement of the Council of Government issued on September 23, 2016, whereby the Council of Government of the Balearic Islands ratifies the Annual Plan for Sustainable Tourism Promotion for the period 2016. *Butlletí Oficial de Les Illes Balears* 122: 28893–28897.

Government of the Balearic Islands. 2015. *Hydrologic plan of the Balearic Islands (Pla Hidrològic de les Illes Balears, Real Decret 701/2015)*.

Graedel, T.E. 1996. On the concept of industrial ecology. *Annual Review of Energy and the Environment* 21(1): 69–98. http://www.annualreviews.org/doi/10.1146/annurev.energy.21.1.69. Accessed February 1, 2017.

Griffiths-Sattenspiel, B. and W. Wilson. 2009. *The Carbon Footprint of Water*.

Guha-Sapir, D., R. Below, and P. Hoyois. 2009. EM-DAT: International Disaster Database. Université Catholique de Louvain – Brussels – Belgium. www.emdat.be Accessed December 2014.

Guinée JB, Gorrée M, Heijungs R, Huppes G, Kleijn R, Koning A de, Oers L van, Wegener Sleeswijk A, Suh S, Udo de Haes HA, Bruijn H de, Duin R van, H.M. 2002. *Handbook on life cycle assessment. Operational guide to the ISO standards. I: LCA in perspective. IIa: Guide. IIb: Operational annex. III: Scientific background*. Kluwer Academic Publishers, ISBN 1-4020-0228-9, Dordrecht, 692 pp.

Guisasola, A., D. de Haas, J. Keller, and Z. Yuan. 2008. Methane formation in sewer systems. *Water Research* 42(6–7): 1421–30. http://www.ncbi.nlm.nih.gov/pubmed/17988709. Accessed February 12, 2013.

Guisasola, A., K.R. Sharma, J. Keller, and Z. Yuan. 2009. Development of a model for assessing methane formation in rising main sewers. *Water Research* 43(11): 2874–84. http://www.ncbi.nlm.nih.gov/pubmed/19423146. Accessed December 23, 2013.

Gutierrez, O., D. Park, K.R. Sharma, and Z. Yuan. 2009. Effects of long-term pH elevation on the sulfate-reducing and methanogenic activities of anaerobic sewer biofilms. *Water Research* 43(9): 2549–57. http://www.ncbi.nlm.nih.gov/pubmed/19345393. Accessed December 23, 2013.

Gutierrez, O., G. Sudarjanto, G. Ren, R. Ganigué, G. Jiang, and Z. Yuan. 2014. Assessment of pH shock as a method for controlling sulfide and methane formation in pressure main sewer systems. *Water Research* 48: 569–78. http://www.ncbi.nlm.nih.gov/pubmed/24210545. Accessed July 29, 2014.

Haes, H.A.U., R. Heijungs, S. Suh, and G. Huppes. 2004. Three Strategies to Overcome the Limitations of Life-Cycle Assessment. *Journal of Industrial Ecology* 8(3): 19–32. http://doi.wiley.com/10.1162/1088198042442351. Accessed September 1, 2016.

Haghighi, A. and A.E. Bakhshipour. 2012. Optimization of Sewer Networks Using an Adaptive Genetic Algorithm. *Water Resources Management* 26(12): 3441–3456. http://link.springer.com/10.1007/s11269-012-0084-3. Accessed August 29, 2014.

Hawkins, T.R., B. Singh, G. Majeau-Bettez, and A.H. Strømman. 2013. Comparative Environmental

Life Cycle Assessment of Conventional and Electric Vehicles. *Journal of Industrial Ecology* 17(1): 53–64. http://doi.wiley.com/10.1111/j.1530-9290.2012.00532.x. Accessed July 10, 2014.

Haynes, H., R. Haynes, and G. Pender. 2008. Integrating socio-economic analysis into decision-support methodology for flood risk management at the development scale (Scotland). *Water and Environment Journal* 22(2): 117–124. http://doi.wiley.com/10.1111/j.1747-6593.2007.00086.x. Accessed November 3, 2016.

Heijungs, R. 2013. Ten easy lessons for good communication of LCA. *International Journal of Life Cycle Assessment*: 1–4.

Hellström, D., U. Jeppsson, and E. Kärrman. 2000. A framework for systems analysis of sustainable urban water management. *Environmental Impact Assessment Review* 20(3): 311–321.

Hertwich, E.G. and J.K. Hammitt. 2001. A decision-analytic framework for impact assessment. *The International Journal of Life Cycle Assessment* 6(5): 265–272. http://link.springer.com/10.1007/BF02978787. Accessed February 13, 2017.

Hischier, R., B. Weidema, H. Althaus, C. Bauer, G. Doka, R. Dones, R. Frischknecht, et al. 2010. Implementation of life cycle impact assessment methods. Final report ecoinvent v2.2 No. 3. Swiss Centre for Life Cycle Inventories, Dübendorf. Swiss Centre for Life Cycle Inventories, Dübendorf.

Hodges, S.H. and J.I. Enyart. 1993. *Standard installation. International Conference on Pipeline Infrastructure*. Texas.

Hospido, A., M.T. Moreira, and G. Feijoo. 2008. A Comparison of Municipal Wastewater Treatment Plants for Big Centres of Population in Galicia (Spain). *International Journal of Life Cycle Assessment* 13(1): 57–64.

Hospido, A., M.T. Moreira, M. Fernández-Couto, and G. Feijoo. 2004. Environmental performance of a municipal wastewater treatment plant. *The International Journal of Life Cycle Assessment* 9(4): 261–271. http://link.springer.com/10.1007/BF02978602.

Hu, H.-Y., Y.-L. Cheng, and J.-Y. Lin. 2007. On-Site Treatment of Septic Tank Effluent by Using a Soil Adsorption System. *Practice Periodical of Hazardous, Toxic, and Radioactive Waste Management* 11(3): 197–206. http://ascelibrary.org/doi/10.1061/%28ASCE%291090-025X%282007%2911%3A3%28197%29. Accessed April 13, 2017.

Huppes, G. and M. Ishikawa. 2005a. Eco-efficiency and Its Terminology. *Journal of Industrial Ecology* 9(4): 43–46.

Huppes, G. and M. Ishikawa. 2005b. A framework for quantified eco-efficiency analysis. *Journal of Industrial Ecology* 9(4): 25–41.

Hvitved-Jacobsen, T. Vollertsen, J. and N. Tanaka. 2000. An integrated aerobic/anaerobic approach for prediction of sulphide formation in sewer. *Water Sci. Technol* 41(6): 107–115.

Hvitved-Jacobsen, T., J. Vollertsen, and A.H. Nielsen. 2002. *Sewer Processes: Microbial and Chemical Process Engineering of Sewer Networks*. 2nd ed. CRC Press.

IBESTAT. Statistical Institute of the Balearic Islands. 2015. Tourism. http://www.ibestat.es/ibestat/estadistiques/economia/turisme/0b70b294-81e0-413a-b7b2-

3cc3a33593a8.

ICGC. Institut Cartogràfic i Geològic de Catalunya. 2016. Cartography and geodesy. http://icgc.cat/.

Idescat. Statistical Institute of Catalonia. 2016. Municipal database. http://www.idescat.cat/territ/BasicTerr?TC=9.

IEA. International Energy Agency. 2014. Spain: Electricity and Heat for 2011. http://www.iea.org/statistics/statisticssearch/report/?country=SPAIN&product=electricityandheat&year=2011. Accessed April 2014.

IECA. Instituto de Estadística y Cartografía de Andalucía. 2010. *Andalusian Municipalities. Basic Data 2010 (Municipios Andaluces. Datos Básicos 2010). Consejería de Economía y Hacienda.*

ieex. Instituto Estadístico de Extremadura. 2011. Socioeconomic atlas of Extremadura 2011 - Summary (Atlas Socioeconómico de Extremadura 2011- Resumen). http://estadistica.gobex.es/ Accessed November 2013.

IGE. Instituto Galego de Estadística. 2009. Primary and secondary distribution of the income per council (Contas de asignación primaria e de distribución secundaria da renda por concellos). http://www.ige.eu/igebdt/esq.jsp?ruta=verEjes.jsp%3FCOD%3D4221%26M%3D%26S%3D%26RET%3D%26ORD%3D&paxina=001&c=0307007004 Accessed November 2013.

IMF. International Monetary Fund. 2015. World Economic Outlook Database. http://www.imf.org/external/pubs/ft/weo/2014/01/weodata/weorept.aspx?sy=2008&ey=2014&scsm=1&ssd=1&sort=country&ds=.&br=1&pr1.x=47&pr1.y=4&c=223&s=PCPIE%252CPCPIEPCH&grp=0&a= Accessed October 2014.

INE. Instituto Nacional de Estadística. 2013. Population and household census, Press notice (Censos de Población y Viviendas 2011, Notas de prensa). http://www.ine.es/prensa/np824.pdf Accessed January 2014.

INE. Spanish Statistical Office. 2014. Air emissions accounting, 2008-2012. http://www.ine.es/jaxi/Tabla.htm?path=/t26/p084/serie/l0/&file=01001.px&L=0.

INE. Spanish Statistical Office. 2015. Spanish National Accounting, 2010. http://www.ine.es/jaxi/menu.do?type=pcaxis&path=/t35/p008/&file=inebase.

INMET. Instituto Nacional de Meteorologia. 2014. Daily accumulated rainfall of Sao Carlos 1961-2014. http://www.inmet.gov.br/portal/ Accessed October 2014.

Institut d'Estudis Catalans and Generalitat de Catalunya. 2016. *Tercer Informe sobre el Canvi Climàtic a Catalunya*.

Insurance Compensation Consortium (Consorcio de Compensación de Seguros). 2015. *Extraordinary risks, Series 1971-2014 (Riesgos extraordinarios, Serie 1971-2014)*. Madrid, Spain.

INTRON. 1995. Environmental profile and environmental measures of a concrete external sewer [Intron report No. 95027] and Environmental profile and environmental measures of an external sewer of PVC and vitrified clay in comparison to concrete [Intron report No. 95195]. *INTRON, Netherlands*.

IPMet. Instituto de Pesquisas Meteorológicas. 2014. Natural Disaster Database (Banco de Dados de

Desastres Naturais). http://www.ipmet.unesp.br/index2.php?abre=ipmet_html/defesa_civil/index.php Accessed January 2015.

ISO 14040:2006. Environmental management - Life cycle assessment - Principles and framework. Geneva, Switzerland: International Organization for Standardization.

ISO 14044:2006. Environmental management - Life cycle assessment - Requirements and guidelines. Geneva, Switzerland: International Organization for Standardization.

ISO 14045:2012. Environmental management — Ecoefficiency assessment of product systems — Principles, requirements and guidelines. Geneva, Switzerland: International Organization for Standardization.

ISO 15686-5:2008. Buildings and constructed assets - Sevice-life planning - Part 5: Life-cycle costing. Geneva, Switzerland: International Organization for Standardization.

Jha, A., R. Bloch, and J. Lamond. 2012. *Cities and Flooding. A Guide to Integrated Urban Flood Risk Management for the 21st Century*. The World Bank, Washington, D.C.

Jha, A., J. Lamond, R. Bloch, N. Bhattacharya, A. Lopez, N. Papachristodoulou, A. Bird, D. Proverbs, J. Davies, and R. Barker. 2011. *Five Feet High and Rising. Cities and Flooding in the 21st Century. Policy Research Working Paper*. http://www-wds.worldbank.org/external/default/WDSContentServer/IW3P/IB/2011/05/03/000158349_20110503095951/Rendered/PDF/WPS5648.pdf. Accessed January 15, 2016.

Jiang, F., D.H.-W. Leung, S. Li, G.-H. Chen, S. Okabe, and M.C.M. van Loosdrecht. 2009a. A biofilm model for prediction of pollutant transformation in sewers. *Water Research* 43(13): 3187–98. http://www.ncbi.nlm.nih.gov/pubmed/19487008. Accessed July 30, 2014.

Jiang, G., O. Gutierrez, K.R. Sharma, and Z. Yuan. 2010. Effects of nitrite concentration and exposure time on sulfide and methane production in sewer systems. *Water Research* 44(14): 4241–51. http://www.ncbi.nlm.nih.gov/pubmed/20554309. Accessed February 12, 2013.

Jiang, G., K.R. Sharma, A. Guisasola, J. Keller, and Z. Yuan. 2009b. Sulfur transformation in rising main sewers receiving nitrate dosage. *Water Research* 43: 4430–4440.

Jin, N.J., H.G. Hwang, and J.H. Yeon. 2013. Structural analysis and optimum design of GRP pipes based on properties of materials. *Construction and Building Materials* 38: 316–326. http://linkinghub.elsevier.com/retrieve/pii/S0950061812006411. Accessed September 1, 2014.

Jolliet, O., R. Müller-Wenk, J. Bare, A. Brent, M. Goedkoop, R. Heijungs, N. Itsubo, et al. 2004. The LCIA midpoint-damage framework of the UNEP/SETAC life cycle initiative. *The International Journal of Life Cycle Assessment* 9(6): 394–404. http://link.springer.com/10.1007/BF02979083. Accessed April 5, 2017.

Jonkman, S.N., M. Bočkarjova, M. Kok, and P. Bernardini. 2008. Integrated hydrodynamic and economic modelling of flood damage in the Netherlands. *Ecological Economics* 66(1): 77–90.

Junta d'Aigües. 1992. *Master plan for flood protection in Maresme (Pla Director de protecció contra avingudes al Maresme)*.

Kadlec, R.H. and S.D. Wallace. 2009. Treatment Wetlands, Second Edition. In *Treatment Wetlands,* *Second Edition*, 965.

Kägi, T., F. Dinkel, R. Frischknecht, S. Humbert, J. Lindberg, S. De Mester, T. Ponsioen, S. Sala, and U.W. Schenker. 2016. Session "Midpoint, endpoint or single score for decision-making?"—SETAC Europe 25th Annual Meeting, May 5th, 2015. *The International Journal of Life Cycle Assessment* 21(1): 129–132. http://link.springer.com/10.1007/s11367-015-0998-0. Accessed August 18, 2016.

Kampschreur, M.J., H. Temmink, R. Kleerebezem, M.S.M. Jetten, and M.C.M. van Loosdrecht. 2009. Nitrous oxide emission during wastewater treatment. *Water Research* 43(17): 4093–103. http://www.ncbi.nlm.nih.gov/pubmed/19666183. Accessed December 11, 2013.

Kennedy, C.A. 2016. Industrial Ecology and Cities. In *Taking Stock of Industrial Ecology*, 69–86. Cham: Springer International Publishing. http://link.springer.com/10.1007/978-3-319-20571-7_4. Accessed February 1, 2017.

Kennedy, C., L. Baker, S. Dhakal, and A. Ramaswami. 2012. Sustainable Urban Systems: An Integrated Approach. *Journal of Industrial Ecology*.

Kennedy, C., J. Cuddihy, and J. Engel-Yan. 2007. The Changing Metabolism of Cities. *Journal of Industrial Ecology* 11(2): 43–59. http://doi.wiley.com/10.1162/jie.2007.1107. Accessed February 1, 2017.

Kennedy, C., S. Pincetl, and P. Bunje. 2010. The study of urban metabolism and its applications to urban planning and design. *Environmental Pollution* 159(8–9): 1965–73. http://www.ncbi.nlm.nih.gov/pubmed/21084139.

Kirtman, B., S.B. Power, J.A. Adedoyin, G.J. Boer, R. Bojariu, I. Camilloni, F.J. Doblas-Reyes, et al. 2013. Near-term Climate Change: Projections and Predictability. *Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*: 953–1028.

Klein, J.J.M. de and A.K. van der Werf. 2014. Balancing carbon sequestration and GHG emissions in a constructed wetland. *Ecological Engineering* 66: 36–42.

Klein G, Krebs M, Hall V, O'Brien T, B.B. 2005. *California â€™ s Water – Energy Relationship. Final Staff Report*.

Knoeri, C., E. Sanyé-Mengual, and H. Althaus. 2013. Comparative LCA of recycled and conventional concrete for structural applications. *International Journal of Life Cycle Assessment* (online)(DOI 10.1007/s11367-012-0544-2).

Kosareo, L. and R. Ries. 2007. Comparative environmental life cycle assessment of green roofs. *Building and Environment* 42(7): 2606–2613. http://linkinghub.elsevier.com/retrieve/pii/S0360132306001648. Accessed July 10, 2014.

Kousky, C. and S.H. Schneider. 2003. Global climate policy: Will cities lead the way? *Climate Policy* 3(4): 359–372.

Kubal, C., D. Haase, V. Meyer, and S. Scheuer. 2009a. Integrated urban flood risk assessment – adapting a multicriteria approach to a city. *Natural Hazards and Earth System Science* 9(6): 1881–1895.

Kubal, C., D. Haase, V. Meyer, and S. Scheuer. 2009b. Integrated urban flood risk assessment – adapting a multicriteria approach to a city. *Natural Hazards and Earth System Science* 9(6): 1881–1895. http://www.nat-hazards-earth-syst-sci.net/9/1881/2009/. Accessed November 3, 2016.

Kundzewicz, Z.W., L.J. Mata, N.W. Arnell, P. Döll, P. Kabat, B. Jiménez, K.A. Miller, T. Oki, Z. Sen, and I.A. Shiklomanov. 2007. *Freshwater resources and their management. Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, M.L. Parry, O.F. Canziani, J.P. Palut*. Cambridge University Press, Cambridge, UK, 173-210.

La Fuente, A. De, R.C. Escariz, A.D. De Figueiredo, and A. Aguado. 2013. Design of macro-synthetic fibre reinforced concrete pipes. *Construction and Building Materials* 43: 523–532.

la Fuente A, Escariz RC, de Figueiredo AD, Molins C, Aguado A, de. 2012. A new design method for steel fibre reinforced concrete pipes. *Construction and Building Materials* 30: 547–555. http://www.sciencedirect.com/science/article/pii/S0950061811006921. Accessed August 29, 2014.

Lahav, O., A. Sagiv, and E. Friedler. 2006. A different approach for predicting H2S(g) emission rates in gravity sewers. *Water Research* 40(2): 259–66. http://www.ncbi.nlm.nih.gov/pubmed/16343590. Accessed February 14, 2013.

Lassaux, S., R. Renzoni, and A. Germain. 2007. Life Cycle Assessment of Water: from the Pumping Station to the Wastewater Treatment Plant. *International Journal* 12(2): 118–126. http://www.csa.com/partners/viewrecord.php?requester=gs&collection=TRD&recid=08130351EN.

Lee, C.C. 2005. *Environmental engineering dictionary*. Government Institutes. https://books.google.es/books?id=f1lnQwrvOSEC&pg=PA81&lpg=PA81&dq=methods+that+have+been+determined+to+be+the+most+effective,+practical+means+of+preventing+or+reducing+pollution+from+nonpoint+source&source=bl&ots=LRW-FRguzI&sig=YLpH84PvFW0qeUzT0ppCuM_4cdU. Accessed April 27, 2017.

Lee, J.Y., H.J. Moon, T.I. Kim, H.W. Kim, and M.Y. Han. 2013. Quantitative analysis on the urban flood mitigation effect by the extensive green roof system. *Environmental Pollution (Barking, Essex : 1987)* 181: 257–61. http://www.ncbi.nlm.nih.gov/pubmed/23892044. Accessed July 16, 2014.

Leeuwen, C.J. van, J. Frijns, A. van Wezel, and F.H.M. van de Ven. 2012. Cities Blueprints: 24 Indicators to Assess the Sustainability of the Urban Water Cycle. *Water Resources Management* 26(8): 2177–2197. http://link.springer.com/10.1007/s11269-012-0009-1. Accessed July 30, 2014.

Lehtoranta, S., R. Vilpas, and T.J. Mattila. 2014. Comparison of carbon footprints and eutrophication impacts of rural on-site wastewater treatment plants in Finland. *Journal of Cleaner Production* 65: 439–446. http://www.sciencedirect.com/science/article/pii/S0959652613005647. Accessed February 5, 2016.

Lemos, D., A.C. Dias, X. Gabarrell, and L. Arroja. 2013. Environmental assessment of an urban water system. *Journal of Cleaner Production* 54: 157–165. http://linkinghub.elsevier.com/retrieve/pii/S0959652613002643. Accessed July 30, 2014.

Listowski, A., H.H. Ngo, W.S. Guo, S. Vigneswaran, H.S. Shin, and H. Moon. 2011. Greenhouse Gas (GHG) Emissions from Urban Wastewater System : Future Assessment Framework and Methodology. *Journal of Water Sustainability* 1(1): 113–125.

Lleonart, I. and X. Tarruella. 2010. *Municipal urban plan of Cabrils - Preliminary sustainability report. http://www.cabrils.cat/pdf/poum/av/textes/IAP%20POUM%20CABRILS_280210.pdf*.

Llopart-Mascaró A, Farreny, R, Gabarrell, X, Rieradevall, J, Gil, A, Martínez, M, Puertas, J, Suárez, J., H. Río, and M. Paraira. 2015. Storm tank against combined sewer overflow: Operation strategies to minimise discharges impact to receiving waters. *Urban Water Journal* 12(3): 219–228. http://dx.doi.org/10.1080/1573062X.2013.868499. Accessed August 1, 2014.

Lorenzo-Toja, Y., C. Alfonsín, M.J.M.J. Amores, X. Aldea, D. Marin, M.T.M.T. Moreira, G. Feijoo, D. Marín-Navarro, M.T.M.T. Moreira, and G. Feijoo. 2016a. Beyond the conventional life cycle inventory in wastewater treatment plants. *Science of Total Environment* 553: 71–82.

Lorenzo-Toja, Y., I. Vázquez-Rowe, M.J. Amores, M. Termes-Rifé, D. Marín-Navarro, M.T. Moreira, and G. Feijoo. 2016b. Benchmarking wastewater treatment plants under an eco-efficiency perspective. *Science of The Total Environment* 566: 468–479.

Lorenzo-Toja, Y., I. Vázquez-Rowe, S. Chenel, D. Marín-Navarro, M.T. Moreira, and G. Feijoo. 2015. Eco-efficiency analysis of Spanish WWTPs using the LCA+DEA method. *Water Research* 68: 651–666. http://www.sciencedirect.com/science/article/pii/S0043135414007386. Accessed November 3, 2014.

Lotito, A.D., P. Wunderlin, A. Joss, M. Kipf, and H. Siegrist. 2012. Nitrous oxide emissions from the oxidation tank of a pilot activated sludge plant. *Water Research* 46: 3563–3573.

Loubet, P., P. Roux, E. Loiseau, and V. Bellon-Maurel. 2014. Life cycle assessments of urban water systems: A comparative analysis of selected peer-reviewed literature. *Water Research* 67: 187–202. http://www.sciencedirect.com/science/article/pii/S004313541400623X. Accessed September 19, 2014.

Lü, Y., S. Liu, and B. Fu. 2012. Ecosystem Service: From Virtual Reality to Ground Truth. *Environmental Science & Technology* 46(7): 4255–4255. http://dx.doi.org/10.1021/es300911q. Accessed June 26, 2015.

Lucas, A.H. 2011. Monitoramento e modelagem de um sistema filtro-vala-trincheira de infiltração em escala real. PhD Thesis. Universidade Federal de São Carlos.

Lucas, A.H., A.P. Barbassa, and R.B. Moruzzi. 2013. Modelagem de um sistema filtro-vala-trincheira de infiltração pelo método de Puls adaptado para calibração de parâmetros. *Revista Brasileira de Recursos Hídricos* 18: 135–236.

Lundin, M., M. Bengtsson, and S. Molander. 2000. Life Cycle Assessment of Wastewater Systems: Influence of System Boundaries and Scale on Calculated Environmental Loads. *Environmental Science & Technology* 34(1): 180–186. http://dx.doi.org/10.1021/es990003f. Accessed March 23, 2015.

Lundström, H., Ölund, G., RydbergG, T. 1996. Life cycle assessment of a concrete sewage pipe. *CIT Ekologik, The Nordic Project on Sustainable Concrete Technology. Chalmers Industriteknik*.

Maa, C., M.T. Ju, X.C. Zhang, and H.Y. Li. 2011. Energy consumption and carbon emissions in a coastal city in China. In *Procedia Environmental Sciences*, 4:1–9.

Mahgoub, E.-S.M.M., N.P. van der Steen, K. Abu-Zeid, and K. Vairavamoorthy. 2010. Towards sustainability in urban water: a life cycle analysis of the urban water system of Alexandria City, Egypt. *Journal of Cleaner Production* 18(10–11): 1100–1106. http://linkinghub.elsevier.com/retrieve/pii/S0959652610000478. Accessed July 30, 2014.

Makropoulos, C.K., K. Natsis, S. Liu, K. Mittas, and D. Butler. 2008. Decision support for sustainable option selection in integrated urban water management. *Environmental Modelling & Software* 23(12): 1448–1460.

Malgrat, P., J.M. Verdejo, and A. Vilalta. 2004. Los depósitos de retención de aguas pluviales de Barcelona. *Tecno Ambiente* 144: 13–17.

Marques, R.C., N.F. da Cruz, and J. Pires. 2015. Measuring the sustainability of urban water services. *Environmental Science & Policy* 54: 142–151.

Marston, A. 1930. *The theory of external loads on closed conduits in the light of the latest experiments. Bulletin 96*. Vol. Iowa Engin. Bulletin 96, Iowa Engineering Experiments Station, Ames, Iowa.

McDonald, R.I. 2008. Global urbanization: Can ecologists identify a sustainable way forward? *Frontiers in Ecology and the Environment*.

McDonnell, M.J. and S.T.A. Pickett. 1990. Ecosystem structure and function along urban-rural gradients: an unexploited opportunity for ecology. *Ecology*.

McGrath, T.J., R.E. Chambers, and P.A. Sharff. 1990. Recent trends in installation standards for plastic pipe. Buried plastic pipe technology. *ASTM STP 1093*: 281–293.

Meehl, G.A., T.F. Stocker, W.D. Collins, P. Friedlingstein, A.T. Gaye, J.M. Gregory, A. Kitoh, et al. 2007. *Global Climate Projections. In: Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. A*. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Mendoza, J.-M., J. Oliver-Solà, X. Gabarrell, and A. Rieradevall J, Josa. 2012. Planning strategies for promoting environmentally suitable pedestrian pavements in cities. *Transportation Research Part D: Transport and Environment* 17(6): 442–450. http://linkinghub.elsevier.com/retrieve/pii/S1361920912000533. Accessed May 23, 2013.

Mentens, J., D. Raes, and M. Hermy. 2006. Green roofs as a tool for solving the rainwater runoff problem in the urbanized 21st century? *Landscape and Urban Planning* 77(3): 217–226. http://linkinghub.elsevier.com/retrieve/pii/S0169204605000496. Accessed July 9, 2014.

MetaBase ITeC. 2010. Online ITeC database: prices, technical details, companies, certificates, product pictures and environmental data. http://www.itec.cat/metabase. Accessed February 2013.

Metcalf and Eddy. Inc. 1991. *Wastewater Engineering: Treatment, Disposal and Reuse*. 3rd ed. McGraw-Hill, New York.

Meteocat. Servei Meteorològic de Catalunya. 2014. Automatic stations (Estacions automàtiques). http://www.meteo.cat/xema/AppJava/Mapper.do Accessed April 2014.

Meteogalicia. 2014. Estacións meteorolóxicas. http://www2.meteogalicia.es/galego/observacion/estacions/estacionsHistorico.asp?Nest=19045&prov=Pontevedra&red=102109&tiporede=agro&idprov=3#. Accessed July 1, 2014.

Mohanakrishnan, J., O. Gutierrez, R.L. Meyer, and Z. Yuan. 2008. Nitrite effectively inhibits sulfide and methane production in a laboratory scale sewer reactor. *Water Research* 42(14): 3961–71. http://www.ncbi.nlm.nih.gov/pubmed/18675440. Accessed December 23, 2013.

Morera, S., C. Remy, J. Comas, and L. Corominas. 2016. Life cycle assessment of construction and renovation of sewer systems using a detailed inventory tool. *The International Journal of Life Cycle Assessment* 21(8): 1121–1133. http://link.springer.com/10.1007/s11367-016-1078-9. Accessed August 3, 2016.

Moser, A.P. 2001. *Buried Plastic Pipe Technology, Issue 1093*. ASTM International. http://books.google.com/books?id=nTf6qPe3lBgC&pgis=1. Accessed August 29, 2014.

Muga, H.E. and J.R. Mihelcic. 2008. Sustainability of wastewater treatment technologies. *Journal of Environmental Management* 88(3): 437–447.

Müller, D.K., C.M. Hall, and D. Keen. 2004. Second home tourism impact, planning and management.: 15–32. http://www.cabdirect.org/abstracts/20043135949.html;jsessionid=F0459400AE8D3561B114D6E7F43F1E06. Accessed April 13, 2016.

Muñoz, I., L. Milà-I-Canals, and A.R. Fernández-Alba. 2010. Life Cycle Assessment of Water Supply Plans in Mediterranean Spain: The Ebro River Transfer Versus the AGUA Programme. *Journal of Industrial Ecology* 14: 902–918.

Murla, D., O. Gutierrez, M. Martinez, D. Suñer, P. Malgrat, and M. Poch. 2016. Coordinated management of combined sewer overflows by means of environmental decision support systems. *Science of The Total Environment* 550: 256–264.

Nielsen, A.H., P. Lens, J. Vollertsen, and T. Hvitved-Jacobsen. 2005. Sulfide-iron interactions in domestic wastewater from a gravity sewer. *Water Research* 39(12): 2747–55. http://www.ncbi.nlm.nih.gov/pubmed/15978649. Accessed July 30, 2014.

Nielsen, A.H., J. Vollertsen, H.S. Jensen, T. Wium-Andersen, and T. Hvitved-Jacobsen. 2008. Influence of pipe material and surfaces on sulfide related odor and corrosion in sewers. *Water Research* 42: 4206–4214.

Nielsen, C.B., J. Schaumann, S. Hvorslev, and P. Sejersen. 1998. *Sewage pipe components in PVC, PP, HDPE and concrete (Afløbskomponenter af PVC, PP, HDPE og beton)*. Danish EPA, Denmark.

Nielsen, P.H., K. Raunkjaer, and T. Hvitved-Jacobsen. 1992. Transformation of wastewater in sewer system—a review. *Water Sci. Technol* 25(6): 17–31.

North, P. 2010. Unsustainable urbanism? Cities, climate change and resource depletion: A Liverpool case study. *Geography Compass*.

Ntelekos, A.A., M. Oppenheimer, J.A. Smith, and A.J. Miller. 2010. Urbanization, climate change and flood policy in the United States. *Climatic Change* 103(3–4): 597–616. http://link.springer.com/10.1007/s10584-009-9789-6. Accessed July 30, 2014.

OBSAM. 2015. *Inidicadors bàsics. Qualitat de l'aigua dels aqüífers. Percentatge de pous que superen els llindars de clorurs i nitrats http://www.obsam.cat/indicadors/medi-fisic/aigua/qualitat/qualitat-aigua-aquifers-1997-2013.pdf.*

Olcina, J. 1994. *Tormentas y granizadas en las tierras alicantinas. Instituto Universitario de Geografía, Alicante (Spain).*

Olcina, J. and A.M. Rico. 2000. Estudios sobre lluvias torrenciales e inundaciones en la Provincia de Alicante (1982-1999). *Universidad de Alcalá. Servicio de Publicaciones* 9: 71–92.

Oliver-Solà, J., X. Gabarrell, and J. Rieradevall. 2009. Environmental impacts of natural gas distribution networks within urban neighborhoods. *Applied Energy* 86(10): 1915–1924.

Opschoor, H. 2011. Local sustainable development and carbon neutrality in cities in developing and emerging countries. *International Journal of Sustainable Development & World Ecology* 18(3): 190–200. http://www.tandfonline.com/doi/abs/10.1080/13504509.2011.570800.

Osada, T., K. Kuroda, and M. Yonaga. 1995. Reducing nitrous oxide gas emissions from fill-and-draw type activated sludge process. *Water Research* 29(6): 1607–1608.

Othman, F. and S. Mortezania. 2010. Efficiency assessment of operational and maintenance techniques to optimize sewer gas amount. *International Journal of the Physical Sciences* 5(9): 1381–1388.

