DECISION-SUPPORT FOR ADAPTIVE AND SUSTAINABLE URBAN WASTEWATER SYSTEM MANAGEMENT IN THE FACE OF UNCERTAINTY

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DECRAREM:

Que aquest treball titulat "Decision-support for adaptive and sustainable urban wastewater system management in the face of uncertainty", que presenta la llicenciada en Matemàtiques Antonia Hadjimichael, per a l'obtenció del títol de doctora, ha estat realitzat sota la nostra direcció i que compleix els requeriments per poder optar a Menció Europea.

I per a que en prengueu coneixement i tingui els efectes que corresponguin, presentem davant la Facultat de Ciències de la Universitat de Girona l'esmentada Tesi, signant aquesta certificació.

Dr. Joaquim Comas Matas   Dr Lluis Corominas Tabares

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"The true meaning of life, Wesley, is to plant trees

under whose shade you do not expect to sit."

- Nelson Henderson, second-generation farmer, as quoted by his son Wesley Henderson in *Under Whose Shade* (1982)

To my parents,

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List of Acronyms and Symbols

ACA: Catalan Water Agency

ASM: Activated Sludge Model

CBA: Cost-Benefit Analysis

CC: Climate Change (LCA impact category)

CH_4: Methane

CDCB: Consorci per la Defensa de la Conca del Besòs

CO_2: Carbon dioxide

COD: Chemical Oxygen Demand

CSO: Combined sewer overflow

CVM: Contingent Valuation Method

DB: Dichlorobenzene

DO: Dissolved oxygen

DST: Decision-support tool

FE: Freshwater Eutrophication (LCA impact category)

FET: Freshwater Ecotoxicity (LCA impact category)

GHG: Greenhouse gas

HICP: Harmonised Index of Consumer Prices

HT: Human Toxicity (LCA impact category)

IPCC: Intergovernmental Panel on Climate Change

LCA: Life Cycle Analysis

ME: Marine Eutrophication (LCA impact category)

N: Nitrogen

N_2O: Nitrous oxide

NH_4^+: Ammonium

NO_3^-: Nitrate

NPV: Net Present Value

P: Phosphorus

PE: Population equivalent

PI: Proportional-Integral

PDF: Probability density function

RBMP: River Basin Management Plan

RCP: Representative concentration pathway

SO_2: Sulphur dioxide

TA: Terrestrial Acidification (LCA impact category)

TN: Total Nitrogen

TSS: Total Suspended Solids

UCT: University Cape Town

UWS: Urban wastewater system

VODE: Variable-coefficient Ordinary Differential Equation

WdD: Waterschap de Dommel

WFD: Water Framework Directive

WWTP: Wastewater treatment plan

Summary

With sustainable development as the new overarching goal, urban wastewater system (UWS) managers are now being asked to take all social, economic, technical and environmental facets related to their decisions into account. In this complex decision-making environment, uncertainty can be formidable. Uncertainty is present both in the ways the system is interpreted stochastically, but also in its natural ever-shifting behaviour. This inherent uncertainty leads to the conclusion that better decisions will be made if the decision-making process is adaptive and iterative. UWS decision-support frameworks exist in the literature, but none of them effectively addresses all these needs. Hence, there is a need for an adaptive framework that supports UWS management by addressing aspects of sustainability and uncertainty of various types. The development of such a framework is the main outcome of this work, and is supported by two demonstrative applications presented in this thesis.

The thesis describes a conceptual framework that can be used to assess environmental and socio-economic impacts of UWS management options under various conditions, both present and future. This is achieved by establishing an adaptive management framework for decision-support that evaluates and compares alternative solutions. Socio-economic aspects such as externalities are taken into account, along with other traditional criteria as necessary. Robustness, reliability and resilience measures are used to evaluate the performance of the system given inputs representing future and present conditions. Also included is a valuation uncertainty analysis that incorporates uncertain valuation assumptions in the decision-making process.

As part of this work, the framework was applied to the Congost UWS in Catalonia, Spain to demonstrate its applicability to a real-world challenge. The Congost UWS represents a typical problem often faced by managers: poor river water quality, an increasing population and more stringent water quality legislation. The application of the framework made use of: i) a Cost Benefit Analysis (CBA) including monetised environmental benefits and damages; ii) a Robustness Analysis of system performance against potential future conditions; iii) Reliability and Resilience Analyses of the system given contextual variability; and iv) a Valuation Uncertainty Analysis of model parameters. Several reactor volume expansions were evaluated following the framework, with the evaluations making use of electricity price

projections, population growth projections and climate change projections. The results of the framework application suggest that the installation of larger volumes would give rise to increased net present values (NPVs) despite larger capital costs. The results also indicate that larger volumes would exhibit increased robustness and resilience. The results were highly dependent on the population estimates, as they appeared to affect the estimated NPVs the most, followed by electricity prices and climate change projections.

An alternative application of the sustainability assessment part of the framework is also presented. This application evaluates the sustainability aspects of four options proposed for upgrading of the UWS of Eindhoven and the Dommel River in the Netherlands, against the base-case “do-nothing” option. The options aim to reduce the overall environmental impact of the Eindhoven UWS by targeting river dissolved oxygen (DO) depletion and ammonia peaks, reducing combined sewer overflows (CSOs) and enhancing nutrient removal. The options were evaluated using Life Cycle Analysis (LCA) with the receiving river included in the boundaries. An integrated model of the UWS has proved to be a powerful tool to analyse and evaluate the proposed measures and is employed in this study. An uncertainty analysis of the estimated impacts was performed to support the outcomes. The study also used the economic concept of shadow prices to assign relative weights of socio-economic importance to the estimated life cycle impacts. This novel integration of tools complements the assessments of this UWS with the inclusion of long-term global environmental impacts and the investigation of trade-offs between different environmental impacts through a single monetary unit.

The objectives of this work have been fulfilled within