Pacione, M. 2009. *Urban geography: a global perspective*. 3rd ed. London: Routledge.

Pan, T., X.-D. Zhu, and Y.-P. Ye. 2011. Estimate of life-cycle greenhouse gas emissions from a vertical subsurface flow constructed wetland and conventional wastewater treatment plants: A case study in China. *Ecological Engineering* 37(2): 248–254. http://www.sciencedirect.com/science/article/pii/S0925857410003083. Accessed April 14, 2016.

Park, K.Y., Y. Inamori, M. Mizuochi, and K.H. Ahn. 2000. Emission and control of nitrous oxide from a biological wastewater treatment system with intermittent aeration. *Journal of Bioscience and Bioengineering* 90(3): 247–52. http://www.ncbi.nlm.nih.gov/pubmed/16232852.

Pearson, L.J., A. Coggan, W. Proctor, and T.F. Smith. 2009. A Sustainable Decision Support Framework for Urban Water Management. *Water Resources Management* 24(2): 363–376. http://link.springer.com/10.1007/s11269-009-9450-1. Accessed July 30, 2014.

Petit-Boix, A., A. Arahuetes, A. Josa, J. Rieradevall, and X. Gabarrell. 2017. Are we preventing flood damage eco-efficiently? An integrated method applied to post-disaster emergency actions. *Science of The Total Environment* 580: 873–881.

Petit-Boix, A., N. Roigé, A. de la Fuente, P. Pujadas, X. Gabarrell, J. Rieradevall, and A. Josa. 2016. Integrated Structural Analysis and Life Cycle Assessment of Equivalent Trench-Pipe Systems for Sewerage. *Water Resources Management* 30(3): 1117–1130.

Petit-Boix, A., D. Sanjuan-Delmás, S. Chenel, D. Marín, C. Gasol, R. Farreny, G. Villalba, et al. 2015a. Assessing the energetic and environmental impacts of the operation and maintenance of Spanish sewer networks from a life-cycle perspective. *Water Resources Management* 29(8): 2581–2597.

Petit-Boix, A., D. Sanjuan-Delmás, C. Gasol, G. Villalba, M. Suárez-Ojeda, X. Gabarrell, A. Josa, and J. Rieradevall. 2014. Environmental Assessment of Sewer Construction in Small to Medium Sized Cities Using Life Cycle Assessment. *Water Resources Management* 28(4): 979–997. http://link.springer.com/10.1007/s11269-014-0528-z. Accessed April 29, 2014.

Petit-Boix, A., E. Sevigné-Itoiz, L.A. Rojas-Gutierrez, A.P. Barbassa, A. Josa, J. Rieradevall, and X. Gabarrell. 2015b. Environmental and economic assessment of a pilot stormwater infiltration system for flood prevention in Brazil. *Ecological Engineering* 84: 194–201. http://www.sciencedirect.com/science/article/pii/S0925857415301427. Accessed September 21, 2015.

Peyvandi, A., P. Soroushian, and S. Jahangirnejad. 2014. Structural design methodologies for concrete pipes with steel and synthetic fiber reinforcement. *ACI Structural Journal* 111(1): 83–91.

Pons, X. 2000. MiraMon v7.1: Geographic Information System and Remote Sensing software. World Administrative Boundaries [ESRI]. http://www.creaf.uab.es/miramon.

Porro, J., L. Guo, K. Sharma, L. Benedetti, S. Van Hulle, P.A. Vanrolleghem, Y. Amerlinck, Z. Yuan, A. Shaw, and I. Nopens. 2011. Towards a benchmarking tool for minimizing wastewater utility greenhouse gas footprints. In *Watermatex 2011 Conference Proceedings, 8th IWA Symposium on Systems Analysis and Integrated Assessment, San Sebastian, Spain.*, 66:2483–95. January. http://www.ncbi.nlm.nih.gov/pubmed/23032782.

PRé Consultants. 2010. SimaPro 7.2.0. PRé Consultants, Amersfoort.

PRé Consultants. 2014. SimaPro 8.0.4. PRé Consultants, Amersfoort.

Pujadas, P., A. Blanco, S. Cavalaro, and A. Aguado. 2015. Performance-Based Procedure for the Definition of Controlled Low-Strength Mixtures. *Journal of Materials in Civil Engineering* 27(11): 6015003. http://ascelibrary.org/doi/abs/10.1061/(ASCE)MT.1943-5533.0001283. Accessed December 10, 2015.

REE. Red Eléctrica de España. 2012. The Spanish electric system (El sistema eléctrico español). Retrieved from http://www.ree.es/sites/default/files/downloadable/inf_sis_elec_ree_2011_v3.pdf.

Rees, W. and M. Wackernagel. 1996. Urban ecological footprints: Why cities cannot be sustainable—And why they are a key to sustainability. *Environmental Impact Assessment Review* 16(4): 223–248.

Righi, S., L. Oliviero, M. Pedrini, A. Buscaroli, and C. Della Casa. 2013. Life Cycle Assessment of management systems for sewage sludge and food waste: centralized and decentralized approaches. *Journal of Cleaner Production* 44: 8–17. http://www.sciencedirect.com/science/article/pii/S0959652612006488. Accessed April 14, 2017.

Risch, E., C. Boutin, P. Roux, S. Gillot, and A. Héduit. 2011. LCA in wastewater treatment – Applicability and limitations for constructed wetland systems: using vertical Reed Bed Filters. In *LCM 2011 International Conference on Life Cycle Management, August 28-31, 2011 Berlin, Germany, 12 Pages*.

Risch, E., O. Gutierrez, P. Roux, C. Boutin, and L. Corominas. 2015. Life cycle assessment of urban wastewater systems: Quantifying the relative contribution of sewer systems. *Water Research* 77: 35–48.

Risch, E., P. Loubet, M. Núñez, and P. Roux. 2014. How environmentally significant is water consumption during wastewater treatment? Application of recent developments in LCA to WWT technologies used at 3 contrasted geographical locations. *Water Research* 57: 20–30. http://www.ncbi.nlm.nih.gov/pubmed/24704901. Accessed July 30, 2014.

Roberts, D.W., D. Kubel, A. Carrie, D. Schoeder, and C. Sorenson. 2008. Cost and Benefits of Complete Water Treatment Plant Automation. *AwwaRF and IWA, London.*

Rodriguez-Garcia, G., M. Molinos-Senante, A. Hospido, F. Hernández-Sancho, M.T. Moreira, and G. Feijoo. 2011. Environmental and economic profile of six typologies of wastewater treatment plants. *Water Research* 45(18): 5997–6010.

Roux, P., I. Mur, E. Risch, and C. Boutin. 2011. Urban planning of sewer infrastructure : Impact of population density and land topography on environmental performances of wastewater treatment systems. In *Presented at the Life Cycle Conference 2011, "Policy: LCM in Public Policy" Session, Berlin*.

RuralCat. 2016. Agrometeorological data - Cabrils. https://www.ruralcat.net/web/guest/agrometeo.estacions?p_p_id=234&p_p_lifecycle=0&p_p_state=normal&p_p_mode=view&p_p_col_id=column-3&p_p_col_count=1&_234_struts_action=%2Fadasasistemas%2Fagrometeo%2Festacio.

Saiz, S., C. Kennedy, B. Bass, and K. Pressnail. 2006. Comparative life cycle assessment of standard and green roofs. *Environmental Science & Technology* 40(13): 4312–6. http://www.ncbi.nlm.nih.gov/pubmed/16856752.

San Francisco Stormwater Design Guidelines. 2010. Appendix A: BMP Fact Sheets. www.sfwater.org/modules/showdocument.aspx?documentid=2778 Accessed October 2014.

San Millán, J. 2008. *Estudi d'inundabilitat de la Riera de Cabrils, al seu pas pel sector C/Manuel Roca - El Barato. Modificació puntual del Pla General d'Ordenació Urbanística Municipal. Vilassar de Mar (Maresme)*.

Sanchez, S.T., J. Woods, M. Akhurst, M. Brander, M. O'Hare, T.P. Dawson, R. Edwards, A.J. Liska, and R. Malpas. 2012. Accounting for indirect land-use change in the life cycle assessment of biofuel supply chains. *Journal of the Royal Society, Interface / the Royal Society* 9(71): 1105–19. http://rsif.royalsocietypublishing.org/content/9/71/1105. Accessed February 29, 2016.

Sanjuan-Delmás, D., E. Hernando-Canovas, P. Pujadas, A. de la Fuente, X. Gabarrell, J. Rieradevall, and A. Josa. 2015a. Environmental and geometric optimisation of cylindrical drinking water storage tanks. *International Journal of Life Cycle Assessment* 20(12): 1612–1624.

Sanjuan-Delmás, D., A. Petit-Boix, C. Gasol, G. Villalba, M. Suárez-Ojeda, X. Gabarrell, A. Josa, and J. Rieradevall. 2014. Environmental assessment of different pipelines for drinking water transport and distribution network in small to medium cities: a case from Betanzos, Spain. *Journal of Cleaner Production* 66: 588–598. http://linkinghub.elsevier.com/retrieve/pii/S0959652613007488. Accessed April 29, 2014.

Sanjuan-Delmás, D., A. Petit-Boix, C.M. Gasol, R. Farreny, G. Villalba, M.E. Suárez-Ojeda, X. Gabarrell, A. Josa, and J. Rieradevall. 2015b. Environmental assessment of drinking water transport and distribution network use phase for small to medium-sized municipalities in Spain. *Journal of Cleaner Production* 87: 573–582. http://linkinghub.elsevier.com/retrieve/pii/S095965261400972X. Accessed January 8, 2015.

Sanyé-Mengual, E., H. Romanos, C. Molina, M.A. Oliver, N. Ruiz, M. Pérez, D. Carreras, et al. 2014. Environmental and self-sufficiency assessment of the energy metabolism of tourist hubs on Mediterranean Islands: The case of menorca (Spain). *Energy Policy* 65: 377–387.

Satterthwaite, D. 1997. Sustainable cities or cities that contribute to sustainable development? *Urban Studies* 34(10): 1667–1691. http://discovery.ucl.ac.uk/1333706/.

Satterthwaite, D. 2008. Cities' contribution to global warming: notes on the allocation of greenhouse gas emissions. *Environment and Urbanization* 20(2): 539–549. http://eau.sagepub.com/cgi/doi/10.1177/0956247808096127%5Cnpapers2://publication/doi/10.1177/0956247808096127.

Schmidt, J.H. and M. Thrane. 2009. *Life cycle assessment of aluminium production in new Alcoa smelter in Greenland*. Aalborg, Denmark.

Schweimer, G.W. and M. Levin. 2000. *Life Cycle Inventory for the Golf A4*.

Scoullos, M.J. 2003. Impact of anthropogenic activities in the coastal region of the Mediterranean Sea. In *International Conference on the Sustainable Development of the Mediterranean and Black Sea Environment, May, Thessaloniki, Greece*.

Semadeni-Davies, A., C. Hernebring, G. Svensson, and L.-G. Gustafsson. 2008. The impacts of climate change and urbanisation on drainage in Helsingborg, Sweden: Combined sewer system. *Journal of Hydrology* 350(1–2): 100–113. http://linkinghub.elsevier.com/retrieve/pii/S0022169407002910. Accessed July 11, 2014.

Sharma, A., G. Tjandraatmadja, A. Grant, T. Grant, and F. Pamminger. 2010. Sustainable sewerage servicing options for peri-urban areas with failing septic systems. *Water Science & Technology* 62(3): 570–585. http://eds.a.ebscohost.com/eds/detail/detail?sid=5d43e025-1300-44f9-af35-20b55aacba64%40sessionmgr4005&vid=0&hid=4110&bdata=JnNpdGU9ZWRzLWxpdmU%3D&preview=false#AN=53834193&db=a9h. Accessed April 5, 2016.

Sharma, K.R., Z. Yuan, D. de Haas, G. Hamilton, S. Corrie, and J. Keller. 2008. Dynamics and dynamic modelling of H2S production in sewer systems. *Water Research* 42(10–11): 2527–38. http://www.ncbi.nlm.nih.gov/pubmed/18336860. Accessed February 14, 2013.

Shehabi, A., J.R. Stokes, and A. Horvath. 2012. Energy and air emission implications of a decentralized wastewater system. *Environmental Research Letters* 7(2): 24007. http://iopscience.iop.org/article/10.1088/1748-9326/7/2/024007. Accessed April 5, 2016.

Shimizu, Y., S. Dejima, and K. Toyosada. 2012. The CO2 Emission Factor of Water in Japan. *Water* 4(4): 759–769. http://www.mdpi.com/2073-4441/4/4/759/. Accessed May 25, 2013.

Short, M.D., A. Daikeler, G.M. Peters, K. Mann, N.J. Ashbolt, R.M. Stuetz, and W.L. Peirson. 2014. Municipal gravity sewers: An unrecognised source of nitrous oxide. *The Science of the Total Environment* 468–469: 211–8. http://www.ncbi.nlm.nih.gov/pubmed/24029693. Accessed December 23, 2013.

Short, M.D., W.L. Peirson, G.M. Peters, and R.J. Cox. 2012. Managing Adaptation of Urban Water Systems in a Changing Climate. *Water Resources Management* 26(7): 1953–1981. http://link.springer.com/10.1007/s11269-012-0002-8. Accessed July 11, 2014.

Silva, J.L. da, M.K. El Debs, and A.T. Beck. 2008. Reliability evaluation of reinforced concrete pipes in crack opening limit state. *RIEM* 1(4): 314–330.

SMRC. The Stormwater Manager's Resource Center. 2014. Stormwater Management Fact Sheets. http://www.stormwatercenter.net/ Accessed October 2014.

Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor, and H.L. (eds) Miller. 2007. *Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, 2007*. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Sonnemann, G. and B. Vigon. 2011. *Global Guidance Principles for Life Cycle Assessment Databases. A basis for greener processes and products*. UNEP SETAC, Life Cycle Initative. http://www.unep.fr/scp/publications/details.asp?id=DTI/1410/PA. Accessed January 12, 2015.

Sousa, M.R.C. De, F.A. Montalto, and S. Spatari. 2012. Using Life Cycle Assessment to Evaluate Green and Grey Combined Sewer Overflow Control Strategies. *Journal of Industrial Ecology* 16(6): 901–913. http://doi.wiley.com/10.1111/j.1530-9290.2012.00534.x. Accessed July 30, 2014.

Søvik, A.K., J. Augustin, K. Heikkinen, J.T. Huttunen, J.M. Necki, S.M. Karjalainen, B. Kløve, et al. 2006. Emission of the greenhouse gases nitrous oxide and methane from constructed wetlands in europe. *Journal of Environmental Quality* 35(6): 2360–73. https://xpv.uab.cat/publications/jeq/abstracts/35/6/,DanaInfo=.admBvgnkukn2zovs8yv-S8A2,SSL+2360. Accessed May 15, 2016.

Spangler, M.G. 1941. The structural design of flexible pipe culverts. In *Bulletin 153. Ames: Iowa Engineering Experiment Station*.

Spanish Ministry of Development. 1999. *Maximum daily rainfall in the Peninsular Spain (Máximas lluvias diarias en la España Peninsular)*. Madrid, Spain.

Spatari, S., Z. Yu, and F. a Montalto. 2011. Life cycle implications of urban green infrastructure. *Environmental Pollution* 159(8–9): 2174–9. http://www.ncbi.nlm.nih.gov/pubmed/21330022. Accessed July 9, 2014.

SPSS Inc. 2009. PASW Statistics for Windows, Version 18.0. Chicago: SPSS Inc.

Strutt, J., S. Wilson, H. Shorney-Darby, A. Shaw, and A. Byers. 2008. Assessing the carbon footprint of water production. *American Water Works Association Journal* 100(6): 80–91.

Swamee, P.K. and A.K. Sharma. 2013. Optimal design of a sewer line using Linear Programming. *Applied Mathematical Modelling* 37(6): 4430–4439. http://linkinghub.elsevier.com/retrieve/pii/S0307904X12005380. Accessed August 15, 2014.

Swarr, T.E., D. Hunkeler, W. Klöpffer, H.-L. Pesonen, A. Ciroth, A.C. Brent, and R. Pagan. 2011. Environmental life-cycle costing: a code of practice. *The International Journal of Life Cycle Assessment* 16(5): 389–391. http://link.springer.com/10.1007/s11367-011-0287-5. Accessed April 6, 2017.

Témez, J. 1991. *Extended and Improved Rational Method. Version of the Highways Administration of Spain. Proc. XXIV Congress. Madrid, Spain. Vol A.*

Termes-Rifé, M., M. Molinos-Senante, F. Hernández-Sancho, and R. Sala-Garrido. 2013. Life cycle costing: a tool to manage the urban water cycle. *Journal of Water Supply: Research and Technology—AQUA*. IWA Publishing, September 7. file:///C:/Users/1218319/Downloads/2. Termes-Rifé etal WSJ 2013.pdf. Accessed April 9, 2015.

The World Bank. 2016. Urban population (% of total). http://data.worldbank.org/indicator/SP.URB.TOTL.IN.ZS Accessed July 2014.

Thomas, A., B.R.K. Mantha, and C.C. Menassa. 2016. A Framework to Evaluate the Life Cycle Costs and Environmental Impacts of Water Pipelines. In *Pipelines 2016*, 1152–1163. Reston, VA: American Society of Civil Engineers, July 11. http://ascelibrary.org/doi/10.1061/9780784479957.107. Accessed August 2, 2016.

Tillman, A.-M., M. Svingby, and H. Lundström. 1998. Life cycle assessment of municipal waste water systems. *The International Journal of Life Cycle Assessment* 3(3): 145–157. http://link.springer.com/10.1007/BF02978823. Accessed April 12, 2016.

Twort, A.C., D.D. Ratnayaka, and M.J. Brandt. 2000. *Water Supply*. Butterworth-Heinemann. http://books.google.com/books?id=wjI_7A8ZMeIC&pgis=1. Accessed September 25, 2014.

U.S. Army Corps of Engineers. 1998. *Engineering and Design. Conduits, culverts, and pipes. Engineer Manual 1110-2-2902*. Washington, DC 20314-1000.

U.S. Department of Commerce Economics and Statistics Administration Economist. Office of the Chief Economist. 2013. *Economic impact of Hurricane Sandy. Potential economic activity lost and gained in New Jersey and New York*. U.S. Department of Commerce.

UDFCD. Urban Drainage and Flood Control District. 2011. *Urban Storm Drainage Criteria Manual Volume 3. Chapter 2: BMP Selection*. Denver, US.

Udo de Haes, H.A., G. Finnveden, M. Goedkoop, E. Hertwich, P. Hofstetter, W. Klöpffer, W. Krewitt, and E. Lindeijer. 2002. Life Cycle Impact Assessment: Striving towards best practice. *Proceedings*.

UN-HABITAT. 2011. Hot Cities: Battle-ground for climate change. Global Report on Human Settlement 2011. http://mirror.unhabitat.org/downloads/docs/E_Hot_Cities.pdf. Accessed May 7, 2016.

UN. United Nations. 2013. *World Economic and Social Survey 2013. Chapter III Towards sustainable cities. http://www.un.org/en/development/desa/policy/wess/wess_current/wess2013/Chapter3.pdf*.

UN. United Nations. Department of Economic and Social Affairs. Population Division. 2012. *World Urbanization Prospects: The 2011 Revision, CD-ROM Edition.*

UNE 53331:1997. *Plastics. Unplastized poly(vinyl chloride) and high and medium density polyethylene (PE) pipes. Criterion for the assessment of pipes for plastics piping systems with car without pressure under external loads*. AENOR, Madrid, Spain.

United Nations. 1992. Agenda 21. In *United Nations Conference on Environment and Development (UNCED)*. http://www.un.org/esa/dsd/agenda21/.

United Nations. 2016. Sustainable Development GOALS - 17 Goals to transform our world. *Sustainable Development Goals - United Nations*. http://www.un.org/sustainabledevelopment/sustainable-development-goals/.

US EPA. 2016. *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2014*. Washington DC, USA. https://www.epa.gov/ghgemissions/inventory-us-greenhouse-gas-emissions-and-sinks-1990-2014. Accessed September 2, 2016.

USDT. United States Department of Transportation. 2014. Stormwater Best Management Practices in an Ultra-Urban Setting: Selection and Monitoring.

http://environment.fhwa.dot.gov/ecosystems/ultraurb/ Accessed December 2014.

USEPA. 1974. *Process Design Manual for Sulfide Control in Sanitary Sewerage Systems*.

USEPA. 2014. Greenhouse Gas Equivalencies Calculator. http://www.epa.gov/cleanenergy/energy-resources/calculator.html#results Accessed December 2014.

USEPA. United States Environmental Protection Agency. 1991. *Handbook: Sewer System Infrastructure Analysis and Rehabilitation*. Office of Research and Development. Cincinnati OH 45268.

USEPA. United States Environmental Protection Agency. 2013a. Best Management Practices (BMPs). http://www.epa.gov/nrmrl/wswrd/wq/stormwater/bmp.html Accessed July 2014.

USEPA. United States Environmental Protection Agency. 2013b. Overview of Greenhouse Gases. http://epa.gov/climatechange/ghgemissions/gases.html Accessed September 2013.

Vahidi, E., E. Jin, M. Das, M. Singh, and F. Zhao. 2015. Comparative Life Cycle Analysis of Materials in Wastewater Piping Systems. *Procedia Engineering* 118: 1177–1188. http://www.sciencedirect.com/science/article/pii/S1877705815021165. Accessed September 23, 2015.

Vahidi, E., E. Jin, M. Das, M. Singh, and F. Zhao. 2016. Environmental life cycle analysis of pipe materials for sewer systems. *Sustainable Cities and Society*.

Valderrama, C., R. Granados, J.L. Cortina, C.M. Gasol, M. Guillem, and A. Josa. 2012. Implementation of best available techniques in cement manufacturing: a life-cycle assessment study. *Journal of Cleaner Production* 25: 60–67. http://linkinghub.elsevier.com/retrieve/pii/S095965261100494X. Accessed March 8, 2013.

Valderrama C, Granados R, Cortina JL, Gasol CM, Josa A. 2013. Comparative LCA of sewage sludge valorisation as both fuel and raw material substitute in clinker production. *Journal of Cleaner Production* 51: 205–213.

Vargas-Parra, M.V. 2015. Optimizing rainwater harvesting systems in urban areas. Doctoral thesis. Universitat Autònoma de Barcelona.

Vargas-Parra, M.V., M.R. Rovira, X. Gabarrell, and G. Villalba. 2014. Cost effective rainwater harvesting system in the Metropolitan area of Barcelona. *Journal of Water Supply: Research and Technology – AQUA*.

Vargas-Parra, M.V., G. Villalba, and X. Gabarrell. 2013. Applying exergy analysis to rainwater harvesting systems to assess resource. *Resources, Conservation and Recycling* 72: 50–59.

Vázquez-Rowe, I., A. Marvuglia, S. Rege, and E. Benetto. 2014. Applying consequential LCA to support energy policy: land use change effects of bioenergy production. *The Science of the Total Environment* 472: 78–89. http://www.sciencedirect.com/science/article/pii/S0048969713012412. Accessed February 29, 2016.

Venkatesh, G. and H. Brattebø. 2011. Energy consumption, costs and environmental impacts for urban water cycle services: Case study of Oslo (Norway). *Energy* 36(2): 792–800. http://linkinghub.elsevier.com/retrieve/pii/S0360544210007309. Accessed January 29, 2013.

Venkatesh, G. and H. Brattebø. 2013. Typifying cities to streamline the selection of relevant environmental sustainability indicators for urban water supply and sewage handling systems: a recommendation. *Environment, Development and Sustainability* 15(3): 765–782. http://link.springer.com/10.1007/s10668-012-9405-1. Accessed August 2, 2016.

Venkatesh, G., J. Hammervold, and H. Brattebø. 2009. Combined MFA-LCA for Analysis of Wastewater Pipeline Networks. *Journal of Industrial Ecology* 13(4): 532–550. http://doi.wiley.com/10.1111/j.1530-9290.2009.00143.x. Accessed February 1, 2013.

Vida, N. and F. Arroyo. 2005. Medio Ambiente decide instalar la depuradora del Maresme en Santa Susanna. *El País*.

Viñolas, B. 2011. Applications and methodology advances in MIVES multicriteria valorations (Aplicaciones y avances de la metodología MIVES en valoraciones multicriterio). Doctoral Thesis. Universitat Politècnica de Catalunya.

Vollertsen, J., A.H. Nielsen, H.S. Jensen, T. Wium-Andersen, and T. Hvitved-Jacobsen. 2008. Corrosion of concrete sewers--the kinetics of hydrogen sulfide oxidation. *The Science of the Total Environment* 394(1): 162–70. http://www.ncbi.nlm.nih.gov/pubmed/18281080. Accessed December 23, 2013.

Wang, B., G. Parcsi, E.C. Sivret, S. Kenny, X. Wang, R.M. Stuetz, and S. Wales. 2012a. Volatile Organic Compound (VOC) Emissions From Sewer Networks in Sydney. In *Proceedings: Ozwater 2012*.

Wang, J., J. Zhang, H. Xie, P. Qi, Y. Ren, and Z. Hu. 2011. Methane emissions from a full-scale A/A/O wastewater treatment plant. *Bioresource Technology* 102(9): 5479–85. http://www.ncbi.nlm.nih.gov/pubmed/21084185. Accessed February 14, 2013.

Wang, R., M.J. Eckelman, and J.B. Zimmerman. 2013. Consequential environmental and economic life cycle assessment of green and gray stormwater infrastructures for combined sewer systems. *Environmental Science & Technology* 47(19): 11189–98. http://www.ncbi.nlm.nih.gov/pubmed/23957532.

Wang, R. and J.B. Zimmerman. 2015. Economic and environmental assessment of office building rainwater harvesting systems in various U.S. cities. *Environmental Science & Technology* 49(3): 1768–78. http://dx.doi.org/10.1021/es5046887. Accessed September 23, 2015.

Wang, X., G. Parcsi, E. Sivret, H. Le, B. Wang, and R.M. Stuetz. 2012b. Odour emission ability (OEA) and its application in assessing odour removal efficiency. *Water Science and Technology : A Journal of the International Association on Water Pollution Research* 66(9): 1828–33. http://www.iwaponline.com/wst/06609/wst066091828.htm. Accessed June 8, 2015.

Ward, M., R. Corsi, R. Morton, T. Knapp, D. Apgar, C. Quigley, C. Easter, J. Witherspoon, A. Pramanik, and W. Parker. 2011. Characterization of natural ventilation in wastewater collection systems. *Water Environmental Research* 83(3): 265–273.

WCED. World Commission On Environment And Development. 1987. *Our Common Future*. *Oxford Paperbacks*. Vol. f. http://www.un-documents.net/wced-ocf.htm.

Weidema, B., C. Bauer, R. Hischier, C. Mutel, T. Nemecek, J. Reinhard, C. Vadenbo, and G. Wernet. 2013. The ecoinvent database: Overview and methodology, Data quality guideline for the ecoinvent database version 3, www.ecoinvent.org.

Weidema, B., T. Ekvall, and R. Heijungs. 2009. *Guidelines for application of deepened and broadened LCA.*

Co-ordination Action for innovation in Life-Cycle Analysis for Sustainability (CALCAS), Deliverable D18 of work package 5.

WHO/UNICEF. 2015. 25 YEARS Progress on Sanitation and Drinking Water: 2015 Update and MDG Assessment. *New York, NY: UNICEF and World Health Organisation*: 4. http://www.who.int/water_sanitation_health/publications/jmp-2015-update/en/.

WHO/UNICEF Joint Monitoring Programme for Water Supply and Sanitation. 2012. *Progress on Drinking Water and Sanitation*. *Update*. http://www.wssinfo.org/fileadmin/user_upload/resources/JMP-report-2012-en.pdf.

Willis, J., R. Barnes, L. Fillmore, and A. Shah. 2010. Quantifying Methane Evolution from Sewers : Phase 1 Results of the. In *Proceedings of the Water Environment Federation, WEFTEC 2010, October 2-6, New Orleans, LA*, 5411–9.

Wolf, M.-A. and T. Ekvall. 2011. Life Cycle Inventory (LCI) Modelling and Attributional/Consequential Issues. *SETAC Globe-Environmental Quality through Science* 12(10). http://globe.setac.org/2011/october/modelling.html. Accessed March 21, 2016.

Wolman, A. 1965. The metabolism of cities. *Scientific American* 213(3): 179–190.

Wong, N.H., S.F. Tay, R. Wong, C.L. Ong, and A. Sia. 2003. Life cycle cost analysis of rooftop gardens in Singapore. *Building and Environment* 38(3): 499–509. http://linkinghub.elsevier.com/retrieve/pii/S0360132302001312.

WSC. 2011. Water and Wastewater facilities. WSC Annual report. http://www.wsc.com.mt/portals/0/Annual%20Report%202011/Annual_Report_2011_-_Water_&_Wastewater_Facilities_3.pdf.

WWAP (World Water Assessment Programme). 2012. *World Water Development Report Volume 4: Managing Water under Uncertainty and Risk. UNESCO*. http://www.unesco.org/new/fileadmin/MULTIMEDIA/HQ/SC/pdf/WWDR4 Volume 1-Managing Water under Uncertainty and Risk.pdf.

Yıldırım, M. and B. Topkaya. 2012. Assessing Environmental Impacts of Wastewater Treatment Alternatives for Small-Scale Communities. *CLEAN - Soil, Air, Water* 40(2): 171–178. http://doi.wiley.com/10.1002/clen.201000423. Accessed April 14, 2016.

Yongsiri, C., S.M. Asce, J. Vollertsen, and T. Hvitved-jacobsen. 2005. Influence of Wastewater Constituents on Hydrogen Sulfide Emission in Sewer Networks. *Journal of Environmental Engineering* 131(12): 1676–1683.

Zahmatkesh, Z., S.J. Burian, M. Karamouz, H. Tavakol-Davani, and E. Goharian. 2015. Low-Impact Development Practices to Mitigate Climate Change Effects on Urban Stormwater Runoff: Case Study of New York City. *Journal of Irrigation and Drainage Engineering* 141(1): 4014043. http://ascelibrary.org/doi/10.1061/(ASCE)IR.1943-4774.0000770. Accessed September 23, 2015.

Zamagni, A., J. Guinée, R. Heijungs, P. Masoni, and A. Raggi. 2012. Lights and shadows in consequential LCA. *The International Journal of Life Cycle Assessment* 17(7): 904–918. http://link.springer.com/10.1007/s11367-012-0423-x. Accessed October 5, 2015.

Zhang, L., P. De Schryver, B. De Gusseme, W. De Muynck, N. Boon, and W. Verstraete. 2008. Chemical and biological technologies for hydrogen sulfide emission control in sewer systems:

a review. *Water Research* 42(1–2): 1–12. http://www.ncbi.nlm.nih.gov/pubmed/17692889. Accessed July 11, 2014.

Zhang, X. and K. Caldeira. 2015. Time scales and ratios of climate forcing due to thermal versus carbon dioxide emissions from fossil fuels. *Geophysical Research Letters*: n/a-n/a. http://doi.wiley.com/10.1002/2015GL063514. Accessed June 9, 2015.

Zhang, X., W. Chen, C. Ma, and S. Zhan. 2012. Modeling the effect of humidity on the threshold friction velocity of coal particles. *Atmospheric Environment* 56: 154–160.

Zhang, X., W. Chen, C. Ma, and S. Zhan. 2013. Modeling particulate matter emissions during mineral loading process under weak wind simulation. *Science of the Total Environment* 449: 168–173.

Zhang, Y., S. Singh, and B.R. Bakshi. 2010a. Accounting for ecosystem services in life cycle assessment, Part I: a critical review. *Environmental Science & Technology* 44(7): 2232–2242.

Zhang, Y.I., B. Anil, and B.R. Bakshi. 2010b. Accounting for ecosystem services in life cycle assessment part II: Toward an ecologically based LCA. *Environmental Science and Technology* 44(7): 2624–2631.

Zhou, Q., P.S. Mikkelsen, K. Halsnæs, and K. Arnbjerg-Nielsen. 2012. Framework for economic pluvial flood risk assessment considering climate change effects and adaptation benefits. *Journal of Hydrology* 414: 539–549.

Appendix

Appendix 1. Supporting data related to Chapter 4

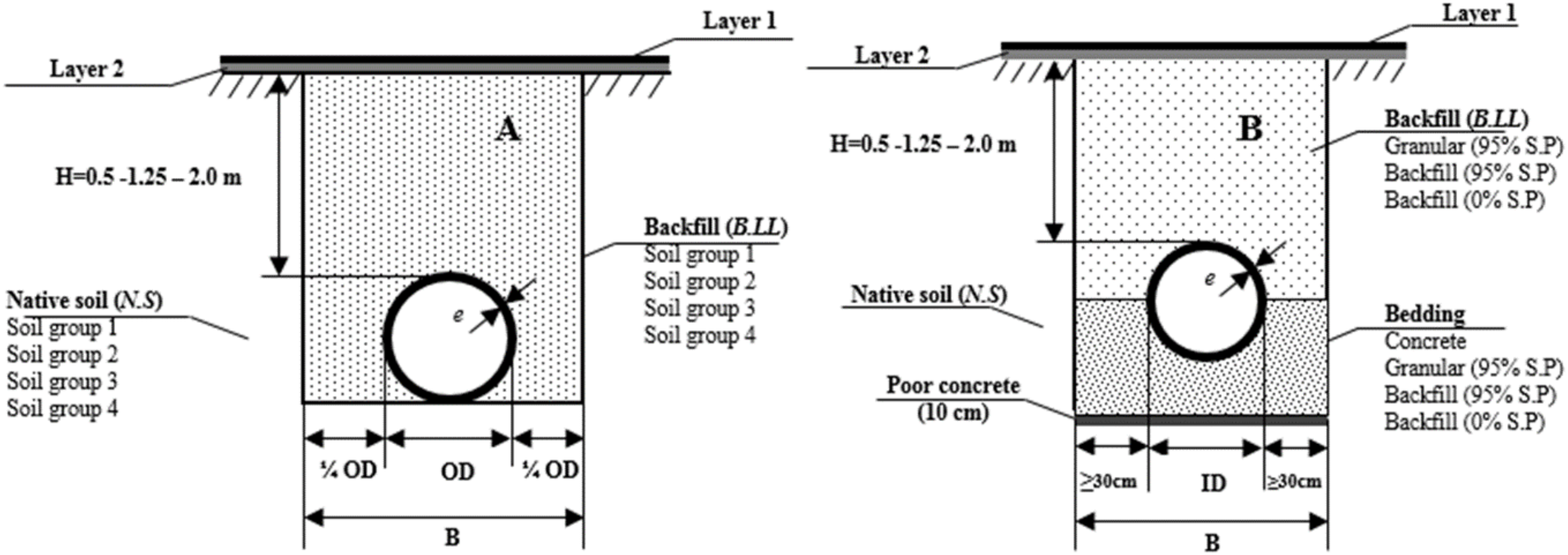
Appendix 1.1 Overall impacts of the constructive solutions (IC: Impact category)

IC	Units	Concrete pipe								Fibrocement pipe							
		CP1				CP2				CP1				CP2			
		Ø110	Ø300	Ø800	Ø1200	Ø110	Ø300	Ø800	Ø1200	Ø110	Ø300	Ø800	Ø1200	Ø110	Ø300	Ø800	Ø1200
AD	kg Sb eq	3.7E-01	4.5E-01	1.4E+00	2.3E+00	4.6E-01	5.6E-01	1.9E+00	3.0E+00	3.8E-01	4.9E-01	1.4E+00	2.1E+00	4.6E-01	6.0E-01	1.8E+00	2.8E+00
A	kg SO_2 eq	2.8E-01	3.4E-01	1.1E+00	1.7E+00	3.4E-01	4.2E-01	1.4E+00	2.3E+00	2.9E-01	3.8E-01	1.1E+00	1.7E+00	3.5E-01	4.7E-01	1.4E+00	2.2E+00
E	kg PO_4^{-3} eq	7.1E-02	8.5E-02	2.7E-01	4.4E-01	8.6E-02	1.1E-01	3.5E-01	5.8E-01	7.2E-02	9.5E-02	2.7E-01	4.1E-01	8.7E-02	1.2E-01	3.4E-01	5.3E-01
GW	kg CO_2 eq	9.4E+01	1.2E+02	3.9E+02	6.6E+02	1.3E+02	1.6E+02	5.8E+02	9.7E+02	9.3E+01	1.2E+02	3.7E+02	5.7E+02	1.3E+02	1.7E+02	5.4E+02	8.5E+02
OD	kg CFC-11 eq	6.9E-06	8.1E-06	2.5E-05	4.1E-05	8.3E-06	9.9E-06	3.2E-05	5.3E-05	6.9E-06	8.6E-06	2.4E-05	3.7E-05	8.2E-06	1.0E-05	3.0E-05	4.7E-05
HT	kg 1,4-DB eq	2.5E+01	3.0E+01	9.9E+01	1.6E+02	3.0E+01	3.7E+01	1.2E+02	2.0E+02	2.5E+01	3.1E+01	8.7E+01	1.3E+02	2.9E+01	3.7E+01	1.1E+02	1.7E+02
POC	kg C_2H_4 eq	9.8E-03	1.2E-02	3.8E-02	6.1E-02	1.2E-02	1.5E-02	4.9E-02	8.1E-02	1.0E-02	1.4E-02	3.9E-02	6.0E-02	1.2E-02	1.7E-02	5.0E-02	7.7E-02
CED	MJ	1.1E+03	1.3E+03	3.7E+03	6.0E+03	1.3E+03	1.5E+03	4.8E+03	7.8E+03	1.1E+03	1.6E+03	4.6E+03	6.9E+03	1.3E+03	1.9E+03	5.6E+03	8.5E+03

IC	Units	HDPE pipe								PVC pipe							
		PP1				PP2				PP1				PP2			
		Ø110	Ø300	Ø800	Ø1200	Ø110	Ø300	Ø800	Ø1200	Ø110	Ø300	Ø800	Ø1200	Ø110	Ø300	Ø800	Ø1200
AD	kg Sb eq	1.1E+00	1.7E+00	7.0E+00	1.4E+01	1.0E+00	1.8E+00	8.0E+00	1.6E+01	9.9E-01	1.1E+00	2.7E+00	4.2E+00	9.5E-01	1.2E+00	3.7E+00	5.9E+00
A	kg SO_2 eq	6.9E-01	8.8E-01	2.6E+00	4.8E+00	6.8E-01	9.5E-01	3.4E+00	6.2E+00	6.8E-01	7.4E-01	1.6E+00	2.5E+00	6.7E-01	8.1E-01	2.4E+00	3.8E+00
E	kg PO_4^{-3} eq	1.7E-01	2.0E-01	4.9E-01	8.3E-01	1.7E-01	2.1E-01	6.7E-01	1.1E+00	1.8E-01	1.9E-01	4.0E-01	6.0E-01	1.7E-01	2.0E-01	5.8E-01	9.2E-01
GW	kg CO_2 eq	1.4E+02	1.9E+02	6.0E+02	1.1E+03	2.5E+02	3.4E+02	1.2E+03	2.2E+03	1.4E+02	1.5E+02	3.5E+02	5.3E+02	2.5E+02	3.1E+02	9.6E+02	1.6E+03
OD	kg CFC-11 eq	1.8E-05	2.0E-05	4.3E-05	6.7E-05	1.6E-05	1.9E-05	5.6E-05	9.1E-05	1.8E-05	1.9E-05	3.9E-05	5.8E-05	1.6E-05	1.8E-05	5.2E-05	8.2E-05
HT	kg 1,4-DB eq	6.5E+01	7.3E+01	1.7E+02	2.9E+02	5.6E+01	7.1E+01	2.2E+02	3.7E+02	6.5E+01	7.0E+01	1.5E+02	2.2E+02	5.7E+01	6.8E+01	2.0E+02	3.1E+02
POC	kg C_2H_4 eq	2.6E-02	3.9E-02	1.5E-01	3.0E-01	2.6E-02	4.2E-02	1.8E-01	3.5E-01	2.4E-02	2.7E-02	6.0E-02	9.1E-02	2.4E-02	2.9E-02	8.8E-02	1.4E-01
CED	MJ	3.1E+03	4.7E+03	1.7E+04	3.5E+04	2.8E+03	4.6E+03	2.0E+04	3.9E+04	3.0E+03	3.3E+03	7.5E+03	1.1E+04	2.6E+03	3.2E+03	9.7E+03	1.5E+04

Appendix 2. Supporting data related to Chapter 5

Appendix 2.1 Parameters considered in the design of (A) plastic and (B) concrete pipes. OD: outer diameter; ID: internal diameter; ND: nominal diameter



Appendix 2.2 Soil types defined by UNE 53331:1997

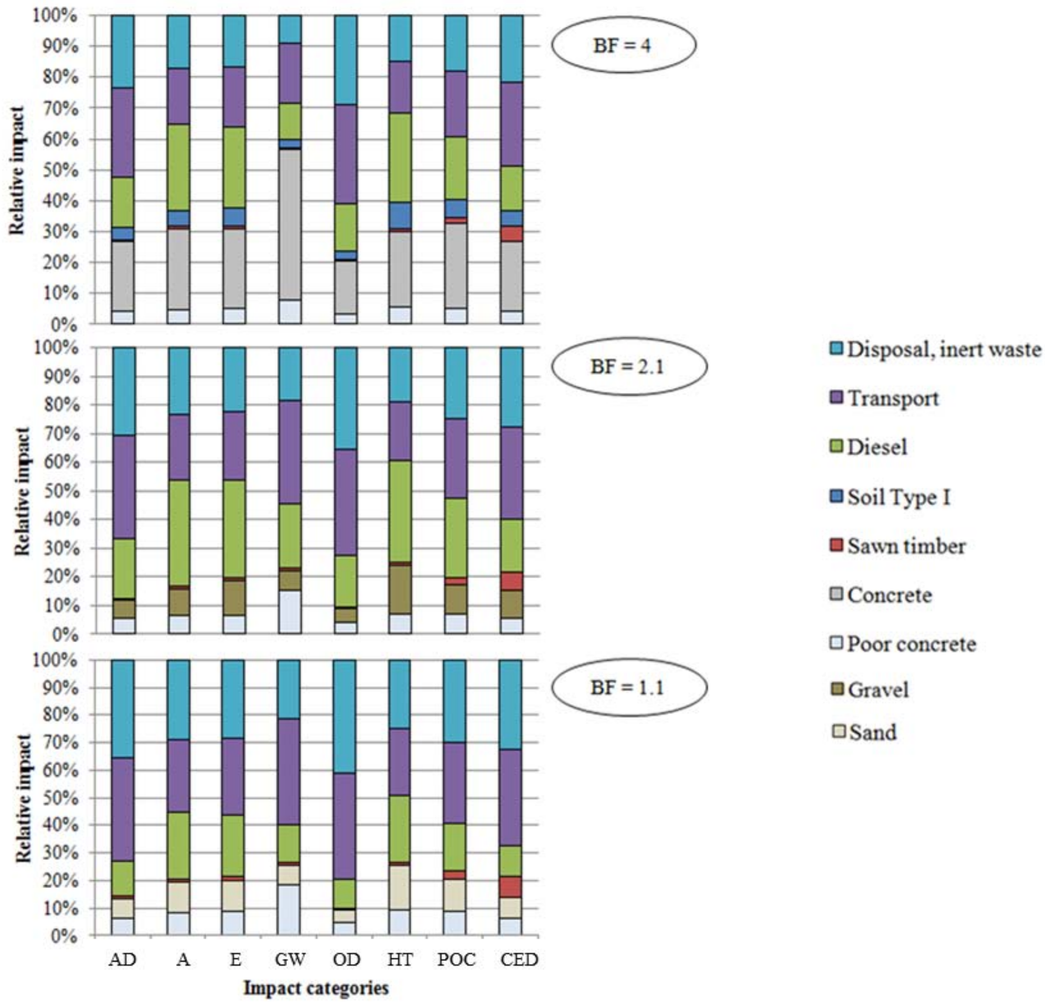
Soil group	Level of soil cohesion	Composition
Group 1	Non-cohesive	Gravel and sand, with less than 5% of particles having a diameter (φ) ≤ 0.06 mm
Group 2	Slightly cohesive	Gravel and a bit clayey or silty sands. Between 5% and 15% of the particles have φ ≤ 0.06 mm
Group 3	Moderately cohesive	Gravels and clayey or silty sands. Between 15% and 40% of the particles have φ ≤ 0.06 mm and it has silt with low plasticity
Group 4	Cohesive	Clay, silt and organic soils

Appendix 2.3 Inventory of the pipe subsystem considering different pavement conditions and traffic loads. Data for one linear metre of pipe and the lifespan of each material

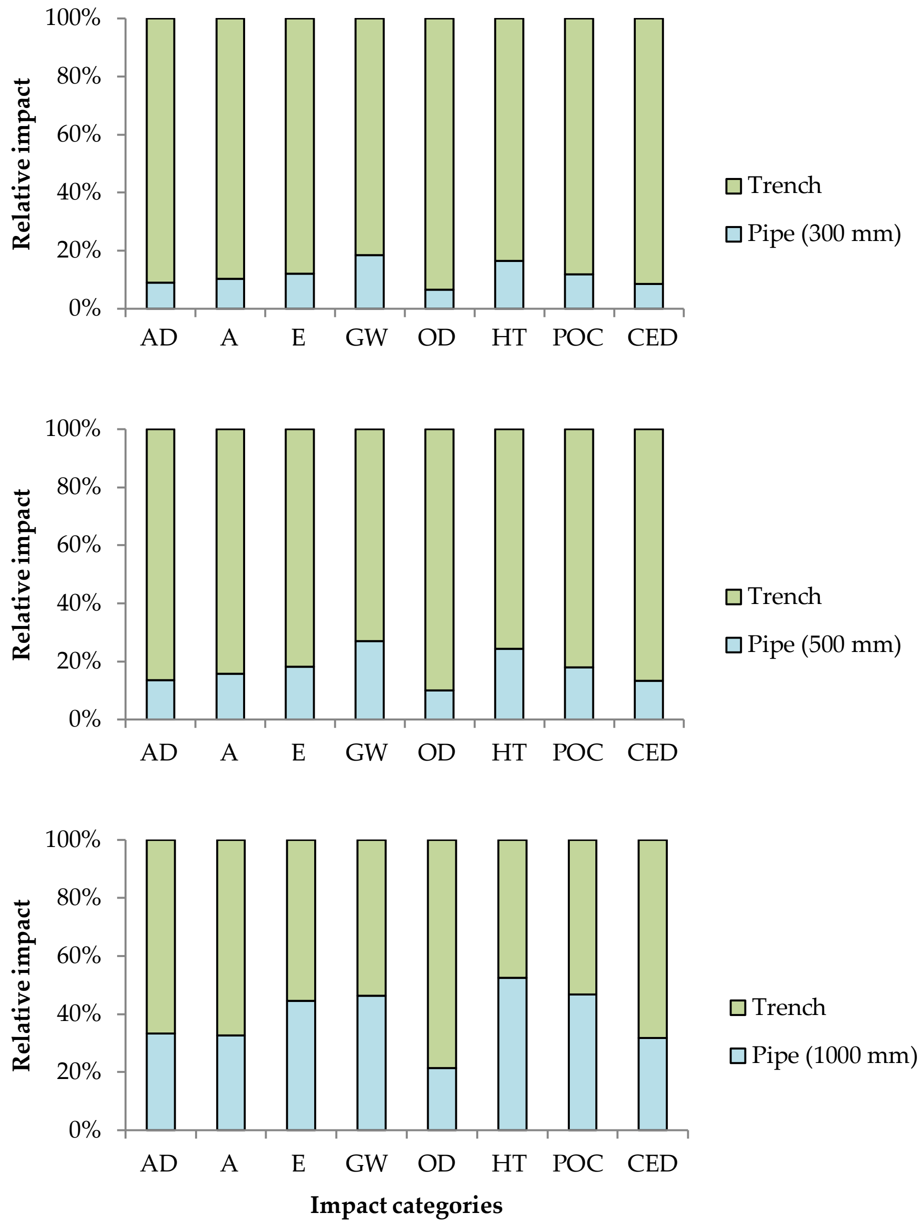
Pavement	DN (mm)	Traffic load (kN)	Concrete (BF=1.1) (50 years) (Case 1)					PVC (50 years)					HDPE (50 years)				
			Steel (kg)	Concrete (kg)	Rubber (kg)	Cement (kg)	Transport (tkm)	PVC + extrusion process (kg)		Polyurethane (kg)	Transport (tkm)		HDPE + extrusion process (kg)		Rubber (kg)	Transport (tkm)	
								Case 2	Case 3		Case 2	Case 3	Case 4	Case 5		Case 4	Case 5
4 cm asphalt + 15 cm concrete slabs	1000	0	11.7	911.4	0.3	0	55.4	104.4	104.4	0.2	20.9	20.9	89.5	72.1	0.6	18.0	14.5
		120	11.7	911.4	0.3	0	55.4	104.4	104.4	0.2	20.9	20.9	110.8	72.1	0.6	22.3	14.5
		260	11.7	911.4	0.3	0	55.4	104.4	104.4	0.2	20.9	20.9	137.0	72.1	0.6	27.5	14.5
		390	11.7	911.4	0.3	0	55.4	104.4	104.4	0.2	20.9	20.9	137.0	72.1	0.6	27.5	14.5
		600	16.2	911.4	0.3	0	55.7	129.5	104.4	0.2	25.9	20.9	161.0	72.1	0.6	32.3	14.5
	500	0	0	286.4	0	6.7	17.6	26.2	26.2	0.1	5.3	5.3	18.1	18.1	0.3	3.7	3.7
		120	0	286.4	0	6.7	17.6	26.2	26.2	0.1	5.3	5.3	27.7	18.1	0.3	5.6	3.7
		260	0	286.4	0	6.7	17.6	26.2	26.2	0.1	5.3	5.3	27.7	18.1	0.3	5.6	3.7
		390	0	286.4	0	6.7	17.6	26.2	26.2	0.1	5.3	5.3	34.3	18.1	0.3	6.9	3.7
		600	2.7	286.4	0	6.7	17.7	32.4	26.2	0.1	6.5	5.3	42.1	18.1	0.3	8.5	3.7
	300	0	0	131.9	0	4.4	8.2	9.7	9.7	0.03	1.9	1.9	6.7	6.7	0.2	1.4	1.4
		120	0	131.9	0	4.4	8.2	9.7	9.7	0.03	1.9	1.9	8.4	6.7	0.2	1.7	1.4
		260	0	131.9	0	4.4	8.2	9.7	9.7	0.03	1.9	1.9	10.4	6.7	0.2	2.1	1.4
		390	0	131.9	0	4.4	8.2	9.7	9.7	0.03	1.9	1.9	12.7	6.7	0.2	2.6	1.4
		600	1.7	131.9	0	4.4	8.3	12.1	9.7	0.03	2.4	1.9	12.7	6.7	0.2	2.6	1.4

Pavement	DN (mm)	Traffic load (kN)	Concrete (BF=1.1) (50 years) (Case 1)					PVC (50 years)					HDPE (50 years)				
			Steel (kg)	Concrete (kg)	Rubber (kg)	Cement (kg)	Transport (tkm)	PVC + extrusion process (kg)		Polyurethane (kg)	Transport (tkm)		HDPE + extrusion process (kg)		Rubber (kg)	Transport (tkm)	
								Case 2	Case 3		Case 2	Case 3	Case 4	Case 5		Case 4	Case 5
0 cm asphalt + 0 cm concrete slabs	1000	0	11.7	911.4	0.3	0	55.4	104.4	104.4	0.2	20.9	20.9	89.5	72.1	0.6	18.0	14.5
		120	16.2	911.4	0.3	0	55.7	129.5	104.4	0.2	25.9	20.9	161.0	72.1	0.6	32.3	14.5
		260	16.2	911.4	0.3	0	55.7	129.5	104.4	0.2	25.9	20.9	161.0	72.1	0.6	32.3	14.5
		390	16.2	911.4	0.3	0	55.7	129.5	104.4	0.2	25.9	20.9	161.0	72.1	0.6	32.3	14.5
		600	26.7	911.4	0.3	0	56.3	0	0	0	0	0	0	72.1	0	0	14.5
	500	0	0	286.4	0	6.7	17.6	26.2	26.2	0.1	5.3	5.3	18.1	18.1	0.3	3.7	3.7
		120	2.7	286.4	0	6.7	17.7	32.4	26.2	0.1	6.5	5.3	34.3	18.1	0.3	6.9	3.7
		260	2.7	286.4	0	6.7	17.7	40.1	26.2	0.1	8.0	5.3	42.1	18.1	0.3	8.5	3.7
		390	2.7	286.4	0	6.7	17.7	40.1	26.2	0.1	8.0	5.3	42.1	18.1	0.3	8.5	3.7
		600	4.4	286.4	0	6.7	17.9	40.1	26.2	0.1	8.0	5.3	51.4	18.1	0.3	10.3	3.7
	300	0	0	131.9	0	4.4	8.2	9.7	9.7	0.03	1.9	1.9	6.7	6.7	0.2	1.4	1.4
		120	1.7	131.9	0	4.4	8.3	12.1	9.7	0.03	2.4	1.9	12.7	6.7	0.2	2.6	1.4
		260	1.7	131.9	0	4.4	8.3	12.1	9.7	0.03	2.4	1.9	15.7	6.7	0.2	3.2	1.4
		390	1.7	131.9	0	4.4	8.3	12.1	9.7	0.03	2.4	1.9	15.7	6.7	0.2	3.2	1.4
		600	1.7	131.9	0	4.4	8.3	15.0	9.7	0.03	3.0	1.9	19.1	6.7	0.2	3.9	1.4

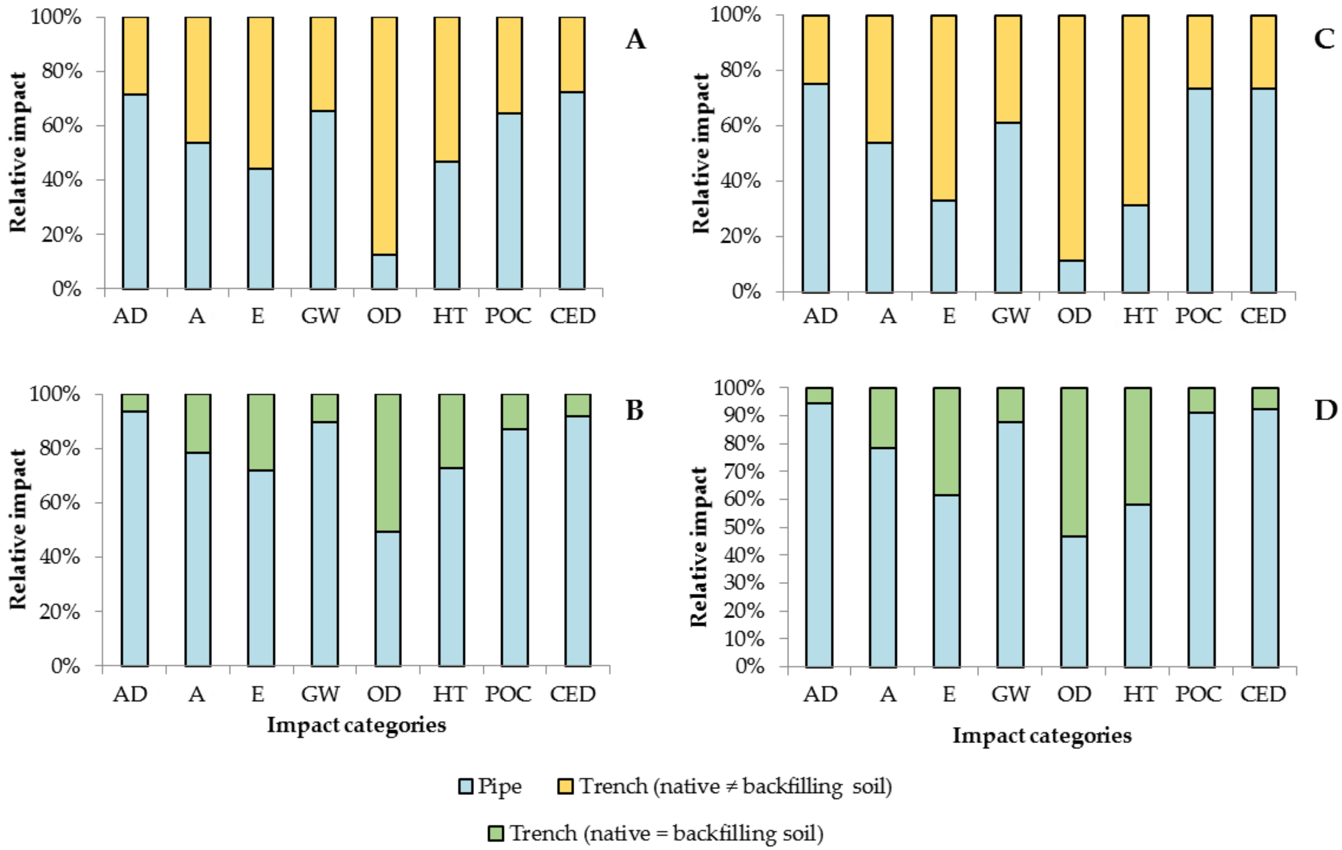
Appendix 2.4 Contribution of the material and energy flows of three different trench configurations to the environmental impacts of the trench subsystem of concrete pipes



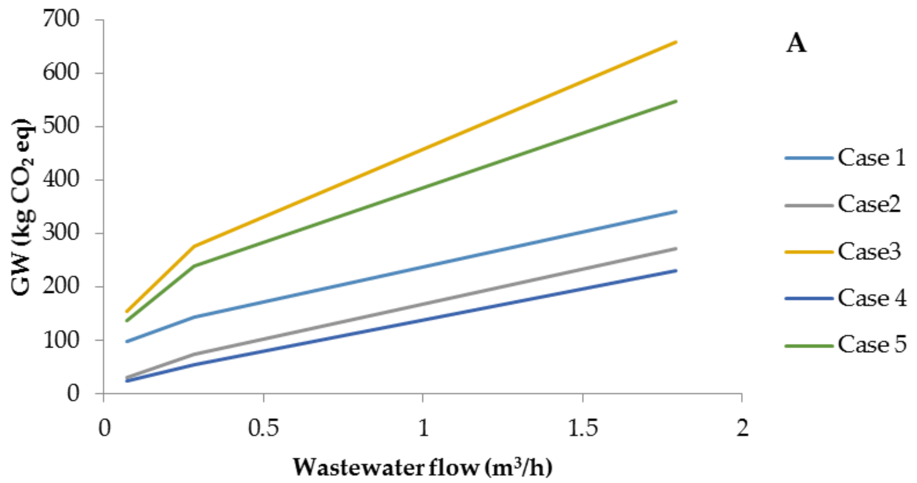
Appendix 2.5 Contribution of pipe and trench subsystems to the total impacts of concrete pipes solutions with a BF=1.1 and diameters of 300, 500 and 1000 mm

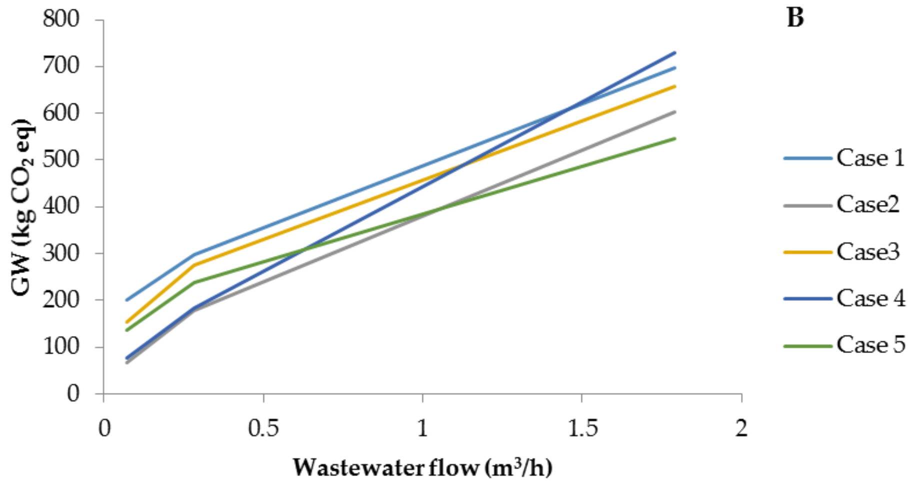


Appendix 2.6 Contribution of the pipe and trench subsystems to the total environmental impacts of the constructive solutions for PVC and HDPE pipes with a diameter of 1000 mm, null traffic load and a pavement layer. A) In PVC pipes, the native soil is replaced with newly extracted materials. B) In PVC pipes, the native soil is re-used as backfilling material. C) In HDPE pipes the native soil is replaced with newly extracted materials. D) In HDPE, the native soil is re-used as backfilling material

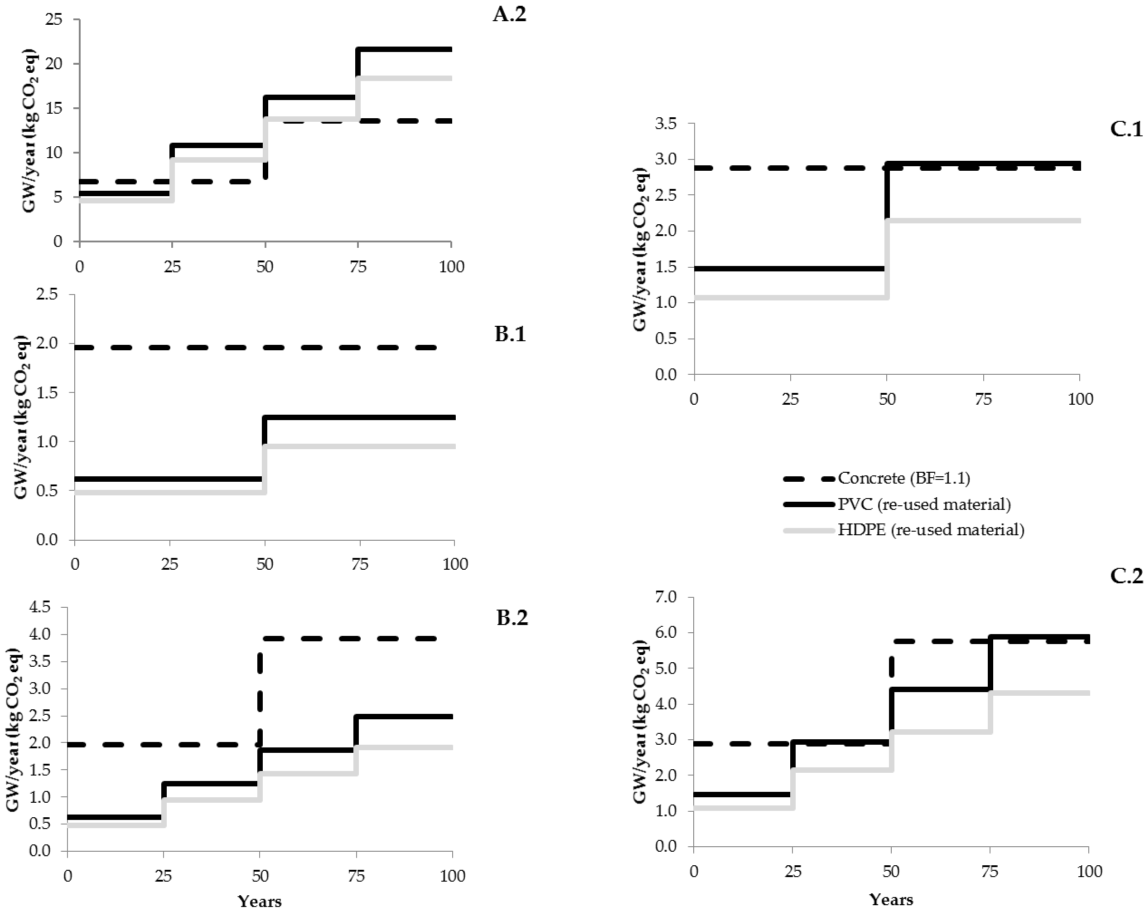


Appendix 2.7 Variation of the Global Warming Potential depending on the wastewater flow and the traffic loads, considering the accomplishment of the mechanical requirements of degraded or inexistent pavements. The three diameters analyzed in this study were considered (300, 500 and 1000 mm). A) No traffic load (0 kN). B) Traffic load = 260 kN





Appendix 2.8 Sensitivity analysis of the annual GWP of pipes with a diameter of: A) 1000 mm, B) 300 mm and C) 500 mm during a time span of 100 years. Different service lives are attributed to the constructive solutions: 1) 100 years to concrete and 50 years to plastics; 2) 50 years to concrete and 25 years to plastics.



Appendix 3. Supporting data related to Chapter 6

Appendix 3.1 Data for each city of the sample (year 2011)

City ID	Registered population	Maximum population	Tourist Equivalent Population (TEP)	Seasonal	Region	Density (inhabitants/km²)	Coastal	Electricity consumption (kWh)*	Length of the sewer system (km)*	Stormwater catchment area (km²)
1	24,512	24,512	25,008	No	Mediterranean	272	Yes	55,291	71	nd
2	100,085	100,085	102,256	No	Subtropical	977	Yes	363,559	305	102.4
3	632	1,264	729	Yes	Mediterranean	34	No	780	13.2	1.8
4	41,326	41,326	41,591	No	Mediterranean	2,545	No	96,422	121.7	5.4
5	28,138	28,138	28,228	No	Mediterranean	1,753	No	nd	60	2.6
6	20,728	20,728	20,765	No	Mediterranean	2,052	No	62,293	36	1.6
7	334,329	500,000	358,868	Yes	Mediterranean	1,661	Yes	670,694	625.2	48.2
8	6,007	11,500	7,267	Yes	Mediterranean	217	Yes	50,787	49.2	8
9	36,872	36,872	37,596	No	Subtropical	1,117	Yes	20,963	154.9	33
10	4,932	12,000	5,210	Yes	Mediterranean	34	No	nd	32	10.5
11	72,062	400,000	93,544	Yes	Mediterranean	1,871	Yes	23,764	203.6	10.4
12	13,537	nd	13,672	nd	Atlantic	560	No	120,326	90.3	24
13	3,086	4,230	3,108	Yes	Mediterranean	753	No	nd	27	nd
14	6,350	6,800	6,396	No	Mediterranean	73	No	22,299	30	87
15	27,000	40,227	38,655	Yes	Mediterranean	1,338	Yes	1,400,000	200	nd
16	33,008	150,000	44,146	Yes	Mediterranean	941	Yes	39,893	215	8.3
17	218,210	313,568	242,428	Yes	Mediterranean	391	Yes	1,952,983	999.3	143.5
18	7,019	7,019	7,065	No	Mediterranean	1,614	No	1830	56.9	2.6
19	9,065	9,097	9,194	No	Subtropical	495	Yes	nd	14.1	nd
20	230,354	230,354	242,030	No	Mediterranean	707	Yes	65,688	477.3	36.1
21	443,657	443,657	446,779	No	Mediterranean	501	No	4,279,034	1,481.6	275.4
22	6,800	20,000	8,414	Yes	Mediterranean	161	Yes	nd	92.5	6.2
23	5,315	5,315	5,374	No	Mediterranean	443	No	1,705	28	2
24	2,108	2,500	2,143	No	Mediterranean	31	No	nd	11	1
25	9,818	11,000	9,934	No	Mediterranean	13	No	nd	50	3.5
26	144,800	144,800	148,258	No	Subtropical	1,419	Yes	88,146	412.6	67
27	21,708	24,400	22,052	No	Mediterranean	656	No	nd	67.7	6
28	32,733	79,673	37,702	Yes	Mediterranean	553	Yes	155,093	81.9	10.5
29	3,115	3,115	3,161	No	Mediterranean	237	No	1,289	34.7	nd

City ID	Registered population	Maximum population	Tourist Equivalent Population (TEP)	Seasonal	Region	Density (inhabitants/km²)	Coastal	Electricity consumption (kWh)*	Length of the sewer system (km)*	Stormwater catchment area (km²)
30	16,738	30,000	17,070	Yes	Mediterranean	1,909	No	5,111	123.4	8.9
31	3,055	4,448	3,237	Yes	Mediterranean	203	No	1,061	28.8	nd
32	42,560	42,560	42,806	No	Mediterranean	121	No	nd	139.4	8
33	13,810	33,000	18,024	Yes	Mediterranean	388	Yes	nd	153.9	nd
34	19,310	20,000	19,417	No	Mediterranean	51	No	nd	90	nd
35	23,870	23,870	24,102	No	Mediterranean	142	No	38,111	78.1	4.7
36	1,400	2,500	1,476	Yes	Atlantic	13	No	nd	30	nd
37	14,134	14,123	14,432	No	Atlantic	165	No	116,545	nd	nd
38	13,345	13,345	13,462	No	Mediterranean	1,931	No	2,168	88.1	5.5
39	72,036	75,448	95,605	No	Mediterranean	197	Yes	432,697	449.1	nd
40	8,966	10,500	9,033	No	Mediterranean	546	No	4,531	60	2.6
41	60,000	65,000	60,599	No	Mediterranean	1,674	No	35114	300.9	45
42	14,688	20,000	15,908	Yes	Atlantic	433	Yes	962,215	105.4	14
43	4,474	5,500	4,576	No	Mediterranean	198	No	nd	25.3	2.5
44	61,576	61,576	63,051	No	Atlantic	521	Yes	891,303	494.6	52
45	7,224	10,200	7,746	Yes	Atlantic	222	Yes	129,061	106.2	14
46	52,200	52,200	52,491	No	Mediterranean	230	No	710,173	147	226.7
47	11,550	11,550	12,196	No	Atlantic	197	Yes	319,370	127.7	31
48	21,094	28,519	21,452	Yes	Mediterranean	367	No	nd	144.2	11.8
49	32,366	91,465	52,338	Yes	Mediterranean	431	Yes	335,963	166	22
50	22,554	25,000	22,766	No	Mediterranean	2,340	No	nd	3.1	4.7
51	16,924	17,500	17,394	No	Subtropical	390	Yes	235,772	31.7	27.7
52	26,615	26,615	29,190	No	Atlantic	387	Yes	587,359	117.2	28
53	15,428	15,428	15,540	No	Mediterranean	404	No	22,754	49.4	nd
54	33,372	170,000	53,995	Yes	Mediterranean	574	Yes	526,295	107.3	12.4
55	136,415	170,000	138,456	Yes	Atlantic	620	No	561,678	659.8	190
56	17,200	17,200	18,039	No	Subtropical	572	Yes	22,769	62.7	27
57	140,184	155,000	144,926	No	Mediterranean	2,248	Yes	33,176	204	277
58	68,181	300,000	74,613	Yes	Mediterranean	3,426	Yes	267,180	152	nd
59	32,140	32,140	32,635	No	Mediterranean	170	No	55,804	242.1	184
60	102,136	395,940	155,633	Yes	Mediterranean	1,430	Yes	54,287	317.9	22
61	6,392	7,000	6,543	No	Atlantic	50	No	nd	40	nd
62	313,437	320,000	315,630	No	Mediterranean	1,587	No	157,197	690.3	57.8
63	9,102	9,102	9,215	No	Mediterranean	643	No	48.4	68.6	nd
64	2,316	2,537	2,479	No	Mediterranean	5	No	nd	80	nd

City ID	Registered population	Maximum population	Tourist Equivalent Population (TEP)	Seasonal	Region	Density (inhabitants/km^2)	Coastal	Electricity consumption (kWh)*	Length of the sewer system (km)*	Stormwater catchment area (km^2)
65	22,532	nd	22,788	nd	Mediterranean	148	No	nd	91	4
66	6,129	20,000	9,334	Yes	Mediterranean	303	Yes	7,764	35.4	30
67	6,810	9,000	6,909	Yes	Mediterranean	42	No	nd	26.9	3.6
68	2,870	3,000	2,902	No	Mediterranean	62	No	nd	11	1.3

* Values equal to zero were excluded from the analysis, as it was not known whether they were real zeros or unrecorded parameters. nd: No Data

City ID	Annual precipitation (mm)	Stormwater runoff (m^3)	Registered drinking water (m^3)	Water flow (m^3)	Income per capita (€)**	Slope (city center – WWTP) (m)
1	196	nd	1.2E+06	nd	12,525	nd
2	324	3.0E+07	6.4E+06	3.6E+07	nd	nd
3	336	5.4E+05	1.0E+05	6.5E+05	nd	64
4	300	1.4E+06	2.1E+06	3.5E+06	10,882	-1
5	454	1.1E+06	1.2E+06	2.3E+06	nd	29
6	454	6.5E+05	9.9E+05	1.6E+06	nd	9
7	336	1.5E+07	2.2E+07	3.6E+07	nd	nd
8	442	3.2E+06	4.0E+05	3.6E+06	nd	17
9	324	9.6E+06	1.5E+06	1.1E+07	nd	nd
10	524	5.0E+06	2.2E+06	7.2E+06	23,706	nd
11	336	3.1E+06	9.7E+06	1.3E+07	nd	nd
12	1010	2.2E+07	8.5E+05	2.3E+07	15,408	23
13	336	nd	3.3E+05	nd	nd	14
14	300	2.3E+07	nd	nd	9,417	-19
15	504	nd	2.2E+06	nd	18,100	-19
16	504	3.7E+06	3.1E+06	6.8E+06	17,100	-29
17	340	4.4E+07	2.1E+07	6.5E+07	11,638	nd
18	357	8.3E+05	4.1E+05	1.2E+06	17,454	nd
19	214	nd	6.9E+05	nd	nd	nd
20	336	1.1E+07	1.3E+07	2.3E+07	nd	nd
21	300	7.4E+07	2.6E+07	1.0E+08	12,637	nd
22	336	1.9E+06	7.8E+05	2.7E+06	nd	nd
23	357	6.4E+05	4.3E+05	1.1E+06	21,047	nd
24	463	4.3E+05	1.2E+05	5.5E+05	nd	46
25	463	1.5E+06	6.0E+05	2.1E+06	11,588	-21
26	214	1.3E+07	8.9E+06	2.2E+07	nd	nd
27	454	2.4E+06	1.7E+06	4.2E+06	nd	nd
28	336	3.2E+06	2.1E+06	5.3E+06	nd	-67
29	357	nd	nd	nd	12,334	nd
30	454	3.6E+06	2.0E+06	5.6E+06	nd	26
31	336	nd	3.0E+05	nd	nd	12
32	536	3.9E+06	2.0E+06	5.9E+06	13,682	86
33	524	nd	2.0E+06	nd	16,842	100
34	534	nd	1.2E+06	nd	nd	-146
35	536	2.3E+06	nd	nd	13,314	86
36	523	nd	8.3E+04	nd	nd	nd
37	817	nd	nd	nd	15,926	11
38	357	1.8E+06	9.3E+05	2.7E+06	19,333	nd
39	336	nd	7.7E+06	nd	nd	nd
40	640	1.5E+06	5.5E+05	2.0E+06	14,300	nd
41	454	1.8E+07	2.4E+06	2.1E+07	nd	nd
42	1690	2.1E+07	1.0E+06	2.2E+07	15,930	7
43	336	7.7E+05	3.9E+05	1.2E+06	nd	171
44	1690	7.9E+07	7.9E+06	8.7E+07	17,233	13
45	1010	1.3E+07	nd	nd	13,717	4
46	396	8.1E+07	3.5E+06	8.4E+07	nd	460
47	1010	2.8E+07	4.4E+05	2.9E+07	13,962	27
48	454	4.8E+06	2.0E+06	6.8E+06	nd	nd
49	340	6.7E+06	2.4E+06	9.1E+06	13,576	-13
50	336	1.4E+06	nd	nd	nd	nd
51	214	5.3E+06	1.1E+06	6.4E+06	nd	36
52	1010	2.5E+07	2.6E+06	2.8E+07	11,552	21
53	357	nd	8.2E+05	nd	16,187	nd
54	336	3.7E+06	3.1E+06	6.8E+06	nd	-53
55	1890	3.2E+08	8.9E+06	3.3E+08	19,611	nd
56	214	5.2E+06	1.3E+06	6.5E+06	nd	nd
57	504	1.3E+08	8.9E+06	1.3E+08	18,500	32
58	524	nd	6.9E+06	nd	17,620	nd
59	340	5.6E+07	2.6E+06	5.9E+07	12,484	nd

City ID	Annual precipitation (mm)	Stormwater runoff (m^3)	Registered drinking water (m^3)	Water flow (m^3)	Income per capita (€)**	Slope (city center – WWTP) (m)
60	336	6.7E+06	7.6E+06	1.4E+07	nd	19
61	523	nd	3.2E+05	nd	10,969	nd
62	435	2.3E+07	2.3E+07	4.6E+07	nd	21
63	357	nd	6.1E+05	nd	14,780	nd
64	196	nd	2.1E+05	nd	9,642	nd
65	463	1.6E+06	1.5E+06	3.1E+06	12,487	nd
66	454	1.2E+07	6.9E+05	1.3E+07	nd	10
67	336	1.1E+06	5.0E+05	1.6E+06	nd	180
68	463	5.2E+05	2.0E+05	7.2E+05	nd	27

** Estimations for 2011 were not found. Therefore, the analysis includes data for the period 2006-2011, when available. nd: No Data

Appendix 3.2 Comparison of the total electricity consumption in kWh under different regional conditions

Variables		N	Total kWh				
			Mean	Median	Minimum	Maximum	Standard Deviation
Population	Small	11	2.0E+04 $_a$	1.8E+03	4.8E+01	1.3E+05	3.9E+04
	Medium	21	2.5E+05 $_a$	9.6E+04	2.2E+03	1.4E+06	3.6E+05
	Large	16	6.6E+05 $_a$	3.2E+05	2.4E+04	4.3E+06	1.1E+06
Population density	Low	13	1.5E+05 $_a$	5.5E+04	7.8E+02	7.1E+05	2.1E+05
	Medium	20	5.6E+05 $_a$	2.0E+05	4.8E+01	4.3E+06	1.0E+06
	High	15	1.9E+05 $_a$	5.4E+04	1.8E+03	1.4E+06	3.7E+05
Climate	Atlantic	8	4.6E+05 $_a$	4.4E+05	1.2E+05	9.6E+05	3.4E+05
	Mediterranean	35	3.3E+05 $_a$	5.1E+04	4.8E+01	4.3E+06	8.0E+05
	Subtropical	5	1.5E+05 $_a$	8.8E+04	2.1E+04	3.6E+05	1.5E+05
Coastal	No	22	2.9E+05 $_a$	2.9E+04	4.8E+01	4.3E+06	9.1E+05
	Yes	26	3.7E+05 $_a$	2.0E+05	7.8E+03	2.0E+06	4.8E+05
Seasonal	No	29	3.0E+05 $_a$	5.6E+04	4.8E+01	4.3E+06	8.0E+05
	Yes	18	4.0E+05 $_a$	1.4E+05	7.8E+02	2.0E+06	5.5E+05
Income per capita	Medium-High	13	3.4E+05 $_a$	1.2E+05	1.7E+03	1.4E+06	4.7E+05
	Medium-Low	14	5.6E+05 $_a$	7.6E+04	4.8E+01	4.3E+06	1.2E+06

$_a$ Values in the same subgroup of variables that share the same subscript are not significantly different at $p< 0.05$

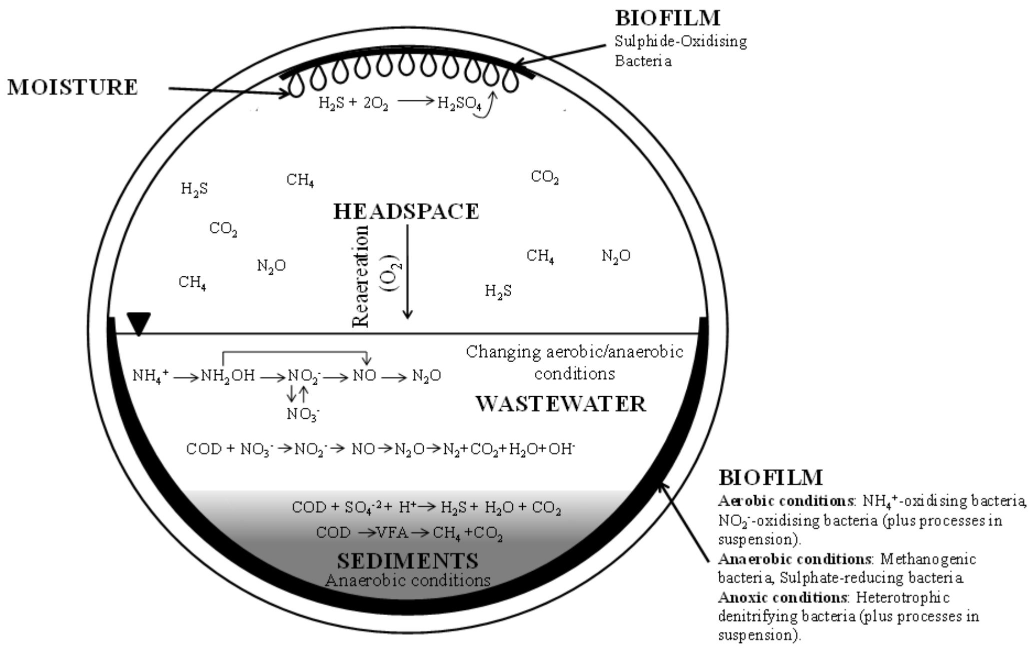
Appendix 3.3 Comparison of the electricity consumption per m^3 of water flow in kWh under different regional conditions

Variables		N	kWh/m^3 of water flow				
			Mean	**Median**	**Minimum**	**Maximum**	**Standard Deviation**
Population	Small	6	3.5E-03$_a$	1.6E-03	6.0E-04	1.4E-02	5.2E-03
	Medium	16	2.1E-02$_a$	1.6E-02	8.1E-04	7.7E-02	2.2E-02
	Large	14	1.0E-02$_a$	3.9E-03	2.5E-04	4.3E-02	1.3E-02
Population density	Low	5	7.1E-03$_a$	8.4E-03	9.5E-04	1.4E-02	5.9E-03
	Medium	18	2.0E-02$_a$	1.0E-02	6.0E-04	7.7E-02	2.1E-02
	High	13	8.0E-03$_a$	1.9E-03	2.5E-04	3.8E-02	1.2E-02
Climate	Atlantic	6	1.5E-02$_a$	1.1E-02	1.7E-03	4.3E-02	1.5E-02
	Mediterranean	25	1.4E-02$_a$	3.4E-03	2.5E-04	7.7E-02	1.9E-02
	Subtropical	5	1.1E-02$_a$	4.0E-03	1.9E-03	3.7E-02	1.5E-02
Coastal	No	15	9.2E-03$_a$	1.7E-03	8.1E-04	4.3E-02	1.4E-02
	Yes	21	1.7E-02$_a$	1.0E-02	2.5E-04	7.7E-02	1.9E-02
Seasonal	No	21	1.1E-02$_a$	3.5E-03	2.5E-04	4.3E-02	1.4E-02
	Yes	14	1.9E-02$_a$	9.9E-03	6.0E-04	7.7E-02	2.2E-02
Income per capita	Medium-High	9	7.8E-03$_a$	1.7E-03	2.5E-04	4.3E-02	1.4E-02
	Medium-Low	8	2.2E-02$_a$	2.4E-02	9.5E-04	4.3E-02	1.6E-02

$_a$Values in the same subgroup of variables that share the same subscript are not significantly different at $p < 0.05$

Appendix 4. Supporting data related to Chapter 7

Appendix 4.1 Biological reactions leading to the production of H_2S, CH_4 and N_2O in a generic sewer pipe. Figure adapted from (Hvitved-Jacobsen et al. 2002)



Under anaerobic conditions, the hydrolysis of complex organic molecules takes place, proceeding sequentially from complex to simple polymers such as proteins, carbohydrates and lipids. This readily biodegradable organic matter is transformed through fermentation into volatile fatty acids (VFA) that can be used by methanogenic bacteria (MB) (El-Fadel and Massoud 2001) and by sulfate-reducing bacteria (SRB) (Zhang et al. 2008) as the electron donor (Guisasola et al. 2008, 2009). Methane and H_2S generated under anaerobic conditions are then emitted where a pressure drop in the sewer line occurs; therefore, these gases will be released through manholes, pump stations and gas release valves (Guisasola et al. 2008) posing a risk to maintenance personnel (Othman and Mortezania 2010) and citizens.

The most common microbial processes involved in N_2O generation in sewers (Kampschreur et al. 2009; Desloover et al. 2012; Short et al. 2014) are nitrification and denitrification. Under aerobic conditions, high concentrations of bioavailable ammonia-nitrogen (>30mg/L) provide suitable conditions for nitrification (Nielsen et al. 1992). During nitrification, ammonium-oxidizing bacteria (AOB) can switch from classical ammonia oxidation with oxygen, to hydroxylamine (NH_2OH) oxidation or to nitrifier denitrification both producing N_2O. Under anaerobic conditions, the abundance of readily biodegradable organic carbon suggests significant potential for sewer N_2O formation via anoxic processes such as (incomplete) denitrification (Short et al. 2014). There are other possible chemical pathways leading to N_2O formation in sewers, such as the reaction between nitrite and hydroxylamine leading to NO and N_2O, and nitrite reduction with organic or inorganic compounds (Van Cleemput 1998).

Appendix 4.2 Summary of CH_4, N_2O and H_2S from the literature and calculation process

Summary of CH_4 emissions		
Authors	**µg CH_4/L**	**Comments**
Willis et al. (2010)	0	Minimum
	3,300,000	Maximum
Porro et al. (2011)	2,999	Average
	8,746	Maximum
Wang et al. (2011)	434	Minimum
	682	Maximum
Gutiérrez et al. (2014)	3,100	Downstream
	200	Wet well
	7,500	Lab scale
Jiang et al. (2010)	4,250	Minimum downstream
	5,500	Maximum downstream
	750	Minimum wet well
	1,000	Maximum wet well
	16,250	Minimum lab
	17,750	Maximum lab
Foley and Lant (2009)	1,000	Minimum wet well
	1,900	Maximum wet well
	2,800	Minimum rising main
	9,290	Maximum rising main
Guisasola et al. (2008)	5,000	Minimum rising main
	30,000	Maximum rising main
	7,500	Minimum lab
	30,000	Maximum lab
Guisasola et al. (2009)	3,500	Minimum rising main
	7,000	Maximum rising main
	1,000	Minimum lab
	7,500	Maximum lab

Calculation process description			
Willis et al. (2010)	**Pumping stations (unventilated wet wells)**		
Average emissions reported in the paper	Minimum:		(measured in the gas phase)
		0	mg CH_4/L
	Maximum:		
		3,300	mg CH_4/L
Assumptions and conversions		1000	µg/mg
Average emissions reported in the paper	Minimum:		
		0	**µg CH_4/L**
	Maximum:		
		3,300,000	**µg CH_4/L**
Porro et al. (2011)	**Headworks WWTP**		
Average emissions reported in the paper		12	mg COD/L (average)
		35	mg COD/L (maximum)
		235.3	kg COD/day
		58.8	kg CH_4/day
		21.1	tons/year

Assumptions and conversions		0.25	kg CH_4/kg COD
		7,000	m^3 wastewater/day
		21,000	3 subnetworks with the same flow
		1,000	µg/mg
		1,000,000	mg/kg
		1,000	L/m^3
		365	days/year
		1,000	kg/ton
Average emissions (estimated)		**2,999**	**µg CH_4/L (average)**
		8,746	**µg CH_4/L (maximum)**
		252.00	kg COD/day
		62.97	kg CH_4/day
		22.99	tons/year
Wang et al. (2011)	**Headworks WWTP**		
Average emissions reported in the paper	Minimum:	4.34	g CH_4/m^2/day
	Maximum:	6.82	g CH_4/m^2/day
	Minimum:	71	kg CH_4/year
	Maximum:	112	kg CH_4/year
Assumptions and conversions		45	Water surface area (m^2)
		450	Volume (m^3) (we assume 450 m^3/day)
		1,000,000	µg/g
		1000	L/m^3
		365	days/year
Average emissions (estimated)	Minimum:	**434**	**µg CH_4/L**
	Maximum:	**682**	**µg CH_4/L**
	Minimum:	71.3	kg CH_4/year
	Maximum:	112.0	kg CH_4/year
Gutiérrez et al. (2014)	**Lab vs field (Baseline results)**		
Average emissions reported in the paper	Downstream:	3.1	mg CH_4/L
	Wet well:	0.2	mg CH_4/L
	Lab scale:	7.5	mg CH_4/L h
Assumptions and conversions		1,000	µg/mg
Average emissions reported in the paper	Downstream:	**3,100**	**µg CH_4/L**
	Wet well:	**200**	**µg CH_4/L**
	Lab scale:	**7,500**	**µg CH_4/L** (considering the concentration of 1 hour)

Jiang et al. (2010)	**Lab vs field**		
Average emissions reported in the paper		17	mg COD/L (minimum downstream)
		22	mg COD/L (maximum downstream)
		3	mg COD/L (minimum wet well)
		4	mg COD/L (maximum wet well)
		65	mg COD/L (minimum lab)
		71	mg COD/L (maximum lab)
Assumptions and conversions		0.25	kg CH_4/kg COD
		1,000	μg/mg
Average emissions (estimated)		**4,250**	**μg CH_4/L** (minimum downstream)
		5,500	**μg CH_4/L** (maximum downstream)
		750	**μg CH_4/L** (minimum wet well)
		1,000	**μg CH_4/L** (maximum wet well)
		16,250	**μg CH_4/L** (minimum lab)
		17,750	**μg CH_4/L** (maximum lab)
Foley and Lant (2009)	**Field (wet well)**		
Average emissions reported in the paper	Minimum:		
		1	mg CH_4/L (wet well)
	Maximum:		
		1.9	mg CH_4/L (wet well)
	Minimum:		
		2.8	mg/L (rising main)
	Maximum:		
		9.29	mg/L(rising main)
Assumptions and conversions		1,000	μg/mg
Average emissions reported in the paper	Minimum:		
		1,000	**μg CH_4/L** (wet well)
	Maximum:		
		1,900	**μg CH_4/L** (wet well)
	Minimum:		
		2,800	**μg CH_4/L** (rising main)
	Maximum:		
		9,290	**μg CH_4/L** (rising main)
Guisasola et al. (2008)	**Rising main**		
Average emissions reported in the paper	Minimum		
		20	mg COD/L (rising)
	Maximum		
		120	mg COD/L (rising)
	Minimum		
		30	mg COD/L (lab)
	Maximum		
		120	mg COD/L (lab)
Assumptions and conversions		0.25	kg CH_4/kg COD
		23	ML/d
		1,000	L/m^3
		1,000	μg/mg
		1,000,000	mg/kg
		365	days/year
		1,000	mg/g
Average emissions (estimated)	Minimum		
		5,000	**μg CH_4/L** (rising main)
	Maximum		
		30,000	**μg CH_4/L** (rising main)

	Minimum		
		7,500	**μg CH_4/L** (lab)
	Maximum		
		30,000	**μg CH_4/L** (lab)
Guisasola et al. (2009)	**Rising main**		
Average emissions reported in the paper	Minimum		
		14	g COD/m^3 (rising main)
	Maximum		
		28	g COD/m^3 (rising main)
	Minimum		
		4	g COD/m^3 (lab)
	Maximum		
		30	g COD/m^3 (lab)
Assumptions and conversions		0.25	kg CH_4/kg COD
		23	ML/d
		1,000	L/m^3
		1,000	μg/mg
		1,000,000	mg/kg
		365	days/year
		1,000	mg/g
Average emissions (estimated)	Minimum		
		3,500	**μg CH_4/L** (rising main)
	Maximum		
		7,000	**μg CH_4/L** (rising main)
	Minimum		
		1,000	**μg CH_4/L** (lab)
	Maximum		
		7,500	**μg CH_4/L** (lab)

Summary of N_2O emissions		
Authors	**µg N_2O/L**	**Comments**
Clemens and Haas (1997)	298.27	Rising sewer
Short et al. (2014)	7.4	Minimum (WWTP)
	9.8	Maximum (WWTP)
Czepiel (1995)	1.6	WWTP primary treatment
	31.0	WWTP secondary treatment
Debruyn et al. (1994)	6.21	Minimum estimate
	12.65	Maximum estimate

Clemens and Haas (1997)	**Rising sewers**	
Average emissions reported in the paper	3.5	g N_2O /person/year
Assumptions and conversions	40	L/s (largest flow rate reported)
	31,536,000	s/year
	1,261,440,000	L/year (largest flow rate reported)
	1,261.44	ML/year
	3.456	ML/day
	0.0028	g/ML/person
	107,499	population Bayreuth 1995
	298.3	g/ML
Average emissions (estimated)	**298.27**	**µg N_2O/L**
Assumptions	Gas emissions will be bigger for the largest flow rate	
Short et al. (2014)	**Gravity sewers (headworks WWTP)**	
Average emissions reported in the paper	**7.4**	**µg N_2O/L** (WWTP A)
	9.8	**µg N_2O/L** (WWTP B)
	7.7	**µg N_2O/L** (WWTP C)
Assumptions	Assumptions: The emission factor is calculated as ($[N_2O]_{gas\_phase}$ x Q_{water})/Population served	
Czepiel (1995)	**WWTP**	
Average emissions reported in the paper	**1.6**	**µg N_2O /L** for primary treatment
	31.0	**µg N_2O /L** for AS secondary treatment
Debruyn et al. (1994)	**Headworks WWTP**	
Average emissions reported in the paper	23	µg N_2O/g SS
Assumptions and conversions	270	mg SS/L (Metcalf and Eddy. Inc 1991)
	550	mg SS/L (Metcalf and Eddy. Inc 1991)
	1000	mg/g
Average emissions (estimated)	**6.21**	**µg N_2O/L**
	12.65	**µg N_2O/L**

Summary of H_2S emissions		
Authors	**µg H_2S/L**	**Comments**
Gutiérrez et al. (2014)	6,000	Field (downstream)
	7,000	Lab scale
Chen and Szostak (2013)	1.5	Field (wet well)
Othman and Mortezania (2010)	15.6	Baseline
Jiang et al. (2009b)	2,800	Lab scale
Jiang et al. (2010)	7,000	Baseline maximum (field)
	5,000	Baseline minimum (field)
	<1,000	Wet well (field)
Dincer and Muezzinoglu (2007)	0.8	Maximum (field: pumping station)
	0.04	Minimum (field: pumping station)
Guisasola et al. (2008)	7,000	Minimum (field: rising main)
	12,000	Maximum (field: rising main)
	6,000	Minimum (lab)
	14,000	Maximum (lab)
Guisasola et al. (2009)	4,000	Minimum (field: rising main)
	7,000	Maximum (field: rising main)
	2,000	Minimum (lab)
	13,000	Maximum (lab)
Lahav et al. (2006)	25,000	Maximum (gravity sewer)
	20,000	Minimum (gravity sewer)
Mohanakrishnan et al. (2008)	25,000	Lab scale
	10,000	Lab scale
Nielsen et al. (2005)	200	Influent WWTP
Sharma et al. (2008)	5,000	Rising main
Yongsiri et al. (2005)	8,000	Maximum (gravity sewer)
	3,000	Minimum (gravity sewer)

Gutiérrez et al. (2014)	**Lab vs field (Baseline results)**	
Average emissions reported in the paper	Downstream:	
	6	mg S/L
	Lab scale:	
	7	mg S/L h
Assumptions and conversions	1,000	µg/mg
Average emissions (estimated)	Downstream:	
	6,000	**µg H_2S/L**
	Lab scale:	
	7,000	**µg H_2S/L (considering the concentration of 1 hour)**
Chen and Szostak (2013)	**Wet well (field)**	
Average emissions reported in the paper	Maximum:	(lower level of detection)
	1	ppm_v
Assumptions and conversions	34.1	g/mol H_2S
	10.3	ºC temp air
	$ppm_v = 0.08205(T/P \cdot M) \cdot (mg/m^3)$	
Average emissions (estimated)	**1.5**	**µg H_2S/L**

Othman and Mortezania (2010)	**Sewer**	
Average emissions reported in the paper	11	ppm_v
Assumptions and conversions	34.1	g/mol H_2S
	20	ºC temp air
	$ppm_v = 0.08205(T/P \cdot M) \cdot (mg/m^3)$	
Average emissions (estimated)	**15.6**	**µg H_2S/L**
Jiang et al. (2009b)	**Lab**	
Average emissions reported in the paper	2.8	mg S/L
Assumptions and conversions	1,000	µg/mg
Average emissions (estimated)	**2,800**	**µg H_2S/L**
Jiang et al. (2010)	**Field**	
Average emissions reported in the paper	Baseline:	
	Maximum:	
	7	mg S/L
	Minimum:	
	5	mg S/L
	Wet well:	
	<1	mg S/L
Assumptions and conversions	1,000	µg/mg
Average emissions (estimated)	Baseline:	
	Maximum:	
	7,000	**µg H_2S/L**
	Minimum:	
	5,000	**µg H_2S/L**
	Wet well:	
	<1,000	**µg H_2S/L**
Dincer and Muezzinoglu (2007)	**Field (pumping station)**	
Average emissions reported in the paper	Maximum:	
	764.2	µg H_2S/m^3
	Minumum:	
	39.6	µg H_2S/m^3
Assumptions and conversions	1,000	L/m^3
	1,000	mL/L
Average emissions (estimated)	Maximum:	
	0.7	**µg H_2S/L**
	Minumum:	
	0.04	**µg H_2S/L**
Guisasola et al. (2008)	**Lab vs field**	
Average emissions reported in the paper	Minimum rising main:	
	7	g S/m^3
	Maximum rising main:	
	12	g S/m^3
	Minimum lab:	
	6	g S/m^3
	Maximum lab:	
	14	g S/m^3
Assumptions and conversions	1,000,000	µg/g
	1,000	L/m^3

Average emissions (estimated)	Minimum rising main:	
	7,000	**μg H_2S/L**
	Maximum rising main:	
	12,000	**μg H_2S/L (considering the concentration of 1 hour)**
	Minimum lab:	
	6,000	**μg H_2S/L**
	Maximum lab:	
	14,000	**μg H_2S/L (considering the concentration of 1 hour)**
Guisasola et al. (2009)	**Lab vs field**	
Average emissions reported in the paper	Minimum rising main:	
	4	g S/m^3
	Maximum rising main:	
	7	g S/m^3
	Minimum lab:	
	2	g S/m^3
	Maximum lab:	
	13	g S/m^3
Assumptions and conversions	1,000,000	μg/g
	1,000	L/m^3
Average emissions (estimated)	Minimum rising main:	
	4,000	**μg H_2S/L**
	Maximum rising main:	
	7,000	**μg H_2S/L (considering the concentration of 1 hour)**
	Minimum lab:	
	2,000	**μg H_2S/L**
	Maximum lab:	
	13,000	**μg H_2S/L (considering the concentration of 1 hour)**
Lahav et al. (2006)	**Gravity sewer**	
Average emissions reported in the paper	Maximum:	
	25	mg S/L
	Minimum:	
	20	mg S/L
Assumptions and conversions	1,000	μg/mg
Average emissions (estimated)	Maximum:	
	25,000	**μg H_2S/L**
	Minimum:	
	20,000	**μg H_2S/L**
Mohanakrishnan et al. (2008)	**Lab**	
Average emissions reported in the paper	Maximum:	
	25	g S/m^3
	Minimum:	
	10	g S/m^3
Assumptions and conversions	1,000,000	μg/g
	1,000	L/m^3
Average emissions (estimated)	Maximum:	
	25,000	**μg H_2S/L**
	Minimum:	
	10,000	**μg H_2S/L**

Nielsen et al. (2005)	Influent to WWTP	
Average emissions reported in the paper	0.2	mg S/L
Assumptions and conversions	1,000	μg/mg
Average emissions (estimated)	**200**	**μg H_2S/L**
Sharma et al. (2008)	**Rising main**	
Average emissions reported in the paper	5	g S/m³
Assumptions and conversions	1,000,000 1,000	μg/g L/m³
Average emissions (estimated)	**5,000**	**μg H_2S/L**
Yongsiri et al. (2005)	**Gravity sewer**	
Average emissions reported in the paper	Maximum: 8 Minimum: 3	 g S/m³ g S/m³
Assumptions and conversions	1,000,000 1,000	μg/g L/m³
Average emissions (estimated)	Maximum: **8,000** Minimum: **3,000**	 **μg H_2S/L** **μg H_2S/L**

Appendix 4.3 Characteristics of Betanzos and Calafell needed for the GHG accounting process

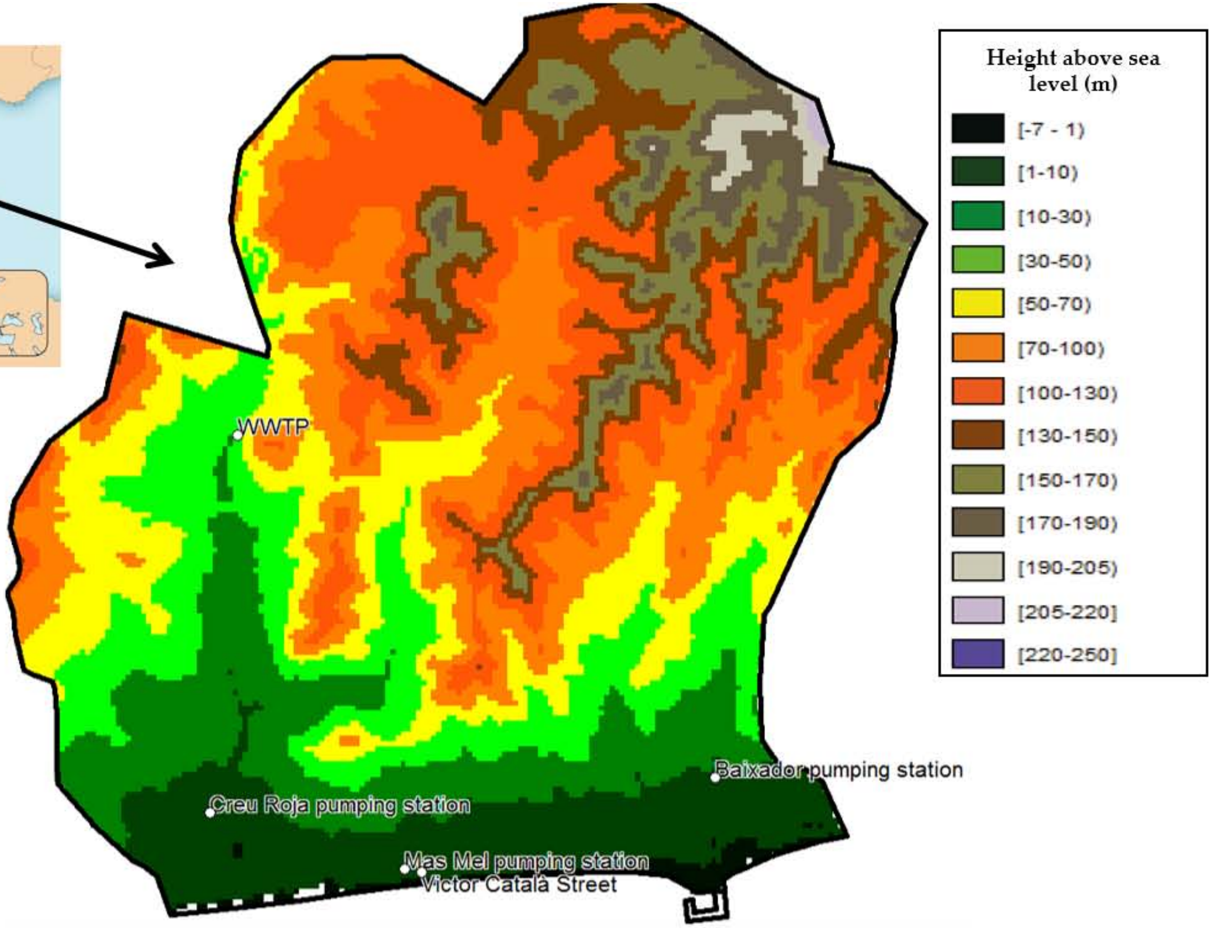
Parameters	Betanzos	Calafell
Climate	Atlantic	Mediterranean
Altitude (m)	33	67
Longitude (º)	8° 12' 42'' W	1° 33' 58'' E
Latitude (º)	43° 16' 50'' N	41° 11' 38'' N
Population*	13,537	24,984
Annual full time equivalents population**	13,672	38,211
City area (km²)	24.2	20.2
Density (residents/km²)*	554.6	1,205.0
Treated wastewater (m³/year)	1,145,699	3,349,749
Wastewater flow at the influent - summer campaign (m³/h)	155.95	375.0
Wastewater flow at the influent - winter campaign (m³/h)	100.4	238.8
Air flow at the influent - summer campaign (m³/h)	14.6	14.1
Air flow at the influent - winter campaign (m³/h)	14.1	13.8
Annual average rainfall (mm)***	>1,000	~500
Accumulated rainfall 1 day before the summer campaign (mm)	0	11.4
Accumulated rainfall 1 day before the winter campaign (mm)	17.0	9.1
Length of the sewer network (km)	77	173
Range of pipe diameters (mm)	63-1,200	160-1,200
Number of pumps	13	34
Number of manholes	1,578	4,154
Wastewater pumping requirements (kWh)	121,591	1,577,054

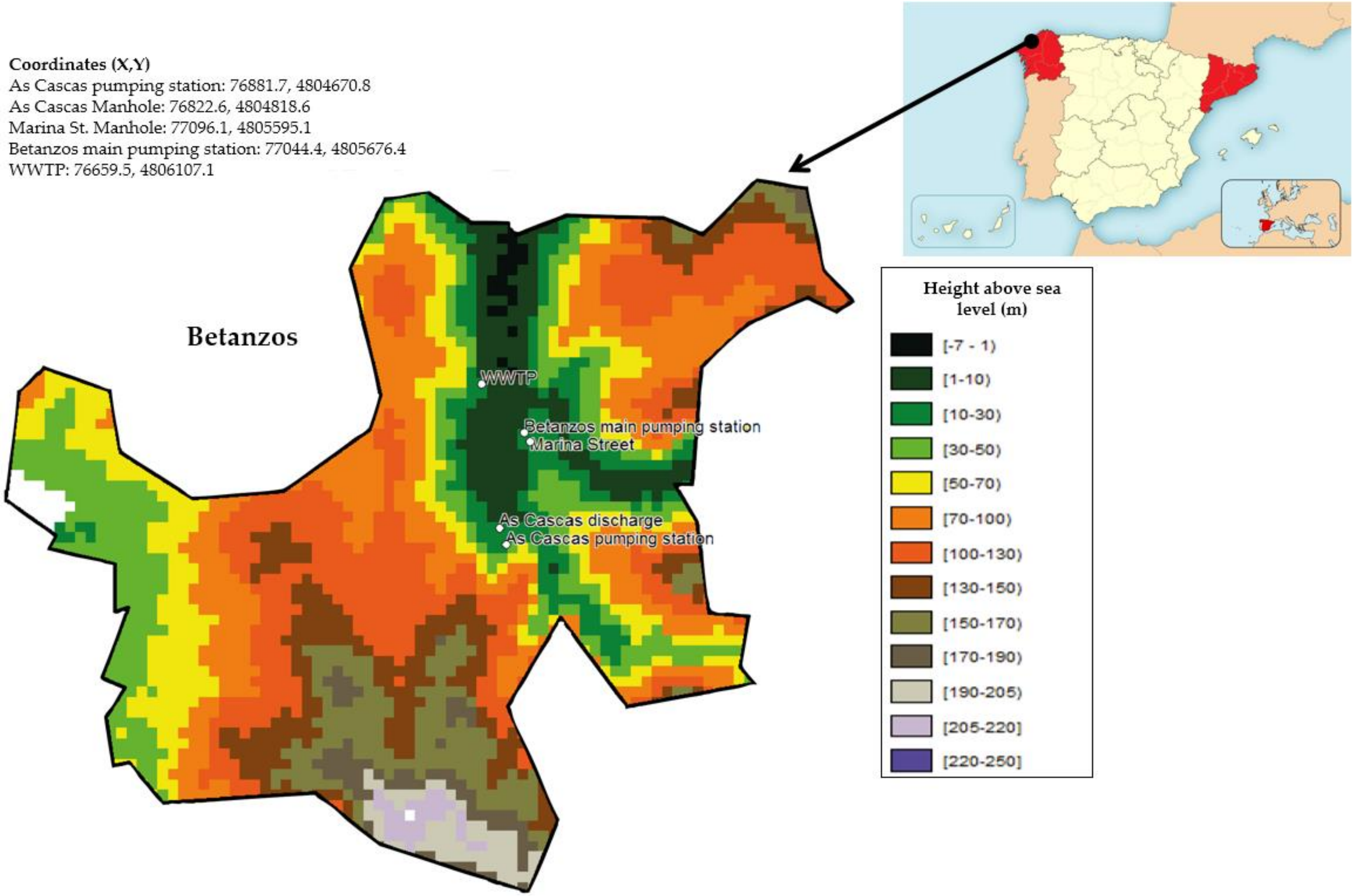
*(INE 2013)**Adapted from Augas de Galicia (2011). ***AEMET (2013). The rest of parameters were supplied by Viaqua and SOREA water utilities.

Appendix 4.4 Location of the sampling sites in Calafell and Betanzos. Maps created using MiraMon v7.1 (Pons 2000)

Coordinates (X,Y)
Baixador pumping station: 383326.4, 4561404.9.
Víctor Català St. Manhole: 381596.7, 4560847.1
Mas Mel pumping station: 381496.4, 4560865.9
Creu Roja pumping station: 380349.6, 4561198.0
WWTP: 380512.5, 4563422.9

Calafell




Coordinates (X,Y)
As Cascas pumping station: 76881.7, 4804670.8
As Cascas Manhole: 76822.6, 4804818.6
Marina St. Manhole: 77096.1, 4805595.1
Betanzos main pumping station: 77044.4, 4805676.4
WWTP: 76659.5, 4806107.1
Betanzos
WWTP
Betanzos main pumping station
Marina Street
As Cascas discharge
As Cascas pumping station
Height above sea level (m)
[-7 - 1)
[1-10)
[10-30)
[30-50)
[50-70)
[70-100)
[100-130)
[130-150)
[150-170)
[170-190)
[190-205)
[205-220]
[220-250]

Appendix 4.5 Average and integrated results of the summer and winter sampling campaigns in Calafell and Betanzos

		Calafell									
		Baixador PS		Victor Català St. MH		Mas Mel PS		Creu Roja PS		Influent to the WWTP	
		Summer	Winter	Summer	Winter	Summer	Winter	Summer	Winter	Summer	Winter
N_2O (µg/L)	Mean	0	n/m	5.9	0	8.1	0	18.3	6.8	0	0
	Maximum	0	n/m	11.6	0	16.9	0	22.4	9.0	0	0
	Minimum	0	n/m	0	0	0	0	0	0	0	0
CH_4 (µg/L)	Mean	1.8	n/m	125.9	10	226.9	32.3	316.7	89.4	150.4	31.8
	Maximum	2.1	n/m	263.8	39.3	618.3	184.8	405.2	127.6	249.4	56.0
	Minimum	1.6	n/m	1.3	0	14.6	2.3	1.5	39.8	44.9	18.9
H_2S (µg/L)	Mean	0	n/m	0.7	0	3.4	0	1.4	0	2.7	4.3

		Betanzos									
		As Cascas PS		As Cascas MH		Marina St. MH		Betanzos Main PS		Influent to the WWTP	
		Summer	Winter	Summer	Winter	Summer	Winter	Summer	Winter	Summer	Winter
N_2O (µg/L)	Mean	0	0	0	0	0	0	0	0	4.2	0
	Maximum	0	0	0	0	0	0	0	0	4.5	0
	Minimum	0	0	0	0	0	0	0	0	3.8	0
CH_4 (µg/L)	Mean	1109	161.1	27.8	87.8	20.2	8.7	42.4	15.1	192.6	80.9
	Maximum	2593.8	348.1	63	367.0	63.0	179.6	104.8	28.3	270.9	257.8
	Minimum	213.1	60.3	1.1	8.1	1.0	3.3	6.5	2.5	137.2	8.2
H_2S (µg/L)	Mean	0	0	0	0	0	0	0	0	0	0

Appendix 5. Supporting data related to Chapter 8

Appendix 5.1 Partial breakdown of the economic costs of each sewer component. Note that these are the total initial costs.

Diameter (mm)	Production	Installation	Machinery	Material	Labor	Total
Pipeline - Data per declared unit: one meter of sewer						
Concrete pipes						
250	5.1 €	173.9 €	16.3 €	73.8 €	88.9 €	179.0 €
300	6.6 €	179.9 €	16.8 €	77.7 €	92.0 €	186.5 €
350	8.0 €	185.6 €	17.2 €	81.2 €	95.2 €	193.6 €
400	9.3 €	235.4 €	25.5 €	118.0 €	101.2 €	244.7 €
500	13.5 €	260.1 €	28.9 €	135.2 €	109.6 €	273.6 €
600	18.9 €	292.6 €	33.3 €	158.7 €	119.4 €	311.5 €
700	24.8 €	321.2 €	37.1 €	180.9 €	128.0 €	346.0 €
800	28.9 €	374.8 €	46.7 €	219.2 €	137.9 €	403.7 €
1000	47.8 €	438.5 €	55.4 €	276.1 €	154.8 €	486.3 €
1200	72.3 €	533.9 €	72.6 €	357.7 €	175.9 €	606.2 €
Fibrocement pipes						
400	44.74 €	230.61 €	24.66 €	150.13 €	100.55 €	275.35 €
PVC pipes						
75	7.12 €	269.53 €	23.91 €	132.77 €	119.97 €	276.65 €
90	9.98 €	271.48 €	24.06 €	136.05 €	121.36 €	281.47 €
110	11.63 €	274.21 €	24.26 €	138.28 €	123.31 €	285.84 €
200	6.80 €	271.94 €	25.12 €	134.30 €	119.32 €	278.74 €
250	7.53 €	275.60 €	25.58 €	135.73 €	121.82 €	283.13 €
315	9.45 €	280.21 €	26.14 €	138.42 €	125.10 €	289.66 €
400	13.33 €	321.88 €	31.96 €	171.99 €	131.26 €	335.21 €
HDPE pipes						
110	12.90 €	274.88 €	24.26 €	139.28 €	124.24 €	287.78 €
200	35.69 €	303.80 €	25.12 €	164.40 €	149.97 €	339.49 €
250	52.93 €	323.80 €	25.58 €	183.11 €	168.05 €	376.74 €
300	23.59 €	340.93 €	26.01 €	152.39 €	186.11 €	364.52 €
350	29.38 €	377.82 €	30.20 €	180.21 €	196.79 €	407.21 €
400	37.80 €	399.15 €	31.96 €	196.46 €	208.53 €	436.95 €
450	48.66 €	411.76 €	33.75 €	215.17 €	211.50 €	460.43 €
500	61.94 €	424.44 €	35.57 €	236.33 €	214.48 €	486.38 €
700	123.43 €	475.75 €	43.19 €	329.58 €	226.41 €	599.18 €
800	158.56 €	546.57 €	51.64 €	403.85 €	249.64 €	705.13 €
1200	358.77 €	686.85 €	74.59 €	695.16 €	275.87 €	1,045.62 €
Appurtenances - Data per declared unit: one unit of sewer appurtenance						
Manhole	383.11 €	183.91 €	7.10 €	428.28 €	131.64 €	567.02 €
Inspection chamber 30x30	11.37 €	53.71 €	6.57 €	27.97 €	30.54 €	65.08 €
Inspection chamber 50x50	27.46 €	69.84 €	12.29 €	49.57 €	35.44 €	97.30 €
Inspection chamber 100x100	92.51 €	234.65 €	24.75 €	241.33 €	61.08 €	327.16 €
Submersible pump (60 m^3/h)	2,680.61 €	124.60 €	0.00 €	2,680.61 €	124.60 €	2,805.21 €
Scupper	44.76 €	86.43 €	7.39 €	62.08 €	61.72 €	131.19 €
Wastewater connection	12.47 €	46.88 €	7.84 €	22.43 €	29.08 €	59.35 €

Appendix 5.2 Midpoint, endpoint and single score results per m^3 of wastewater. Relative contribution of the production and installation of pipes and appurtenances to the impacts and costs

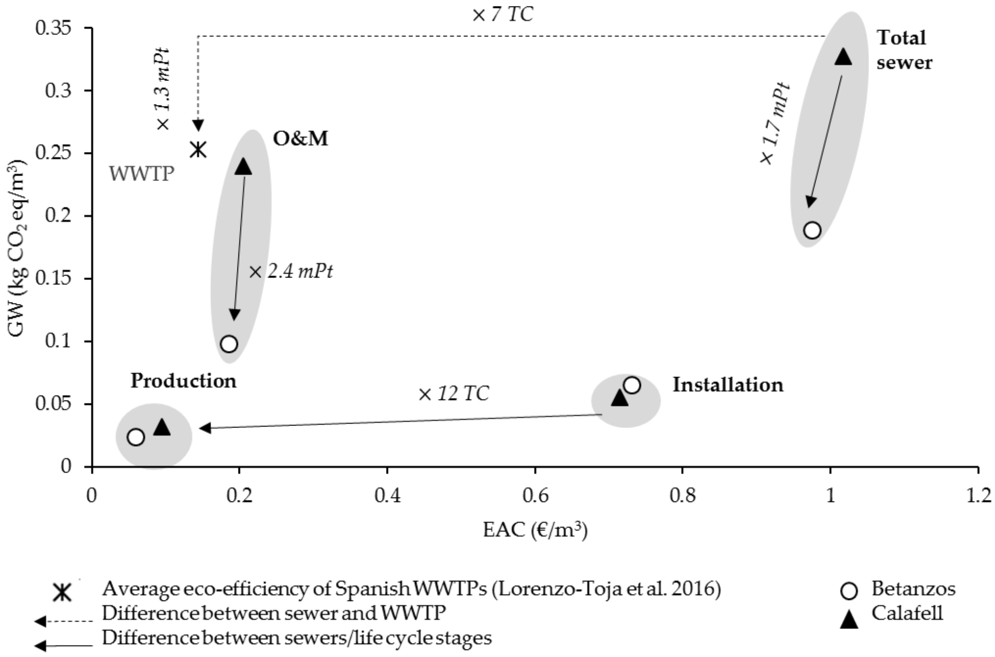
BETANZOS							
Indicator			Units	Concrete pipes		Plastic pipes	
				Production	Installation	Production	Installation
Midpoints	Global warming	GW	kg CO_2 eq	3.3E-03	2.8E-02	3.1E-03	3.1E-02
	Ozone depletion	OD	kg CFC-11 eq	1.3E-10	2.0E-09	4.5E-11	4.3E-09
	Terrestrial acidification	TA	kg SO_2 eq	7.4E-06	8.2E-05	9.0E-06	1.6E-04
	Freshwater eutrophication	FE	kg P eq	3.6E-07	2.0E-06	1.8E-07	3.5E-06
	Marine eutrophication	ME	kg N eq	3.9E-07	4.4E-06	4.8E-07	9.1E-06
	Human toxicity	HT	kg 1,4-DB eq	4.9E-04	2.6E-03	3.4E-04	4.1E-03
	Photochemical oxidant formation	POF	kg NMVOC	8.2E-06	1.2E-04	1.4E-05	2.6E-04
	Particulate matter formation	PMF	kg PM10 eq	3.3E-06	3.7E-05	3.2E-06	7.7E-05
	Terrestrial ecotoxicity	TET	kg 1,4-DB eq	8.4E-08	1.0E-06	1.1E-07	2.3E-06
	Freshwater ecotoxicity	FET	kg 1,4-DB eq	1.1E-05	5.9E-05	8.9E-06	1.1E-04
	Marine ecotoxicity	MET	kg 1,4-DB eq	1.1E-05	6.4E-05	8.3E-06	1.2E-04
	Ionising radiation	IR	kBq U^{235} eq	4.8E-04	3.4E-03	2.1E-04	6.0E-03
	Agricultural land occupation	ALO	m^{2}a	1.3E-03	9.3E-03	1.3E-04	2.9E-02
	Urban land occupation	ULO	m^{2}a	4.6E-05	4.3E-04	5.1E-06	1.2E-03
	Natural land transformation	NLT	m^2	5.9E-07	9.5E-06	1.2E-07	2.7E-05
	Water depletion	WD	m^3	4.8E-05	7.6E-04	1.6E-05	2.3E-03
	Metal depletion	MD	kg Fe eq	2.8E-04	8.8E-04	3.4E-05	2.0E-03
	Fossil depletion	FD	kg oil eq	4.0E-04	5.1E-03	1.5E-03	1.0E-02
& CED	Cumulative energy demand	CED	MJ	3.1E-02	3.0E-01	9.0E-02	6.5E-01
Endpoints	Human health		DALY	5.9E-09	5.1E-08	5.3E-09	6.6E-08
	Ecosystems		species.yr	4.4E-11	3.6E-10	2.6E-11	6.7E-10
	Resources		$	8.6E-05	9.0E-04	2.6E-04	1.9E-03
Single score			Pt	2.5E-04	2.4E-03	4.4E-04	4.0E-03
Economic cost	Equivalent annual cost	EAC	€	1.0E-02	1.8E-01	1.7E-02	5.2E-01

	Indicator		Units	Appurtenance		Pumping energy	Cleaning	Reposition	Others
				Production	Installation				
BETANZOS									
Midpoints	Global warming	GW	kg CO_2 eq	1.8E-02	7.3E-03	3.9E-02	0.0E+00	6.0E-02	0.0E+00
	Ozone depletion	OD	kg CFC-11 eq	9.6E-10	5.4E-10	3.0E-09	0.0E+00	5.9E-09	0.0E+00
	Terrestrial acidification	TA	kg SO_2 eq	6.4E-05	2.4E-05	2.4E-04	0.0E+00	2.6E-04	0.0E+00
	Freshwater eutrophication	FE	kg P eq	1.1E-05	4.3E-07	1.2E-05	0.0E+00	1.6E-05	0.0E+00
	Marine eutrophication	ME	kg N eq	4.1E-06	1.3E-06	6.8E-06	0.0E+00	1.5E-05	0.0E+00
	Human toxicity	HT	kg 1,4-DB eq	2.1E-02	6.0E-04	1.0E-02	0.0E+00	2.7E-02	0.0E+00
	Photochemical oxidant formation	POF	kg NMVOC	5.2E-05	3.6E-05	1.3E-04	0.0E+00	3.6E-04	0.0E+00
	Particulate matter formation	PMF	kg PM10 eq	3.8E-05	1.1E-05	7.4E-05	0.0E+00	1.3E-04	0.0E+00
	Terrestrial ecotoxicity	TET	kg 1,4-DB eq	2.4E-06	2.6E-07	1.0E-06	0.0E+00	5.2E-06	0.0E+00
	Freshwater ecotoxicity	FET	kg 1,4-DB eq	4.3E-04	1.3E-05	2.3E-04	0.0E+00	5.8E-04	0.0E+00
	Marine ecotoxicity	MET	kg 1,4-DB eq	4.4E-04	1.4E-05	2.4E-04	0.0E+00	6.0E-04	0.0E+00
	Ionising radiation	IR	kBq U^{235} eq	3.9E-03	6.7E-04	2.9E-02	0.0E+00	1.1E-02	0.0E+00
	Agricultural land occupation	ALO	m^2a	6.4E-04	1.5E-04	8.5E-04	0.0E+00	3.0E-02	0.0E+00
	Urban land occupation	ULO	m^2a	2.3E-04	5.1E-05	1.7E-04	0.0E+00	1.5E-03	0.0E+00
	Natural land transformation	NLT	m^2	3.4E-06	1.8E-06	5.6E-06	0.0E+00	3.3E-05	0.0E+00
	Water depletion	WD	m^3	2.2E-04	8.5E-05	3.1E-04	0.0E+00	2.6E-03	0.0E+00
	Metal depletion	MD	kg Fe eq	9.2E-03	1.8E-04	8.7E-04	0.0E+00	1.2E-02	0.0E+00
	Fossil depletion	FD	kg oil eq	4.2E-03	1.4E-03	1.1E-02	0.0E+00	1.8E-02	0.0E+00
& CED	Cumulative energy demand	CED	MJ	2.5E-01	6.4E-02	9.7E-01	0.0E+00	1.1E+00	0.0E+00
Endpoints	Human Health		DALY	5.0E-08	1.4E-08	8.2E-08	0.0E+00	1.4E-07	0.0E+00
	Ecosystems		species.yr	1.6E-10	6.4E-11	3.3E-10	0.0E+00	9.3E-10	0.0E+00
	Resources		$	1.4E-03	2.4E-04	1.9E-03	0.0E+00	3.8E-03	0.0E+00
	Single score		Pt	2.8E-03	6.2E-04	4.3E-03	0.0E+00	8.0E-03	0.0E+00
Equivalent annual cost		EAC	€	3.2E-02	2.4E-02	2.3E-02	0.0E+00	5.2E-02	1.1E-01

CALAFELL							
Indicator			Units	Concrete pipes		Plastic pipes	
				Production	Installation	Production	Installation
Midpoints	Global warming	GW	kg CO_2 eq	2.3E-03	2.0E-02	1.4E-02	2.9E-02
	Ozone depletion	OD	kg CFC-11 eq	8.7E-11	1.4E-09	2.1E-10	4.1E-09
	Terrestrial acidification	TA	kg SO_2 eq	4.8E-06	5.9E-05	4.8E-05	1.5E-04
	Freshwater eutrophication	FE	kg P eq	2.4E-07	1.4E-06	6.5E-07	3.3E-06
	Marine eutrophication	ME	kg N eq	2.4E-07	3.2E-06	1.1E-06	8.5E-06
	Human toxicity	HT	kg 1,4-DB eq	3.5E-04	1.9E-03	6.2E-04	3.9E-03
	Photochemical oxidant formation	POF	kg NMVOC	5.6E-06	8.7E-05	6.0E-05	2.4E-04
	Particulate matter formation	PMF	kg PM10 eq	2.1E-06	2.7E-05	1.6E-05	7.2E-05
	Terrestrial ecotoxicity	TET	kg 1,4-DB eq	5.7E-08	7.4E-07	2.0E-07	2.2E-06
	Freshwater ecotoxicity	FET	kg 1,4-DB eq	7.7E-06	4.3E-05	2.4E-05	1.1E-04
	Marine ecotoxicity	MET	kg 1,4-DB eq	7.8E-06	4.6E-05	2.2E-05	1.2E-04
	Ionising radiation	IR	kBq U^{235} eq	3.0E-04	2.5E-03	9.7E-04	5.6E-03
	Agricultural land occupation	ALO	m^2a	1.3E-04	6.0E-03	6.1E-04	2.5E-02
	Urban land occupation	ULO	m^2a	2.2E-05	3.1E-04	2.1E-05	1.1E-03
	Natural land transformation	NLT	m^2	3.4E-07	6.8E-06	5.6E-07	2.5E-05
	Water depletion	WD	m^3	3.1E-05	5.5E-04	3.3E-05	2.2E-03
	Metal depletion	MD	kg Fe eq	2.3E-04	6.4E-04	1.4E-04	1.8E-03
	Fossil depletion	FD	kg oil eq	2.6E-04	3.7E-03	1.1E-02	9.9E-03
& CED	Cumulative energy demand	CED	MJ	1.6E-02	2.2E-01	5.3E-01	6.0E-01
Endpoints	Human Health		DALY	4.1E-09	3.7E-08	2.4E-08	6.2E-08
	Ecosystems		species.yr	2.1E-11	2.5E-10	1.2E-10	6.0E-10
	Resources		$	5.9E-05	6.5E-04	1.8E-03	1.8E-03
Single score			Pt	1.7E-04	1.7E-03	2.8E-03	3.8E-03
Economic cost	Equivalent annual cost	EAC	€	6.3E-03	1.2E-01	6.6E-02	5.7E-01

	CALAFELL								
	Indicator		Units	Appurtenance		Pumping energy	Cleaning	Reposition	Others
				Production	Installation				
Midpoints	Global warming	GW	kg CO_2 eq	1.5E-02	5.9E-03	1.7E-01	2.1E-03	6.5E-02	0.0E+00
	Ozone depletion	OD	kg CFC-11 eq	8.0E-10	4.5E-10	1.4E-08	3.2E-10	5.6E-09	0.0E+00
	Terrestrial acidification	TA	kg SO_2 eq	5.3E-05	2.0E-05	1.0E-03	6.1E-06	2.7E-04	0.0E+00
	Freshwater eutrophication	FE	kg P eq	9.3E-06	3.5E-07	5.4E-05	2.4E-07	1.4E-05	0.0E+00
	Marine eutrophication	ME	kg N eq	3.4E-06	1.1E-06	3.0E-05	3.2E-07	1.4E-05	0.0E+00
	Human toxicity	HT	kg 1,4-DB eq	1.7E-02	4.8E-04	4.5E-02	3.1E-04	2.2E-02	0.0E+00
	Photochemical oxidant formation	POF	kg NMVOC	4.4E-05	3.0E-05	5.7E-04	8.9E-06	3.8E-04	0.0E+00
	Particulate matter formation	PMF	kg PM10 eq	3.1E-05	9.1E-06	3.3E-04	2.9E-06	1.3E-04	0.0E+00
	Terrestrial ecotoxicity	TET	kg 1,4-DB eq	1.9E-06	2.2E-07	4.6E-06	3.6E-07	4.6E-06	0.0E+00
	Freshwater ecotoxicity	FET	kg 1,4-DB eq	3.5E-04	1.1E-05	1.0E-03	7.3E-06	4.9E-04	0.0E+00
	Marine ecotoxicity	MET	kg 1,4-DB eq	3.5E-04	1.2E-05	1.0E-03	9.6E-06	5.1E-04	0.0E+00
	Ionising radiation	IR	kBq U^{235} eq	3.3E-03	5.4E-04	1.3E-01	3.2E-04	1.0E-02	0.0E+00
	Agricultural land occupation	ALO	m^2a	5.5E-04	1.2E-04	3.8E-03	1.1E-05	2.6E-02	0.0E+00
	Urban land occupation	ULO	m^2a	1.9E-04	4.3E-05	7.4E-04	4.8E-05	1.4E-03	0.0E+00
	Natural land transformation	NLT	m^2	2.8E-06	1.5E-06	2.5E-05	7.6E-07	3.0E-05	0.0E+00
	Water depletion	WD	m^3	1.9E-04	7.3E-05	1.4E-03	8.7E-06	2.5E-03	0.0E+00
	Metal depletion	MD	kg Fe eq	7.4E-03	1.5E-04	3.9E-03	1.1E-04	9.8E-03	0.0E+00
	Fossil depletion	FD	kg oil eq	3.5E-03	1.1E-03	5.0E-02	7.3E-04	2.5E-02	0.0E+00
& CED	Cumulative energy demand	CED	MJ	2.1E-01	5.3E-02	4.3E+00	3.6E-02	1.4E+00	0.0E+00
Endpoints	Human Health		DALY	4.1E-08	1.1E-08	3.6E-07	4.0E-09	1.4E-07	0.0E+00
	Ecosystems		species.yr	1.4E-10	5.2E-11	1.5E-09	1.9E-11	9.1E-10	0.0E+00
	Resources		$	1.1E-03	2.0E-04	8.6E-03	1.3E-04	4.9E-03	0.0E+00
Single score			Pt	2.3E-03	5.1E-04	1.9E-02	2.5E-04	9.4E-03	0.0E+00
Equivalent annual cost		EAC	€	2.4E-02	2.1E-02	4.6E-02	3.3E-02	5.1E-03	1.2E-01

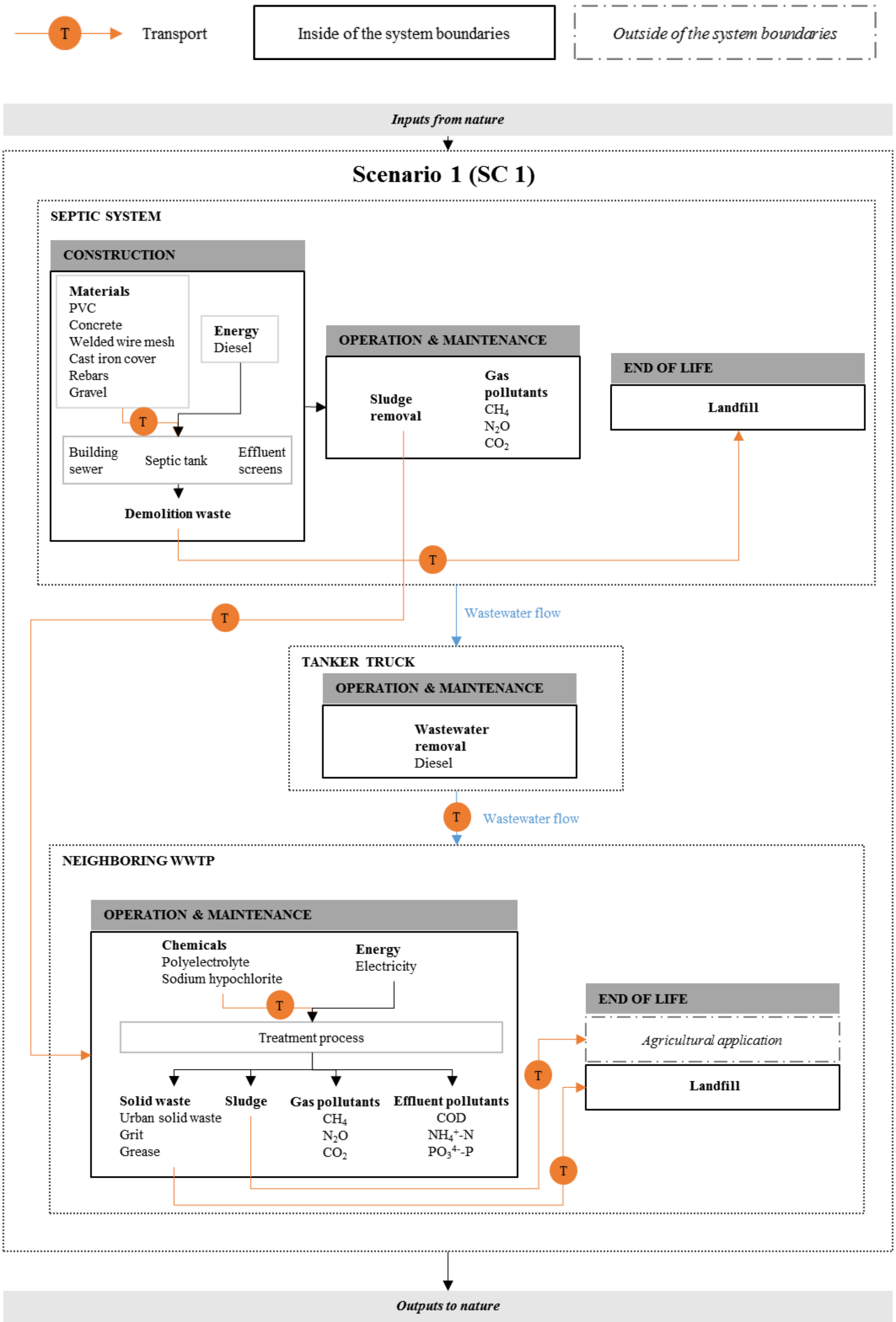
Appendix 5.3 Eco-efficiency portfolio illustrating the differences amongst life cycle stages in Betanzos and Calafell through the GW indicator

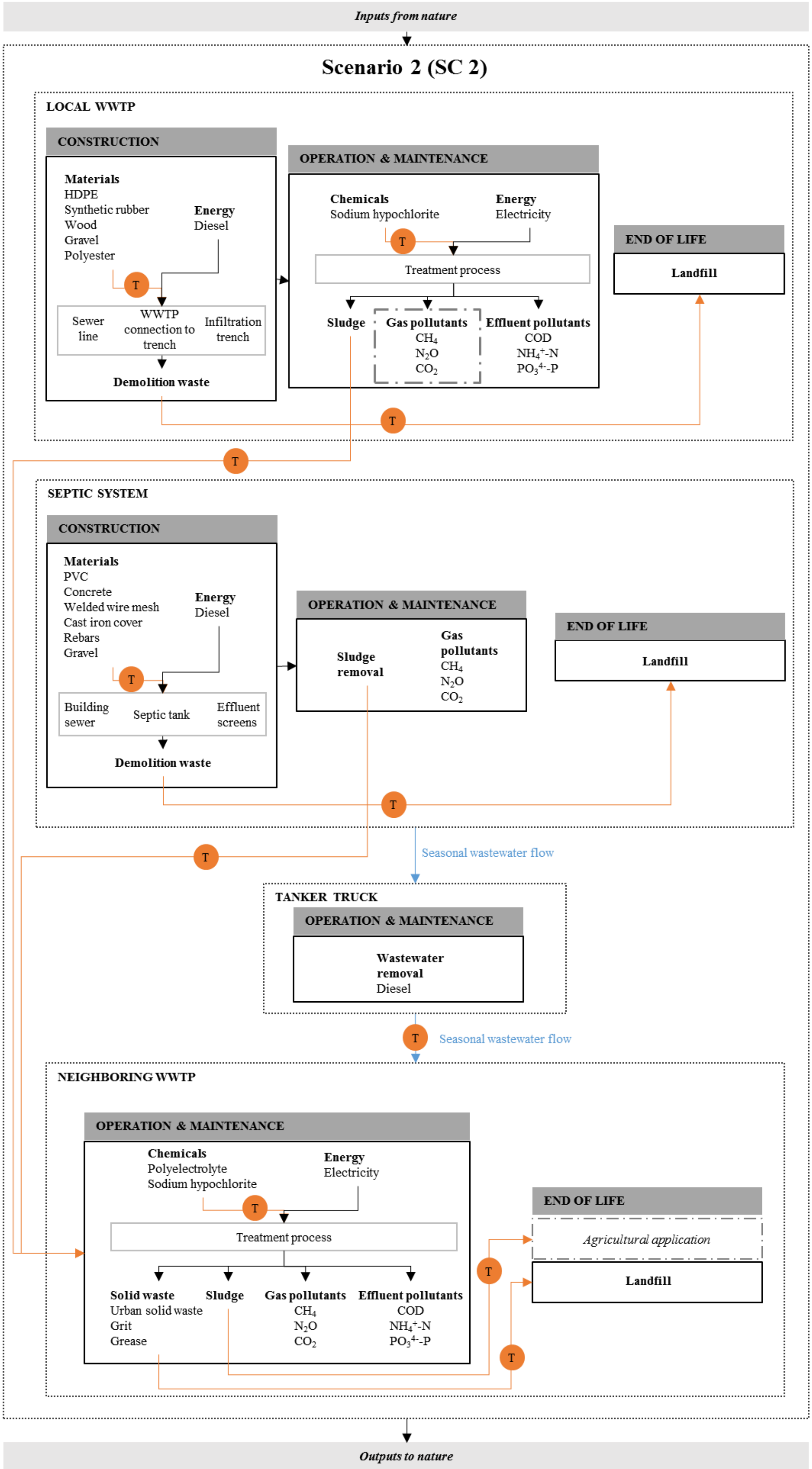


Total sewer = Production + Installation + O&M

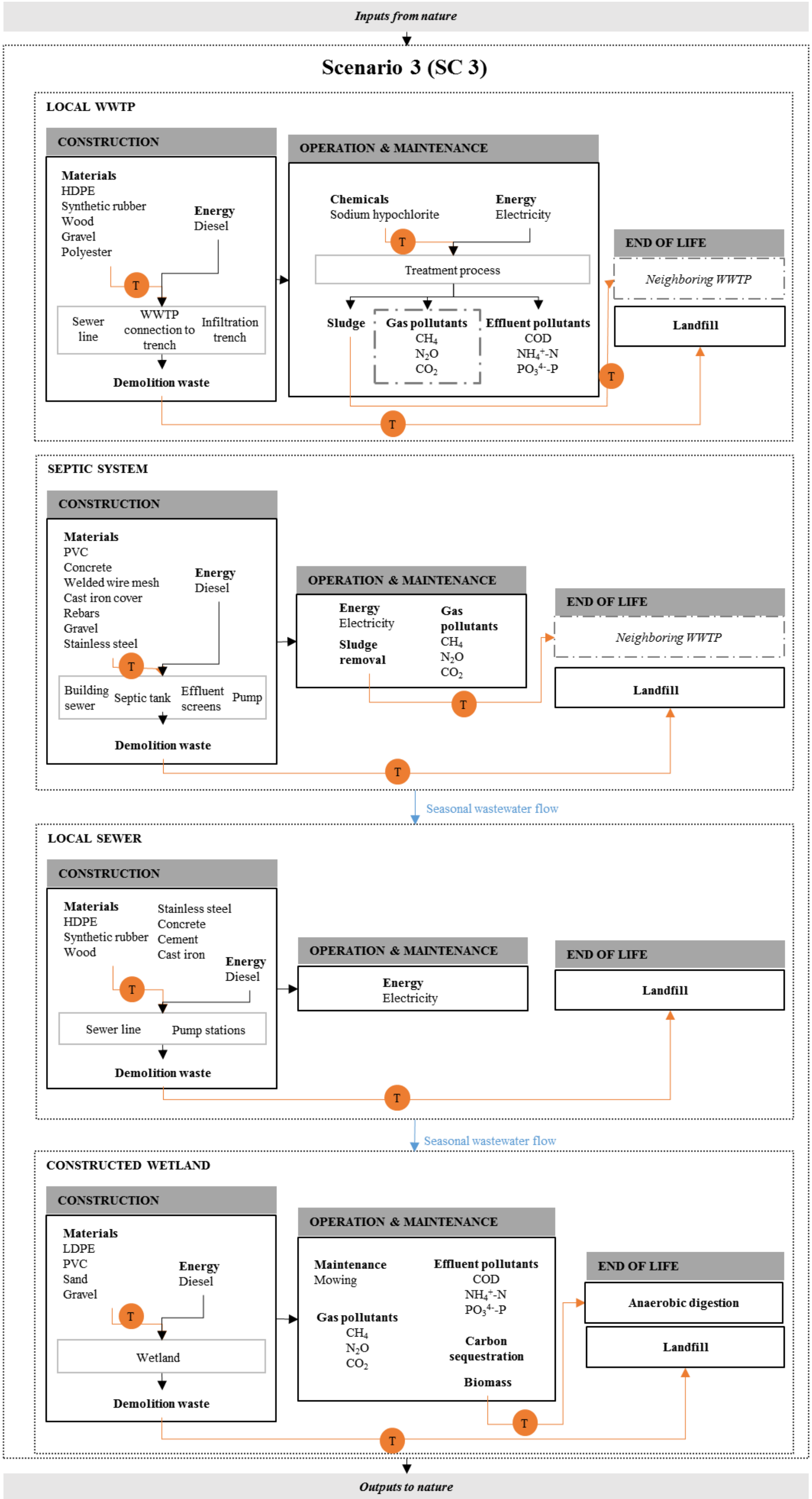
Appendix 6. Supporting data related to Chapter 9

Appendix 6.1 System boundaries of treatment scenarios 1, 2, 3, 4 and 5

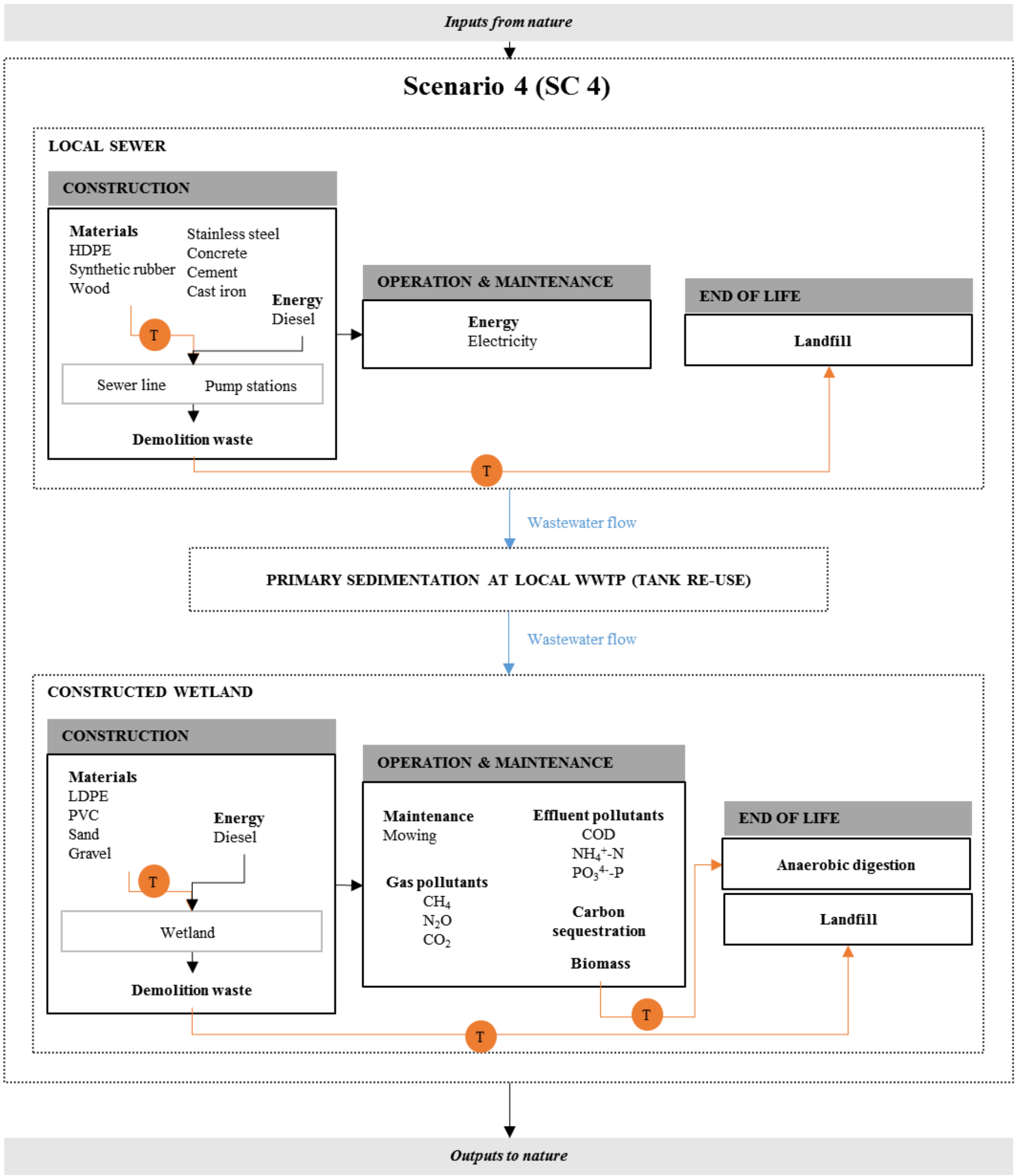




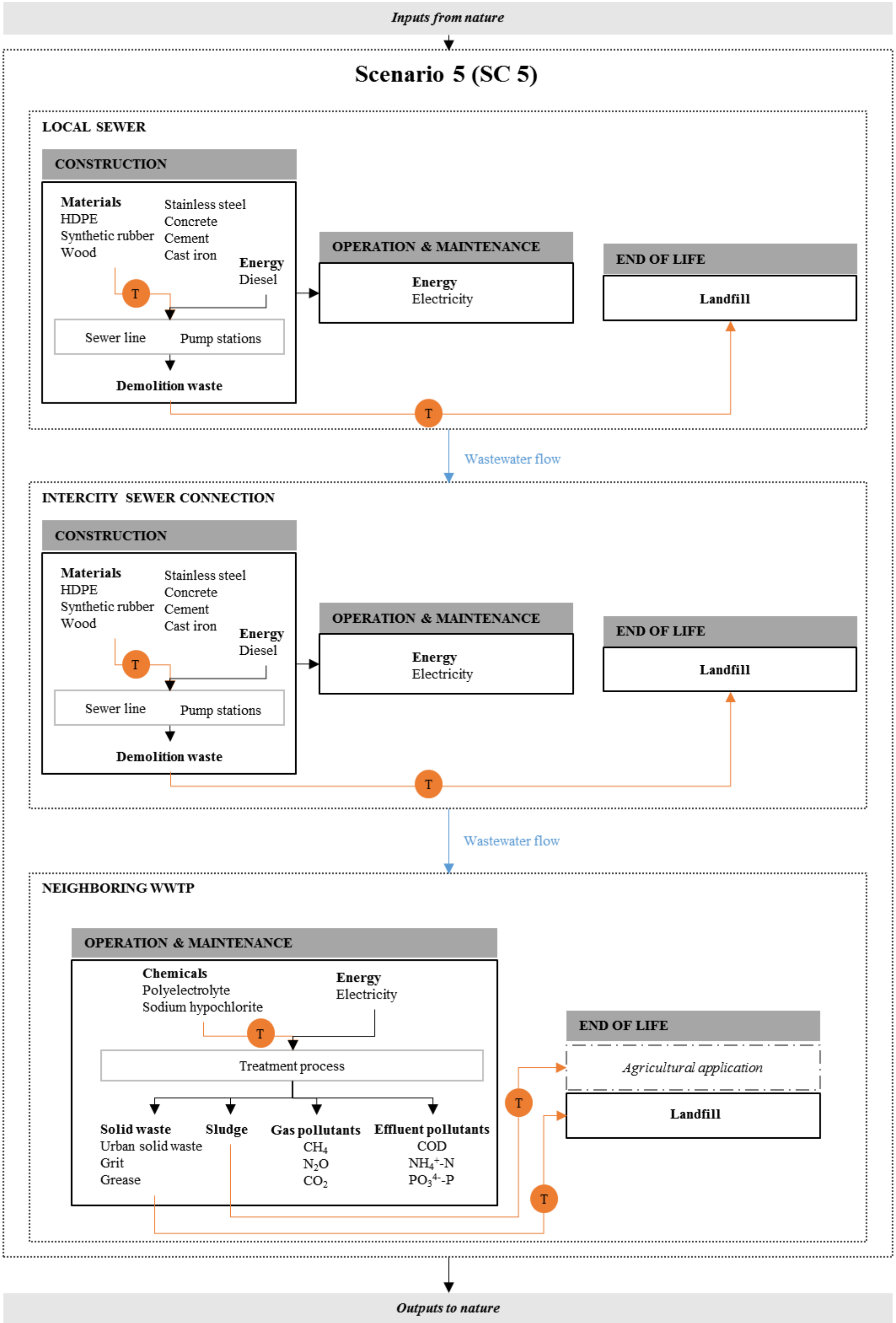
Inputs from nature
Scenario 2 (SC 2)
LOCAL WWTP
CONSTRUCTION
Materials
HDPE
Synthetic rubber
Wood
Gravel
Polyester
Energy
Diesel
T
Sewer line
WWTP connection to trench
Infiltration trench
Demolition waste
OPERATION & MAINTENANCE
Chemicals
Sodium hypochlorite
Energy
Electricity
Treatment process
Sludge
Gas pollutants
CH_4
N_2O
CO_2
Effluent pollutants
COD
NH_4^+-N
PO_3^{4-}-P
END OF LIFE
Landfill
SEPTIC SYSTEM
CONSTRUCTION
Materials
PVC
Concrete
Welded wire mesh
Cast iron cover
Rebars
Gravel
Energy
Diesel
Building sewer
Septic tank
Effluent screens
Demolition waste
OPERATION & MAINTENANCE
Sludge removal
Gas pollutants
CH_4
N_2O
CO_2
END OF LIFE
Landfill
Seasonal wastewater flow
TANKER TRUCK
OPERATION & MAINTENANCE
Wastewater removal
Diesel
Seasonal wastewater flow
NEIGHBORING WWTP
OPERATION & MAINTENANCE
Chemicals
Polyelectrolyte
Sodium hypochlorite
Energy
Electricity
Treatment process
Solid waste
Urban solid waste
Grit
Grease
Sludge
Gas pollutants
CH_4
N_2O
CO_2
Effluent pollutants
COD
NH_4^+-N
PO_3^{4-}-P
END OF LIFE
Agricultural application
Landfill
Outputs to nature



Inputs from nature
Scenario 3 (SC 3)
LOCAL WWTP
CONSTRUCTION
Materials
HDPE
Synthetic rubber
Wood
Gravel
Polyester
Energy
Diesel
T
Sewer line
WWTP connection to trench
Infiltration trench
Demolition waste
OPERATION & MAINTENANCE
Chemicals
Sodium hypochlorite
Energy
Electricity
T
Treatment process
Sludge
Gas pollutants
CH_4
N_2O
CO_2
Effluent pollutants
COD
NH_4^+-N
PO_3^{4-}-P
END OF LIFE
Neighboring WWTP
Landfill
T
T
SEPTIC SYSTEM
CONSTRUCTION
Materials
PVC
Concrete
Welded wire mesh
Cast iron cover
Rebars
Gravel
Stainless steel
Energy
Diesel
T
Building sewer
Septic tank
Effluent screens
Pump
Demolition waste
OPERATION & MAINTENANCE
Energy
Electricity
Sludge removal
Gas pollutants
CH_4
N_2O
CO_2
T
END OF LIFE
Neighboring WWTP
Landfill
T
Seasonal wastewater flow
LOCAL SEWER
CONSTRUCTION
Materials
HDPE
Synthetic rubber
Wood
Stainless steel
Concrete
Cement
Cast iron
Energy
Diesel
T
Sewer line
Pump stations
Demolition waste
OPERATION & MAINTENANCE
Energy
Electricity
END OF LIFE
Landfill
T
Seasonal wastewater flow
CONSTRUCTED WETLAND
CONSTRUCTION
Materials
LDPE
PVC
Sand
Gravel
Energy
Diesel
T
Wetland
Demolition waste
OPERATION & MAINTENANCE
Maintenance
Mowing
Effluent pollutants
COD
NH_4^+-N
PO_3^{4-}-P
Gas pollutants
CH_4
N_2O
CO_2
Carbon sequestration
Biomass
END OF LIFE
Anaerobic digestion
Landfill
T
T
Outputs to nature



Inputs from nature
Scenario 4 (SC 4)
LOCAL SEWER
CONSTRUCTION
Materials
HDPE
Synthetic rubber
Wood
Stainless steel
Concrete
Cement
Cast iron
Energy
Diesel
T
Sewer line
Pump stations
Demolition waste
OPERATION & MAINTENANCE
Energy
Electricity
END OF LIFE
Landfill
T
Wastewater flow
PRIMARY SEDIMENTATION AT LOCAL WWTP (TANK RE-USE)
Wastewater flow
CONSTRUCTED WETLAND
CONSTRUCTION
Materials
LDPE
PVC
Sand
Gravel
Energy
Diesel
T
Wetland
Demolition waste
OPERATION & MAINTENANCE
Maintenance
Mowing
Effluent pollutants
COD
NH_4^+-N
PO_3^{4-}-P
Gas pollutants
CH_4
N_2O
CO_2
Carbon sequestration
Biomass
END OF LIFE
Anaerobic digestion
Landfill
T
T
Outputs to nature



Inputs from nature
Scenario 5 (SC 5)
LOCAL SEWER
CONSTRUCTION
Materials
HDPE
Synthetic rubber
Wood
Stainless steel
Concrete
Cement
Cast iron
Energy
Diesel
T
Sewer line
Pump stations
Demolition waste
OPERATION & MAINTENANCE
Energy
Electricity
END OF LIFE
Landfill
T
Wastewater flow
INTERCITY SEWER CONNECTION
CONSTRUCTION
Materials
HDPE
Synthetic rubber
Wood
Stainless steel
Concrete
Cement
Cast iron
Energy
Diesel
T
Sewer line
Pump stations
Demolition waste
OPERATION & MAINTENANCE
Energy
Electricity
END OF LIFE
Landfill
T
Wastewater flow
NEIGHBORING WWTP
OPERATION & MAINTENANCE
Chemicals
Polyelectrolyte
Sodium hypochlorite
Energy
Electricity
T
Treatment process
Solid waste
Urban solid waste
Grit
Grease
Sludge
Gas pollutants
CH_4
N_2O
CO_2
Effluent pollutants
COD
NH_4^+-N
PO_3^{4-}-P
END OF LIFE
Agricultural application
Landfill
T
T
Outputs to nature

Appendix 6.2 Life cycle inventory data of each scenario (nd: no data)

Life cycle stage	System	Sub-system	Data per year		Data per m³, based on 50,316 m³/year	
SCENARIO 1 (SC1)						
Septic system (246 units)						
Construction	Building sewer	PVC pipe	47	kg	9.3E-04	kg
	Septic tank	Concrete	54	m^3	1.1E-03	m^3
		Excavated soil	1,490	kg	3.0E-02	kg
		Welded wire mesh	324	kg	6.4E-03	kg
		Cast iron manhole cover	373	kg	7.4E-03	kg
		Rebars	255	kg	5.1E-03	kg
		Gravel	36,235	kg	7.2E-01	kg
		Diesel	11,023	MJ	2.2E-01	MJ
	Effluent screens	PVC mesh screen	38	kg	7.5E-04	kg
Operation	Maintenance	Sludge removal	3,465	kg	6.9E-02	kg
	Emissions to air	Methane (CH_4)	2,989	kg	5.9E-02	kg
		Nitrous oxide (N_2O)	1.1	kg	2.1E-05	kg
		Carbon dioxide (CO_2)	7,132	kg	1.4E-01	kg
Transport	Raw material	To construction site	4,978	tkm	9.9E-02	tkm
	Sludge	To WWTP	45	tkm	9.0E-04	tkm
	Demolition waste	To landfill	8,371	tkm	1.7E-01	tkm
End of life	Demolition waste	Sanitary landfill	167,423	kg	3.3E+00	kg
	Sludge	Land application	nd	m^3	nd	m^3
Tanker truck						
Operation	Wastewater removal	Diesel	60,403	MJ	1.2E+00	MJ
	Wastewater transport	To WWTP	654,108	tkm	1.3E+01	tkm
Neighboring WWTP						
Operation	Energy and chemicals	Electricity	28,290	kWh	5.6E-01	kWh
		Cationic polyelectrolyte	42	kg	8.4E-04	kg
		Sodium hypochlorite	4	kg	7.0E-05	kg
	Emissions to seawater	Ammonium (NH_4^+-N)	264	kg	5.3E-03	kg
		Nitrite (NO_2-N)	36	kg	7.1E-04	kg
		Nitrate (NO_3-N)	781	kg	1.6E-02	kg
		Phosphorus (PO_3^{4-}-P)	289	kg	5.7E-03	kg
	Emissions to air	Methane (CH_4)	nd	kg	nd	kg
		Nitrous oxide (N_2O)	nd	kg	nd	kg
		Carbon dioxide (CO_2)	nd	kg	nd	kg
	Solid waste generation	Sludge	46,794	kg	9.3E-01	kg
		Urban solid waste	1,404	kg	2.8E-02	kg
		Grit	1,343	kg	2.7E-02	kg
		Grease	189	kg	3.8E-03	kg
Transport	Raw material	To WWTP	1.4	tkm	2.7E-05	tkm
	Solid waste	To landfill	147	tkm	2.9E-03	tkm
		To agriculture	1,508	tkm	3.0E-02	tkm
End of life	Sludge	Land application	nd	m^3	nd	m^3
	Solid waste	Sanitary landfill	2,936	kg	5.8E-02	kg
Demolition of existing infrastructure (local WWTP – internal pipe)						
End of life	Demolition waste	To landfill	1.7	tkm	3.4E-05	tkm
		Sanitary landfill	34	kg	6.7E-04	kg

SCENARIO 2 (SC2)						
Life cycle stage	**System**	**Sub-system**	**Data per year**		**Data per m³, based on 50,316 m³/year**	
Septic system (172 units)						
Construction	Building sewer	PVC pipe	33	kg	6.5E-04	kg
	Septic tank	Concrete	43	m^3	8.5E-04	m^3
		Excavated soil	1,007	kg	2.0E-02	kg
		Welded wire mesh	257	kg	5.1E-03	kg
		Cast iron manhole cover	261	kg	5.2E-03	kg
		Rebars	191	kg	3.8E-03	kg
		Gravel	30,037	kg	6.0E-01	kg
		Diesel	8,981	MJ	1.8E-01	MJ
	Effluent screens	PVC mesh screen	26	kg	5.2E-04	kg
Operation	Maintenance	Sludge removal	2,531	kg	5.0E-02	kg
	Emissions to air	Methane (CH_4)	1,889	kg	3.8E-02	kg
		Nitrous oxide (N_2O)	0.6	kg	1.2E-05	kg
		Carbon dioxide (CO_2)	4,166	kg	8.3E-02	kg
Transport	Raw material	To construction site	4,005	tkm	8.0E-02	tkm
	Sludge	To WWTP	33	tkm	6.5E-04	tkm
	Demolition waste	To landfill	6,726	tkm	1.3E-01	tkm
End of life	Demolition waste	Sanitary landfill	134,513	kg	2.7E+00	kg
	Sludge	Land application	nd	m^3	nd	m^3
Tanker truck						
Operation	Wastewater removal	Diesel	60,403	MJ	1.2E+00	MJ
	Wastewater transport	To WWTP	413,405	tkm	8.2E+00	tkm
Neighboring WWTP						
Operation	Energy and chemicals	Electricity	17,879	kWh	3.6E-01	kWh
		Cationic polyelectrolyte	27	kg	5.3E-04	kg
		Sodium hypochlorite	2	kg	4.5E-05	kg
	Emissions to seawater	Ammonium (NH_4^+-N)	167	kg	3.3E-03	kg
		Nitrite (NO_2-N)	23	kg	4.5E-04	kg
		Nitrate (NO_3-N)	493	kg	9.8E-03	kg
		Phosphorus (PO_3^{4-} -P)	183		3.6E-03	
	Emissions to air	Methane (CH_4)	nd	kg	nd	kg
		Nitrous oxide (N_2O)	nd	kg	nd	kg
		Carbon dioxide (CO_2)	nd	kg	nd	kg
	Solid waste generation	Sludge	29,574	kg	5.9E-01	kg
		Urban solid waste	887	kg	1.8E-02	kg
		Grit	849	kg	1.7E-02	kg
		Grease	120	kg	2.4E-03	kg
Transport	Raw material	To WWTP	0.9	tkm	1.7E-05	tkm
	Solid waste	To landfill	93	tkm	1.8E-03	tkm
		To agriculture	963	tkm	1.9E-02	tkm
End of life	Sludge	Land application	nd	m^3	nd	m^3
	Solid waste	Sanitary landfill	1,856	kg	3.7E-02	kg
Local sewer						
Construction	Gravity sewer line (880 meters)	HDPE pipe	141	kg	2.8E-03	kg
		Synthetic rubber	2.1	kg	4.1E-05	kg
		Native soil	11	m^3	2.1E-04	m^3
		Diesel	1,672	MJ	3.3E-02	MJ
		Wood support	0.26	m^3	5.2E-06	m^3
	Pavement	Gravel	3,520	kg	7.0E-02	kg

SCENARIO 2 (SC2)						
Life cycle stage	**System**	**Sub-system**	**Data per year**		**Data per m^3, based on 50,316 m^3/year**	
		Concrete	4,627	kg	9.2E-02	kg
		Asphalt	3,722	kg	7.4E-02	kg
		Diesel	394	MJ	7.8E-03	MJ
Transport	Raw material	To construction site	365	tkm	7.3E-03	tkm
	Demolition waste	Demolition waste	608	tkm	1.2E-02	tkm
End of life	To landfill	To landfill	12,169	kg	2.4E-01	kg
Local WWTP						
Construction	WWTP connection to sand filter (208 meters)	HDPE pipe	33	kg	6.6E-04	kg
		Synthetic rubber	0.5	kg	9.8E-06	kg
		Native soil	2	m^3	5.0E-05	m^3
		Diesel	395	MJ	7.9E-03	MJ
		Wood support	0.1	m^3	1.2E-06	m^3
	Infiltration trench	Gravel	7,840	kg	1.6E-01	kg
		Perforated PVC pipe	2.9	kg	5.8E-05	kg
		Native soil	3,230	kg	6.4E-02	kg
		Polyester liner	2.0	kg	4.0E-05	kg
		Diesel	36,874	MJ	7.3E-01	MJ
Operation	Energy and chemicals	Electricity	6,500	kWh	1.3E-01	kWh
		Sodium hypochlorite	120	kg	2.4E-03	kg
	Maintenance	Sludge removal	3,333	kg	6.6E-02	kg
	Emissions to groundwater	Total nitrogen (TN)	802	kg	1.6E-02	kg
		Total phosphorus (TP)	277	kg	5.5E-03	kg
	Emissions to air	Methane (CH_4)	nd	kg	nd	kg
		Nitrous oxide (N_2O)	nd	kg	nd	kg
		Carbon dioxide (CO_2)	nd	kg	nd	kg
Transport	Raw material	To construction site	241	tkm	4.8E-03	tkm
	Sludge	To WWTP	43	tkm	8.6E-04	tkm
	Demolition waste	To landfill	3	tkm	6.6E-05	tkm
End of life	Sludge	Land application	nd	m^3	nd	m^3
	Demolition waste	Sanitary landfill	66	kg	1.3E-03	kg

SCENARIO 3 (SC3)						
Life cycle stage	**System**	**Sub-system**	**Data per year**		**Data per m³, based on 50,316 m³/year**	
		Septic system (172 units)				
Construction	Building sewer	PVC pipe	33	kg	6.5E-04	kg
	Septic tank	Concrete	43	m³	8.5E-04	m³
		Excavated soil	1,007	kg	2.0E-02	kg
		Welded wire mesh	257	kg	5.1E-03	kg
		Cast iron manhole cover	261	kg	5.2E-03	kg
		Rebars	191	kg	3.8E-03	kg
		Gravel	30,037	kg	6.0E-01	kg
		Diesel	8,981	MJ	1.8E-01	MJ
	Effluent screens	PVC mesh screen	26	kg	5.2E-04	kg
	Pump	Stainless steel	77	kg	1.5E-03	kg
	Service lateral	PVC pipe	110	kg	2.2E-03	kg
Operation	Pumping	Electricity	16452	kWh	3.3E-01	kWh
	Maintenance	Sludge removal	2,531	kg	5.0E-02	kg
	Emissions to air	Methane (CH_4)	1,889	kg	3.8E-02	kg
		Nitrous oxide (N_2O)	0.6	kg	1.2E-05	kg
		Carbon dioxide (CO_2)	4,166	kg	8.3E-02	kg
Transport	Raw material	To construction site	4,011	tkm	8.0E-02	tkm
	Sludge	To WWTP	33	tkm	6.5E-04	tkm
	Demolition waste	To landfill	6,735	tkm	1.3E-01	tkm
End of life	Demolition waste	Sanitary landfill	134,701	kg	2.7E+00	kg
	Sludge	Land application	nd	m³	nd	m³
		Local sewer				
Construction	Pressurized sewer line (7.4 km)	HDPE pipe	473	kg	9.4E-03	kg
		Synthetic rubber	8	kg	1.5E-04	kg
		Native soil	51	m³	1.0E-03	m³
		Diesel	8,089	MJ	1.6E-01	MJ
		Wood support	3	m³	4.9E-05	m³
	Pump stations (7 units + 2 backup pumps each; 1.5 kW)	Stainless steel	14	kg	2.8E-04	kg
		Concrete block	923	kg	1.8E-02	kg
		Cement	69	m³	1.4E-03	m³
		Concrete	0.1	MJ	2.3E-06	MJ
		Cast iron	39	m³	7.7E-04	m³
	Pavement	Gravel	18,254	kg	7.0E-02	kg
		Concrete	4,627	kg	9.2E-02	kg
		Asphalt	3,722	kg	7.4E-02	kg
		Diesel	394	MJ	7.8E-03	MJ
Operation	Pumping	Electricity	19,902	kWh	4.0E-01	kWh
Transport	Raw material	To construction site	920	tkm	1.8E-02	tkm
	Demolition waste	To landfill	1,533	tkm	3.0E-02	tkm
End of life	Demolition waste	Sanitary landfill	30,662	kg	6.1E-01	kg
		Constructed wetland (3.5 ha)				
Construction	Plastic liner	LDPE	1,135	kg	2.5E-02	kg
	Pipes	PVC	79	kg	1.7E-03	kg
	Layer under wetland	Sand	17,013	kg	3.7E-01	kg
	Medium	Gravel	2,319,436	kg	5.1E+01	kg
	Excavation	Diesel	88,391	MJ	1.9E+00	MJ
Operation	Mowing	Rotary mower	3.2	ha	7.0E-05	ha

SCENARIO 3 (SC3)						
Life cycle stage	**System**	**Sub-system**	**Data per year**		**Data per m^3, based on 50,316 m^3/year**	
	Vegetation	*Phragmites australis* (dry weight)	15,134	kg	1.3E+00	kg
	Emissions to air	Methane (CH_4)	1,587	kg	3.2E-02	kg
		Nitrous oxide (N_2O)	16	kg	3.2E-04	kg
		Carbon dioxide (CO_2)	21,602	kg	4.3E-01	kg
	Emissions to groundwater	Ammonium (NH_4^+-N)	159	kg	3.2E-03	kg
		Nitrate (NO_3^--N	1,113	kg	2.2E-02	kg
		Total phosphorus (TP)	127	kg	2.5E-03	kg
	Avoided emissions	Carbon sequestration (CO_2)	-24,416	kg	-2.1E+00	kg
Transport	Raw material	To construction site	70,130	tkm	1.5E+00	tkm
	Biomass	To neighboring WWTP	151	tkm	1.3E-02	tkm
End of life	Biomass	Anaerobic digestion	15,134	kg	1.3E+00	kg
Local WWTP						
Construction	WWTP connection to sand filter (208 meters)	HDPE pipe	33	kg	6.6E-04	kg
		Synthetic rubber	0.5	kg	9.8E-06	kg
		Native soil	2	m^3	5.0E-05	m^3
		Diesel	395	MJ	7.9E-03	MJ
		Wood support	0.1	m^3	1.2E-06	m^3
	Infiltration trench	Gravel	7,840	kg	1.6E-01	kg
		Perforated PVC pipe	2.9	kg	5.8E-05	kg
		Native soil	3,230	kg	6.4E-02	kg
		Polyester liner	2.0	kg	4.0E-05	kg
		Diesel	36,874	MJ	7.3E-01	MJ
Operation	Energy and chemicals	Electricity	6,500	kWh	1.3E-01	kWh
		Sodium hypochlorite	120	kg	2.4E-03	kg
	Maintenance	Sludge removal	3,333	kg	6.6E-02	kg
	Emissions to groundwater	Total nitrogen (TN)	802	kg	1.6E-02	kg
		Total phosphorus (TP)	154	kg	3.1E-03	kg
	Emissions to air	Methane (CH_4)	nd	kg	nd	kg
		Nitrous oxide (N_2O)	nd	kg	nd	kg
		Carbon dioxide (CO_2)	nd	kg	nd	kg
Transport	Raw material	To construction site	249	tkm	5.0E-03	tkm
	Sludge	To WWTP	43	tkm	8.6E-04	tkm
	Demolition waste	To landfill	16	tkm	3.1E-04	tkm
End of life	Sludge	Land application	nd	m^3	nd	m^3
	Demolition waste	Sanitary landfill	345	kg	6.9E-03	kg

SCENARIO 4 (SC4)						
Life cycle stage	**System**	**Sub-system**	**Data per year**		**Data per m^3, based on 50,316 m^3/year**	
Local sewer						
Construction	Pressurized sewer line (7.4 km)	HDPE pipe	2,681	kg	5.3E-02	kg
		Synthetic rubber	44	kg	8.7E-04	kg
		Native soil	160	m^3	3.2E-03	m^3
		Diesel	25,331	MJ	5.0E-01	MJ
		Wood support	3	m^3	6.0E-05	m^3
	Pump stations (7 units + 2 backup pumps each; 1.5 kW)	Stainless steel	10	kg	2.0E-04	kg
		Concrete block	660	kg	1.3E-02	kg
		Cement	49	m^3	9.8E-04	m^3
		Concrete	0	MJ	1.7E-06	MJ
		Cast iron	28	m^3	5.5E-04	m^3
	Pavement	Gravel	49,670	kg	9.9E-01	kg
		Concrete	65,291	kg	1.3E+00	kg
		Asphalt	52,526	kg	1.0E+00	kg
		Diesel	5,554	MJ	1.1E-01	MJ
Operation	Pumping	Electricity	9,531	kWh	1.9E-01	kWh
Transport	Raw material	To construction site	5,202	tkm	1.0E-01	tkm
	Demolition waste	To landfill	8,670	tkm	1.7E-01	tkm
End of life	Demolition waste	Sanitary landfill	173,395	kg	3.4E+00	kg
Constructed wetland (3.8 ha)						
Construction	Plastic liner	LDPE	1,333	kg	2.7E-02	kg
	Pipes	PVC	86	kg	1.7E-03	kg
	Layer under wetland	Sand	20,000	kg	4.0E-01	kg
	Medium	Gravel	2,726,589	kg	5.4E+01	kg
	Excavation	Diesel	112,048	MJ	2.2E+00	MJ
Operation	Mowing	Rotary mower	3.7	ha	7.4E-05	ha
	Vegetation	*Phragmites australis* (dry weight)	17,790	kg	3.5E-01	kg
	Emissions to air	Methane (CH_4)	1,587	kg	3.2E-02	kg
		Nitrous oxide (N_2O)	16	kg	3.2E-04	kg
		Carbon dioxide (CO_2)	21,602	kg	4.3E-01	kg
	Emissions to groundwater	Ammonium (NH_4^+-N)	159.0	kg	3.2E-03	kg
		Nitrate (NO_3^--N	1,113	kg	2.2E-02	kg
		Total phosphorus (TP)	127	kg	2.5E-03	kg
	Avoided emissions	Carbon sequestration (CO_2)	-28,702	kg	-5.7E-01	kg
Transport	Raw material	To construction site	82,440	tkm	1.6E+00	tkm
	Biomass	To neighboring WWTP	178	tkm	3.5E-03	tkm
End of life	Biomass	Anaerobic digestion	17,790	kg	3.5E-01	kg
Demolition of existing infrastructure (septic tanks & WWTP connection)						
End of life	Demolition waste	To landfill	394	tkm	7.8E-03	tkm
		Sanitary landfill	7,879	kg	1.6E-01	kg

SCENARIO 5 (SC5)						
Life cycle stage	System	Sub-system	Data per year		Data per m^3, based on 50,316 m^3/year	
Local sewer						
Construction	Pressurized sewer line (3.1 km) + gravity sewer line (6.9 km)	HDPE pipe	2,681	kg	5.3E-02	kg
		Synthetic rubber	43.7	kg	8.7E-04	kg
		Native soil	160.0	m^3	3.2E-03	m^3
		Diesel	25,331	MJ	5.0E-01	MJ
		Wood support	3.0	m^3	6.0E-05	m^3
	Pump stations (7 units + 2 backup pumps each; 3 kW)	Stainless steel	44	kg	8.8E-04	kg
		Concrete block	923	kg	1.8E-02	kg
		Cement	69	kg	1.4E-03	kg
		Concrete	0.1	m^3	2.3E-06	m^3
		Cast iron	39	kg	7.7E-04	kg
	Pavement	Gravel	49,670	kg	9.9E-01	kg
		Concrete	65,291	kg	1.3E+00	kg
		Asphalt	52,526	kg	1.0E+00	kg
		Diesel	5,554	MJ	1.1E-01	MJ
Operation	Pumping	Electricity	9,531	kWh	1.9E-01	kWh
Transport	Raw material	To construction site	5,202	tkm	1.0E-01	tkm
	Demolition waste	To landfill	8,670	tkm	1.7E-01	tkm
End of life	Demolition waste	Sanitary landfill	173,395	kg	3.4E+00	kg
Intercity sewer connection						
Construction	Pressurized sewer line (6.4 km) + gravity sewer line (20 meters)	HDPE pipe	1,034	kg	2.1E-02	kg
		Synthetic rubber	15.3	kg	3.0E-04	kg
		Native soil	77.2	m^3	1.5E-03	m^3
		Diesel	12,228	MJ	2.4E-01	MJ
		Wood support	1.93	m^3	3.8E-05	m^3
	Pump stations (1 unit + 2 backup pumps; 5.2 kW)	Cast iron	15	kg	2.9E-04	kg
		Concrete block	264	kg	5.2E-03	kg
		Cement	20	kg	3.9E-04	kg
		Concrete	0.03	m^3	6.6E-07	m^3
		Cast iron	11	kg	2.2E-04	kg
	Pavement	Gravel	25,726	kg	5.1E-01	kg
		Concrete	33,817	kg	6.7E-01	kg
		Asphalt	27,205	kg	5.4E-01	kg
		Diesel	2,877	MJ	5.7E-02	MJ
Operation	Pumping	Electricity	18,552	kWh	3.7E-01	kWh
Transport	Raw material	To construction site	2,680	tkm	5.3E-02	tkm
	Demolition waste	To landfill	4,466	tkm	8.9E-02	tkm
End of life	Demolition waste	Sanitary landfill	89,325	kg	1.8E+00	kg
Neighboring WWTP						
Operation	Energy and chemicals	Electricity	28,290	kWh	5.6E-01	kWh
		Cationic polyelectrolyte	42	kg	8.4E-04	kg
		Sodium hypochlorite	4	kg	7.0E-05	kg
	Emissions to seawater	Ammonium (NH_4^+-N)	264	kg	5.3E-03	kg
		Nitrite (NO_2-N)	36	kg	7.1E-04	kg
		Nitrate (NO_3-N)	781	kg	1.6E-02	kg
		Phosphorus (PO_3^{4-}-P)	289	kg	5.7E-03	kg
	Emissions to air	Methane (CH_4)	nd	kg	nd	kg

SCENARIO 5 (SC5)						
Life cycle stage	**System**	**Sub-system**	**Data per year**		**Data per m³, based on 50,316 m³/year**	
		Nitrous oxide (N_2O)	nd	kg	nd	kg
		Carbon dioxide (CO_2)	nd	kg	nd	kg
	Solid waste generation	Sludge	46,794	kg	9.3E-01	kg
		Urban solid waste	1,404	kg	2.8E-02	kg
		Grit	1,343	kg	2.7E-02	kg
		Grease	189	kg	3.8E-03	kg
Transport	Raw material	To WWTP	1.4	tkm	2.7E-05	tkm
	Solid waste	To landfill	147	tkm	2.9E-03	tkm
		To agriculture	1,508	tkm	3.0E-02	tkm
End of life	Sludge	Land application	nd	m^3	nd	m^3
	Solid waste	Sanitary landfill	2,936	kg	5.8E-02	kg
Demolition of existing infrastructure (septic tanks & WWTP connection)						
End of life	Demolition waste	To landfill	6,727	tkm	1.3E-01	tkm
		Sanitary landfill	134,547	kg	2.7E+00	kg

Appendix 6.3 Septic tank and wetland sizing methods

Septic tank sizing

According to Crites and Tchobanoglous (1998):

$$V = 3.65 \times (Q_{average}) \times PF$$

Where is the internal volume of the septic tank (in cubic feet), $Q_{average}$ is the average daily wastewater flow produced by each building (in gallon per day), and PF is the peaking factor, set at 1.5 based on Fuchs et al.(Fuchs et al. 2011).

The tank was sized using the summer wastewater generation for safety reasons, as this flow was expected to be higher than the winter wastewater generation.

The inner height of tank was set at 2 meters, and assumed that the longest side doubled the length of the shortest side. The concrete thickness was 0.2 meters (Fuchs et al. 2011).

Wetland sizing

According to the equations and assumptions published by Crites and Tchobanoglous (1998), Fuchs et al. (2011), and Kadlec and Wallace (2009), we sized the wetland for a given nitrogen concentration removal. In the wetland included in scenario 3 (SC3), the daily wastewater flow that we considered was the one produced by the seasonal population. We assumed that the wetland would not deteriorate during the non-seasonal periods, but this is an issue to consider in detailed engineering analyses. However, the seasonal period corresponds to the time with the largest plant uptake (e.g., spring-summer). In SC4, the total wastewater production was considered.

$$V = L \times W \times d$$

Where V is the wetland volume (m^3), L is length (m), W is the width (m), and d is the depth (m)

Assuming that $L = 0.5W \rightarrow W = \sqrt{\frac{A}{0.5}} = \left(\frac{A}{0.5}\right)^{\frac{1}{2}}$

$$d = \frac{A_c}{W} + 0.2 = \frac{\left(\frac{Q}{0.1\ ks}\right)}{\left(\frac{A}{0.5}\right)^{\frac{1}{2}}} + 0.2$$

$$A = \frac{0.0365 \times Q \times 1000}{k_t} \times \ln\left(\frac{C_i - C^*}{C_e - C^*}\right)$$

The general equation generally depends on the wastewater flow:

$$V = Q\left(\frac{\sqrt{0.5}}{0.1\ ks}\sqrt{\frac{36.5\ Q}{k_t} \times \ln\left(\frac{C_i - C^*}{C_e - C^*}\right)} + \frac{7.3}{k_t} \times \ln\left(\frac{C_i - C^*}{C_e - C^*}\right)\right)$$

Where Q is the wastewater flow rate (m^3/day), A_c is the cross-sectional area (m^2), A is the area (m^2), k is the hydraulic conductivity (set at 1000 m/day), s is the slope (set at 0.005 m/m), k_t is the rate constant (set at 1.17 $days^{-1}$), C_i is the influent nitrogen concentration (86.6 mg/L), C^* is the background nitrogen concentration (0.2 mg/L), and C_e is the effluent nitrogen concentration (5 mg/L).

Appendix 6.4 Complete set of environmental results of each scenario

Impact category	Units	SC1	SC2	SC3	SC4	SC5
Global warming	kg CO_2 eq	5.1E+00	3.6E+00	4.4E+00	3.1E+00	2.7E+00
Ozone depletion	kg CFC-11 eq	5.2E-07	3.9E-07	2.7E-07	4.6E-07	5.8E-07
Terrestrial acidification	kg SO_2 eq	1.2E-02	9.7E-03	1.4E-02	1.3E-02	1.2E-02
Freshwater eutrophication	kg P eq	4.1E-04	2.1E-03	3.3E-03	1.5E-03	4.2E-04
Marine eutrophication	kg N eq	1.5E-02	2.6E-02	3.0E-02	1.2E-02	1.7E-02
Human toxicity	kg 1,4-DB eq	1.9E+00	1.4E+00	1.4E+00	1.3E+00	1.7E+00
Photochemical oxidant formation	kg NMVOC	1.2E-02	1.3E-02	1.5E-02	5.4E-02	7.3E-02
Particulate matter formation	kg PM10 eq	6.0E-03	4.8E-03	5.5E-03	5.1E-03	4.5E-03
Terrestrial ecotoxicity	kg 1,4-DB eq	1.7E-03	1.1E-03	4.0E-04	4.3E-04	4.2E-04
Freshwater ecotoxicity	kg 1,4-DB eq	5.3E-02	4.1E-02	5.6E-02	4.4E-02	5.7E-02
Marine ecotoxicity	kg 1,4-DB eq	5.9E-02	4.5E-02	5.3E-02	4.2E-02	5.3E-02
Ionizing radiation	kBq U^{235} eq	2.7E-01	2.1E-01	2.4E-01	3.2E-01	3.3E-01
Agricultural land occupation	m^2a	5.4E-02	6.3E-02	2.1E-01	3.1E-01	5.1E-01
Urban land occupation	m^2a	1.4E-01	9.7E-02	6.5E-02	6.8E-02	6.1E-02
Natural land transformation	m^2	8.8E-04	6.5E-04	1.0E-03	1.5E-03	8.4E-04
Water depletion	m^3	6.4E-03	4.9E-03	1.5E-02	1.3E-02	1.8E-03
Metal depletion	kg Fe eq	2.6E-01	2.0E-01	2.6E-01	1.3E-01	9.0E-02
Fossil depletion	kg oil eq	1.1E+00	8.4E-01	7.4E-01	1.2E+00	1.5E+00
Cumulative energy demand	MJ	5.1E+01	3.9E+01	3.7E+01	5.9E+01	7.1E+01
Single score	mPt	4.4E+02	5.1E+02	6.1E+02	4.6E+02	3.9E+02

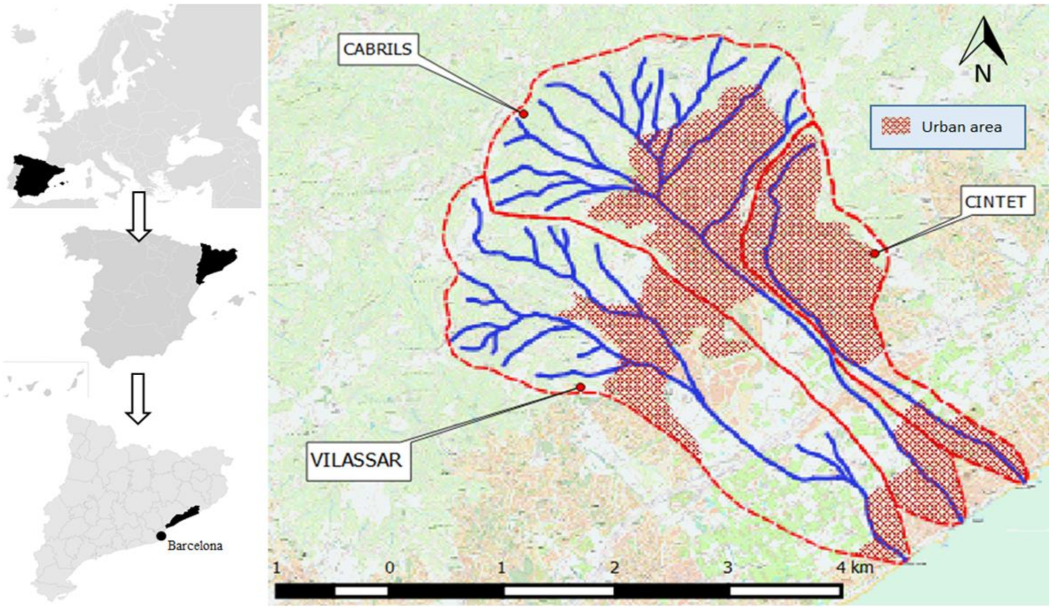
Best Worst

Appendix 6.5 Mean and standard deviation of each scenario and impact indicator obtained through the Monte Carlo simulation. Shaded cells highlight the pairs of scenarios that are not significantly different from each other according to the ANOVA test (95% confidence, p-value < 0.0001)

Impact category	Units	SC1	SC2	SC3	SC4	SC5
Global warming	kg CO_2 eq	5.9E+00 ± 4.5E-01	4.0E+00 ± 3.0E-01	5.8E+00 ± 8.0E-01	4.6E+00 ± 6.4E-01	3.8E+00 ± 1.5E-01
Ozone depletion	kg CFC-11 eq	5.9E-07 ± 4.1E-08	4.0E-07 ± 2.8E-08	5.8E-07 ± 1.5E-07	8.9E-07 ± 2.5E-07	9.1E-07 ± 4.4E-07
Terrestrial acidification	kg SO_2 eq	1.5E-02 ± 1.8E-03	1.2E-02 ± 1.6E-03	2.7E-02 ± 6.6E-03	2.7E-02 ± 6.5E-03	1.7E-02 ± 6.7E-03
Freshwater eutrophication	kg P eq	5.3E-04 ± 7.5E-05	2.9E-03 ± 7.3E-04	4.6E-03 ± 6.9E-04	2.0E-03 ± 2.8E-04	6.0E-04 ± 2.2E-04
Marine eutrophication	kg N eq	1.5E-02 ± 2.0E-04	2.2E-02 ± 2.2E-03	2.8E-02 ± 2.5E-03	1.4E-02 ± 1.6E-03	1.8E-02 ± 1.4E-03
Human toxicity	kg 1,4-DB eq	2.7E+00 ± 4.2E-01	2.1E+00 ± 3.0E-01	2.8E+00 ± 6.3E-01	2.4E+00 ± 7.1E-01	3.0E-00 ± 1.4E-00
Photochemical oxidant formation	kg NMVOC	1.5E-02 ± 2.1E-03	1.3E-02 ± 2.2E-03	3.3E-02 ± 7.6E-03	9.5E-02 ± 3.7E-02	1.2E-01 ± 6.0E-02
Particulate matter formation	kg PM10 eq	7.4E-03 ± 8.7E-04	5.7E-03 ± 6.9E-04	1.2E-02 ± 3.1E-03	1.1E-02 ± 2.9E-03	6.5E-03 ± 2.6E-03
Terrestrial ecotoxicity	kg 1,4-DB eq	1.8E-03 ± 3.8E-05	9.7E-04 ± 2.0E-04	9.5E-04 ± 2.6E-04	9.8E-04 ± 2.5E-04	6.7E-04 ± 3.1E-04
Freshwater ecotoxicity	kg 1,4-DB eq	7.5E-02 ± 1.1E-02	5.9E-02 ± 7.8E-03	1.1E-01 ± 2.1E-02	8.5E-02 ± 2.0E-02	8.8E-02 ± 2.6E-02
Marine ecotoxicity	kg 1,4-DB eq	8.0E-02 ± 1.0E-02	6.0E-02 ± 6.8E-03	1.0E-01 ± 2.1E-02	8.1E-02 ± 1.9E-02	8.3E-02 ± 2.6E-02
Ionizing radiation	kBq U^{235} eq	3.2E-01 ± 2.5E-02	2.2E-01 ± 1.5E-02	5.1E-01 ± 1.2E-01	6.4E-01 ± 1.5E-01	5.0E-01 ± 2.3E-01
Agricultural land occupation	m^2a	6.9E-02 ± 7.7E-03	5.5E-02 ± 6.1E-03	3.8E-01 ± 1.2E-01	5.4E-01 ± 2.1E-01	8.2E-01 ± 4.2E-01
Urban land occupation	m^2a	1.7E-01 ± 1.1E-02	1.0E-01 ± 1.2E-02	1.5E-01 ± 4.0E-02	1.5E-01 ± 3.5E-02	1.0E-01 ± 4.1E-02
Natural land transformation	m^2	8.8E-04 ± 3.6E-05	5.3E-04 ± 8.2E-05	2.6E-03 ± 7.8E-04	3.3E-03 ± 8.2E-04	1.2E-03 ± 7.1E-04
Water depletion	m^3	1.1E-02 ± 2.0E-03	8.6E-03 ± 1.6E-03	3.7E-02 ± 1.0E-02	3.1E-02 ± 9.1E-03	3.0E-03 ± 7.9E-04
Metal depletion	kg Fe eq	4.1E-01 ± 8.5E-02	3.2E-01 ± 6.3E-02	5.4E-01 ± 1.3E-01	2.9E-01 ± 7.0E-02	1.4E-01 ± 6.5E-02
Fossil depletion	kg oil eq	1.3E+00 ± 9.3E-02	8.6E-01 ± 5.9E-02	1.5E+00 ± 3.6E-01	2.3E+00 ± 6.3E-01	2.3E+00 ± 1.1E+00
Cumulative energy demand	MJ	5.9E+01 ± 4.4E+00	4.1E+01 ± 2.8E+00	7.7E+01 ± 1.8E+01	1.1E+02 ± 3.1E+01	1.1E+02 ± 5.2E+01
Single score	mPt	5.2E+02 ± 4.4E+01	6.0E+02 ± 6.9E+01	9.5E+02 ± 1.5E+02	7.8E+02 ± 1.5E+02	6.0E+02 ± 2.7E+02

Appendix 7. Supporting data related to Chapter 10

Appendix 7.1 Location of the ephemeral streams under analysis and urban areas. Vilassar: Stream A, Cabrils: Stream B and Cintet: Stream C.



Source: ICGC. Institut Cartogràfic i Geològic de Catalunya. Cartography and geodesy. http://icgc.cat/.

Appendix 7.2 Parameters and method applied for assessing the hydrologic performance of the streams

Summary variables			**Cabrils (Stream B)**	**Vilassar (Stream A)**	**Cintet (Stream C)**
Length of the main channel (km)			5.82	5.68	2.20
Average slope of the main course (adimensional)			0.075	0.075	0.075
Regional factor I_1/I_d (adimensional)			11.5	11.5	11.5
Watershed area (km²)			5,690	5,569	4,450
Q_P (m³/s)	T=50 years	RC=0.4	78.2	33.5	13.3
		RC=0.6		50.3	19.9
	T=100 years	RC=0.4	87.1	38.2	15.1
		RC=0.6		57.3	22.7
	T=500 years	RC=0.4	108	51.3	20.3
		RC=0.6		77	30.4

T: Return period; RC: Threshold runoff coefficient; Q: Water flow

Maximum daily rainfall according to return period PdT (mm)	
T = 50 years	159
T = 100 years	184
T = 500 years	244

Return period	Maximum draft (m)	Minimum draft (m)	Maximum speed (m/s)	Minimum speed (m/s)	Average speed (m/s)
50	1.7	1.08	6.86	5.09	6.11
100	1.82	1.15	7.12	5.15	6.34
500	2.52	1.32	7.65	3.03	6.62

Source: San Millán, J. 2008. Estudi d'inundabilitat de la Riera de Cabrils, al seu pas pel sector C/Manuel Roca - El Barato. Modificació puntual del Pla General d'Ordenació Urbanística Municipal. Vilassar de Mar (Maresme).

Equations applied to calculate the expected water flow using the Rational Method (Témez variant). Source: Témez, J. 1991. Extended and Improved Rational Method. Version of the Highways Administration of Spain. Proc. XXIV Congress. Madrid, Spain. Vol A.

$$Q_P = K\,\frac{RC \times I \times A}{3.6}$$

$K = 1 + \frac{T_c^{1.25}}{T_c^{1.25} + 14}$	$RC = \frac{(P_d' - P'_o) \times (P_d' + 23 \times P'_o)}{(P'_d + 11 \times P'_o)^2}$ $P'_0 = \beta \times P_0$ $P'_d = P_d \times K_A$ $K_A = 1 \rightarrow A<1$ $K_A = 1 - \frac{\log A}{15} \rightarrow 1 \le A \le 3{,}000$	$I = I_t = I_d \times \left(\frac{I_1}{I_d}\right)^{\frac{28^{0.1} - t^{0.1}}{28^{0.1} - 1}}$ $I_d = \frac{P'_d}{24}$ $t = T_c = 0.3 \times \left(\frac{L}{J^{1/4}}\right)^{0.76}$

Where:

Q_P	Water flow (m3/s)	I_d	Average daily intensity (mm/h)
K	Uniformity coefficient	I_t	Intensity for T_c (mm/h)
RC	Threshold runoff coefficient	I_1/I_d	Torrential factor
I	Rainfall intensity (mm/h)	K_a	Rainfall reduction factor
A	Watershed area (km2)	P'_d	Corrected maximum rainfall (mm/h)
T_c	Concentration time (h)	I'_d	Corrected average daily intensity (mm/h)
P_d	Maximum rainfall in 24 h (mm)	I'_t	Corrected intensity for Tc (mm/h)
P_0	Threshold runoff (mm)	L	Length of the channel (km)
P'_0	Corrected Threshold runoff (mm)	J	Slope
β	Moisture correction factor		

Appendix 7.3 Life cycle inventory of the emergency actions. Breakdown of material, energy and money flows. *The transport of materials is embedded in the price of the material itself

Inventory items	EMERGENCY ACTION - A1				EMERGENCY ACTION – A2				EMERGENCY ACTION – A3			
	Material and energy flows		Monetary flows (2015)		Material and energy flows		Monetary flows (2015)		Material and energy flows		Monetary flows (2015)	
	Quantity	Units	Quantity	Units	Quantity	Units	Quantity	Units	Quantity	Units	Quantity	Units
Workman	-		11,394	€	-		6,849	€	-		237	€
Skilled construction worker	-		31,410	€	-		13,423	€	-		2,578	€
Labor overhead	-		1,433	€	-		659	€	-		102	€
Construction machinery + Diesel	289915	MJ	43,163	€	74,378	MJ	10,104	€	107,453	MJ	16,631	€
Soil from external sources	2054	m^3	19,879	€	1,131	m^3	10,948	€	0	m^3	0	€
Sand	0	kg	0	€	0	kg	0	€	0	kg	0	€
Riprap	1,174,558	kg	15,833	€	0	kg	0	€	1,209,850	kg	17,023	€
Wood formwork (10 uses)	3.5	m^3	1,255	€	1.6	m^3	569	€	0.1	m^3	45	€
Wood strut (3 uses)	3.9	m^3	1,491	€	1.8	m^3	676	€	0.1	m^3	53	€
Wood plank (10 uses)	0.0	m^3	0	€	0.0	m^3	0	€	0.0	m^3	0	€
Concrete (H-100)	0.0	m^3	0	€	0.0	m^3	0	€	0.0	m^3	0	€
Concrete (H-200)	542.6	m^3	32,302	€	180	m^3	10,689	€	13	m^3	750	€
Concrete (H-150)	16.3	m^3	819	€	18	m^3	887	€	0	m^3	0	€
Concrete (H-175)	0.0	m^3	0	€	0	m^3	0	€	0	m^3	0	€
Reinforcing steel	16128	kg	14,638	€	12,325	kg	11,186	€	0	kg	0	€
Metal formwork (150 uses)	0	kg	0	€	0	kg	0	€	0	kg	0	€
Water	0.08	m^3	0.1	€	0	m^3	0	€	0	m^3	0	€
Portland cement	248	kg	26	€	0	kg	0	€	0	kg	0	€
Cement mortar	2.6	m^3	322	€	0	m^3	0	€	0	m^3	0	€
Gray panot slabs	5000	kg	494	€	0	kg	0	€	0	kg	0	€
Concrete slabs	1.9	m^3	186	€	0	m^3	0	€	0	m^3	0	€
Asphalt	50,400	kg	2,318	€	0	kg	0	€	0	kg	0	€
Internal transport	0	tkm	0	€	0	tkm	0	€	0	tkm	0	€
Transport to construction site*	178,659	tkm	-	€	70,270	tkm	-	€	37,207	tkm	-	€
Waste to landfill	2,218	m^3	15276	€	156	m^3	1,073	€	351	m^3	2,421	€
Transport to landfill	36,594	tkm	8775	€	3,142	tkm	616	€	5,675	tkm	1,391	€

Inventory items	EMERGENCY ACTION – A4				EMERGENCY ACTION – A5				EMERGENCY ACTION – A6			
	Material and energy flows		Monetary flows (2015)		Material and energy flows		Monetary flows (2015)		Material and energy flows		Monetary flows (2015)	
	Quantity	Units	Quantity	Units	Quantity	Units	Quantity	Units	Quantity	Units	Quantity	Units
Workman	-		58,843	€	-		13,570	€	-		27,048	€
Skilled construction worker	-		165,678	€	-		6,220	€	-		53,296	€
Labor overhead	-		4,421	€	-		343	€	-		8,542	€
Construction machinery + Diesel	3,111,081	MJ	245,491	€	201,679	MJ	22,992	€	2,077,914	MJ	272,505	€
Soil from external sources	0	m³	0	€	0	m³	0	€	0	m³	0	€
Sand	0	kg	0	€	0	kg	0	€	0	kg	0	€
Riprap	2,880,000	kg	40,522	€	156,000	kg	2,195	€	8,797,360	kg	123,779	€
Wood formwork (10 uses)	2.8	m³	1,000	€	0.3	m³	96	€	4.0	m³	1,413	€
Wood strut (3 uses)	3.1	m³	1,189	€	0.3	m³	114	€	4.4	m³	1,680	€
Wood plank (10 uses)	1.0	m³	254	€	0.0	m³	0	€	0.0	m³	0	€
Concrete (H-100)	218.0	m³	14,367	€	0.0	m³	0	€	0.0	m³	0	€
Concrete (H-200)	1,494	m³	110,071	€	62	m³	3,520	€	0	m³	0	€
Concrete (H-150)	0	m³	0	€	0	m³	0	€	39	m³	3,073	€
Concrete (H-175)	0	m³	0	€	714	m³	67,615	€	908	m³	86,141	€
Reinforcing steel	147,813	kg	134,154	€	0	kg	0	€	41,025	kg	37,234	€
Metal formwork (150 uses)	3	kg	154	€	0	kg	0	€	0	kg	0	€
Water	0	m³	0	€	0	m³	0	€	0	m³	0	€
Portland cement	0	kg	0	€	0	kg	0	€	0	kg	0	€
Cement mortar	0	m³	0	€	1	m³	72	€	0	m³	0	€
Gray panot slabs	0	kg	0	€	0	kg	0	€	0	kg	0	€
Concrete slabs	0	m³	0	€	36	m³	3,594	€	0	m³	0	€
Asphalt	0	kg	0	€	0	kg	0	€	0	kg	0	€
Internal transport	0	tkm	0	€	0	tkm	0	€	137,844	tkm	34,357	€
Transport to construction site*	225,095	tkm	-	€	63,136	tkm	-	€	336,290	tkm	-	€
Waste to landfill	96	m³	661	€	399	m³	2,746	€	110	m³	757	€
Transport to landfill	2,304	tkm	380	€	7,335	tkm	1,577	€	2,638	tkm	435	€

Inventory items	EMERGENCY ACTION – A7				TOTAL			
	Material and energy flows		Monetary flows (2015)		Material and energy flows		Monetary flows (2015)	
	Quantity	Units	Quantity	Units	Quantity	Units	Quantity	Units
Workman	-		34,859	€	-		152,800	€
Skilled construction worker	-		39,658	€	-		312,264	€
Labor overhead	-		1,918	€	-		17,418	€
Construction machinery + Diesel	2,718,462	MJ	147,799	€	8,580,881	MJ	758,685	€
Soil from external sources	0	m^3	0	€	3,185	m^3	30,827	€
Sand	29,835	kg	526	€	29,835	kg	526	€
Riprap	3,741,500	kg	52,643	€	17,959,268	kg	251,994	€
Wood formwork (10 uses)	2.7	m^3	966	€	15	m^3	5,344	€
Wood strut (3 uses)	3.0	m^3	1,148	€	17	m^3	6,350	€
Wood plank (10 uses)	0.0	m^3	0	€	1	m^3	254	€
Concrete (H-100)	138.8	m^3	9,144	€	357	m^3	23,511	€
Concrete (H-200)	411	m^3	30,298	€	2,702	m^3	187,630	€
Concrete (H-150)	0	m^3	0	€	72	m^3	4,779	€
Concrete (H-175)	0	m^3	0	€	1,621	m^3	153,757	€
Reinforcing steel	17,482	kg	15,866	€	234,773	kg	213,078	€
Metal formwork (150 uses)	0	kg	0	€	3	kg	154	€
Water	938	m^3	1,530	€	938	m^3	1,530	€
Portland cement	0	kg	0	€	248	kg	26	€
Cement mortar	0	m^3	0	€	4	m^3	393	€
Gray panot slabs	0	kg	0	€	5,000	kg	494	€
Concrete slabs	0	m^3	0	€	38	m^3	3,780	€
Asphalt	0	kg	0	€	50,400	kg	2,318	€
Internal transport	931,005	tkm	100,958	€	1,068,848	tkm	135,315	€
Transport to construction site*	154,582	tkm	-	€	1,065,239	tkm	-	€
Waste to landfill	86	m^3	593	€	3,415	m^3	23,527	€
Transport to landfill	2,064	tkm	340	€	59,751	tkm	13,515	€

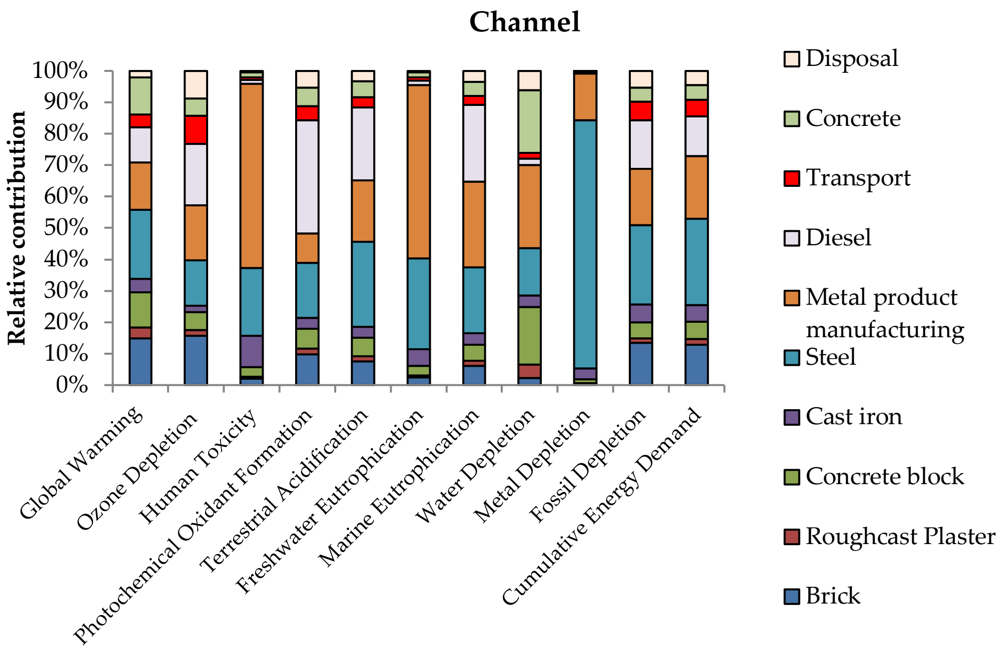
Appendix 7.4 Environmental, economic and hydrologic data for flooding events with damage (1997-2014)

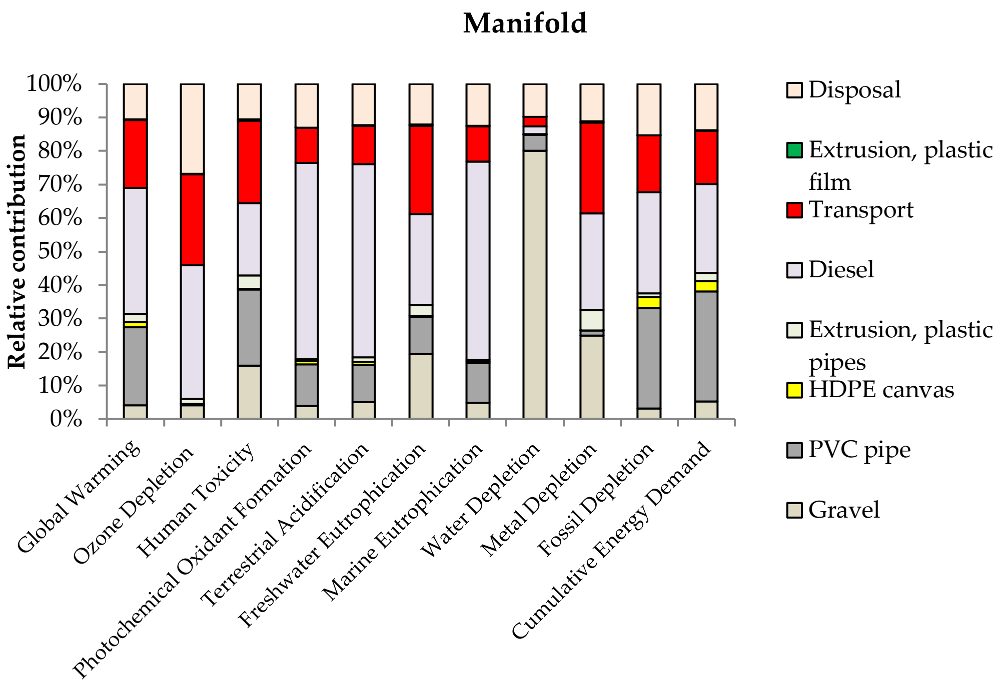
Event	Economic loss (€)	Environmental loss (kg CO_2 eq)	Maximum rainfall intensity (mm/h)	Accumulated rainfall (mm)	Estimated water flow (m^3/s)	Do-nothing scenario	
						Estimated economic loss (€)	Estimated environmental loss (kg CO_2 eq)
2-Sep-96	***2,540,410***	***261,838***	***176.4***	***158.3***	***42.7***		
25-Jun-97	522	64	nd	nd	nd	nd	nd
8-Feb-98	1,195	147	nd	nd	nd	nd	nd
7-Jul-98	1,153	142	66.0	21.9	5.9	351,242	36,202
3-Dec-98	4,372	518	55.2	116.4	31.4	1,866,516	192,381
6-Sep-99	56,779	8,034	3.6	1.4	0.4	22,946	2,365
14-Sep-99	128,392	13,010	132.0	93.9	25.3	1,506,226	155,246
19-Sep-99	7,078	869	19.2	22.4	6.0	359,173	37,020
4-Oct-99	5,904	657	nd	nd	nd	nd	nd
16-Apr-00	4,022	58	nd	nd	nd	nd	nd
10-Jun-00	3,565	438	86.4	71.3	19.2	1,143,549	117,865
5-Aug-00	19,460	2,172	4.8	2.1	0.6	34,728	3,579
19-Sep-00	55,394	6,015	175.2	62.0	16.7	994,474	102,500
4-Oct-00	812	100	nd	nd	nd	nd	nd
13-Oct-00	1,776	26	nd	nd	nd	nd	nd
31-Jul-02	6,899	653	56.4	65.7	17.7	1,053,804	108,615
1-Aug-02	3,942	484	31.2	23.7	6.4	380,002	39,167
23-Sep-02	7,774	390	66.0	18.1	4.9	290,334	29,925
24-Sep-02	4,776	69	109.2	32.71	8.8	524,765	54,087
24-Nov-02	10,164	147	30.0	26.2	7.1	420,120	43,302
27-Feb-03	23,907	2,935	18.0	14.2	3.8	227,693	23,468
31-Mar-03	8,529	124	nd	nd	nd	nd	nd
15-Aug-03	1,671	205	nd	nd	nd	nd	nd
17-Aug-03	18,551	1,944	102.0	30.3	8.2	486,033	50,095
7-Sep-03	15,253	1,725	129.6	47.9	12.9	768,398	79,198
17-Sep-03	6,556	95	nd	nd	nd	nd	nd
27-Sep-03	3,370	49	50.4	37.7	10.2	604,770	62,333
17-Oct-03	1,705	25	57.6	32.9	8.9	527,499	54,369

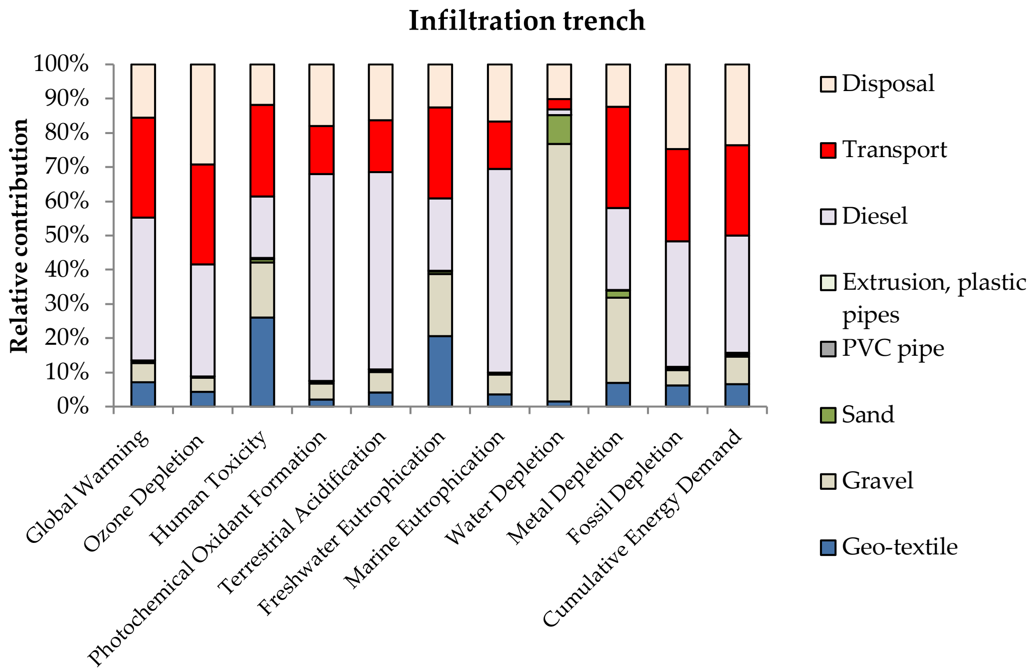
Event	Economic loss (€)	Environmental loss (kg CO_2 eq)	Maximum rainfall intensity (mm/h)	Accumulated rainfall (mm)	Estimated water flow (m^3/s)	Do-nothing scenario	
						Estimated economic loss (€)	Estimated environmental loss (kg CO_2 eq)
23-Nov-03	3,476	50	12.0	4.9	1.3	78,080	8,048
11-Jul-04	13,863	201	26.4	19.1	5.1	306,389	31,579
31-Oct-04	666	10	61.2	18.08	19.3	1,145,397	118,055
12-Oct-05	5,452	669	27.6	18.24	25.7	1,530,404	157,738
14-Oct-05	9,805	1,191	38.4	20.5	5.5	328,873	33,897
12-Aug-06	2,736	336	6.0	2.0	0.5	32,071	3,306
15-Aug-06	21,009	1,763	115.2	28.5	7.7	457,196	47,123
24-Aug-06	983	109	12.0	5.0	nd	nd	nd
12-Sep-06	2,449	122	28.8	15.9	4.3	255,067	26,290
13-Sep-06	18,378	1,429	48.0	40.7	11.0	652,896	67,294
12-Aug-07	82,584	7,752	nd	56.9	15.3	912,788	94,081
10-May-08	5,022	156	nd	60.5	16.3	970,540	100,033
14-Aug-10	959	14	nd	32.2	8.7	516,552	53,241
17-Sep-10	2,349	261	nd	48.5	13.1	778,036	80,192
5-Nov-11	4,211	517	nd	6.0	1.6	96,252	9,921
18-Jul-13	3,326	408	nd	24.6	6.6	394,633	40,675
27-Aug-13	18,188	2,017	nd	41.4	11.2	664,138	68,452
8-Sep-13	1,354	20	nd	14.5	3.9	232,609	23,975
15-Jun-14	6,623	191	nd	19.0	5.1	304,798	31,415
28-Jul-14	349	5	nd	18.7	5.0	299,985	30,919
29-Jul-14	65,081	7,694	nd	24.9	6.7	399,445	41,171
22-Aug-14	4,947	175	nd	23.4	6.3	375,382	38,690

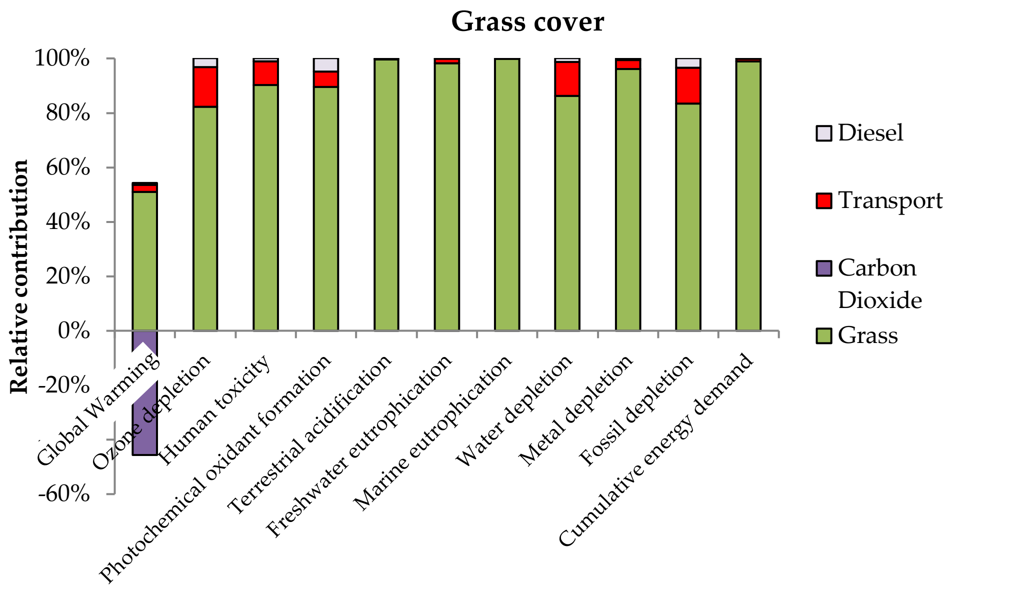
Appendix 8. Supporting data related to Chapter 11

Appendix 8.1 Environmental impacts of the different components of the FST under analysis



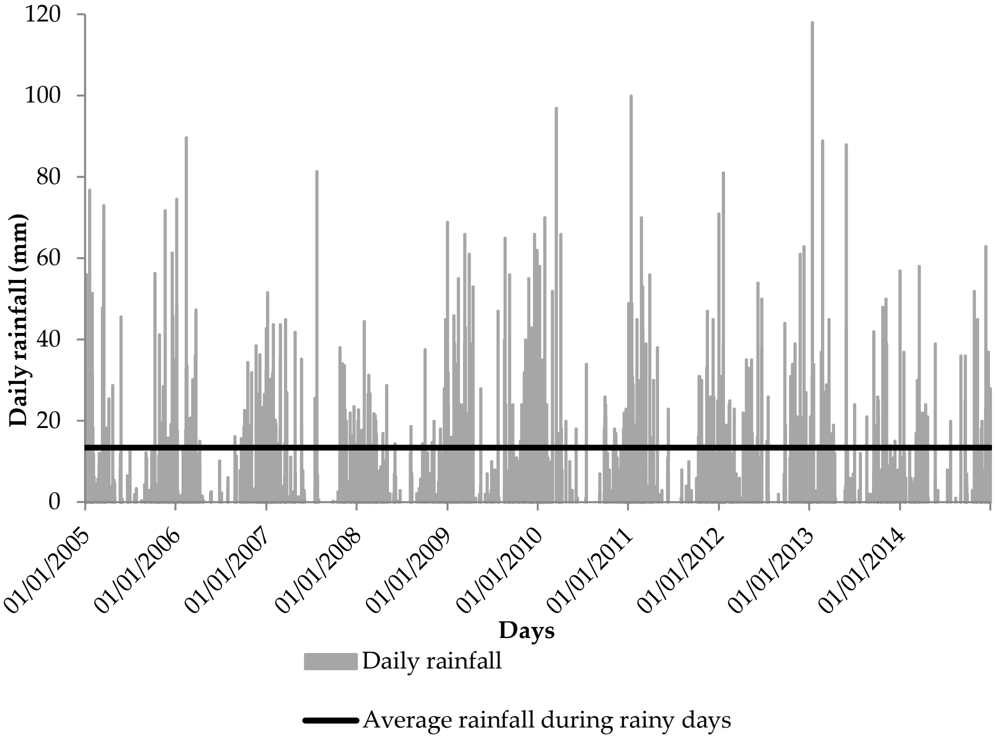



Infiltration trench
Relative contribution
100%
90%
80%
70%
60%
50%
40%
30%
20%
10%
0%
Global Warming
Ozone Depletion
Human Toxicity
Photochemical Oxidant Formation
Terrestrial Acidification
Freshwater Eutrophication
Marine Eutrophication
Water Depletion
Metal Depletion
Fossil Depletion
Cumulative Energy Demand
Disposal
Transport
Diesel
Extrusion, plastic pipes
PVC pipe
Sand
Gravel
Geo-textile

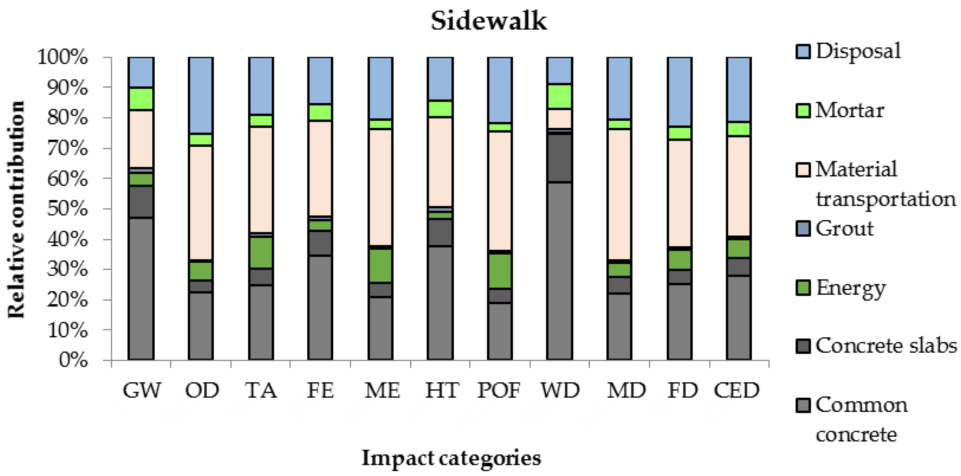

Grass cover
Relative contribution
100%
80%
60%
40%
20%
0%
-20%
-60%
Global Warming
Ozone depletion
Human toxicity
Photochemical oxidant formation
Terrestrial acidification
Freshwater eutrophication
Marine eutrophication
Water depletion
Metal depletion
Fossil depletion
Cumulative energy demand
Diesel
Transport
Carbon Dioxide
Grass

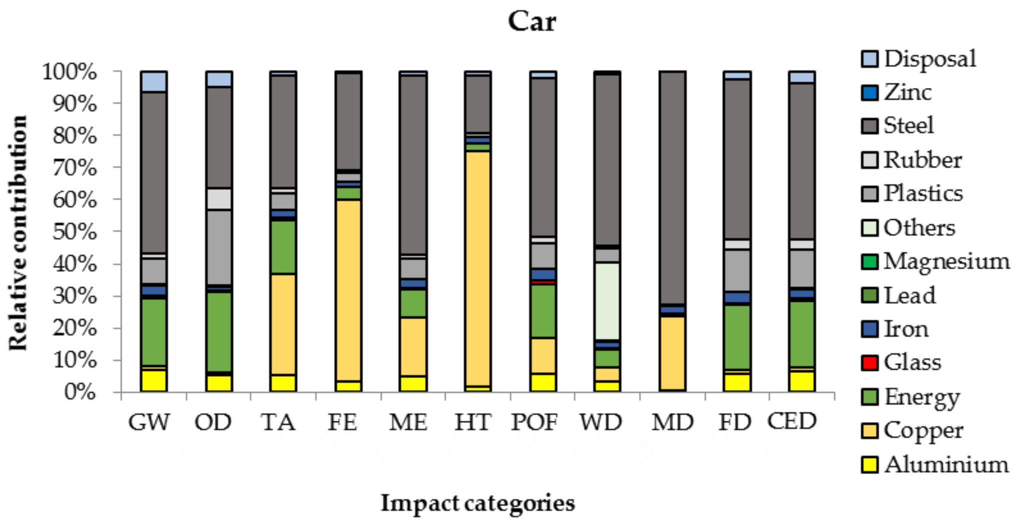
Appendix 9. Supporting data related to Chapter 12

Appendix 9.1 Evolution of the daily rainfall events in São Carlos from 2005-2014. Data from INMET (2014)



Appendix 9.2 Relative contribution of the processes considered in the life cycle of one car and one m² of concrete sidewalk.




Car
Relative contribution
100%
90%
80%
70%
60%
50%
40%
30%
20%
10%
0%
GW
OD
TA
FE
ME
HT
POF
WD
MD
FD
CED
Impact categories
Disposal
Zinc
Steel
Rubber
Plastics
Others
Magnesium
Lead
Iron
Glass
Energy
Copper
Aluminium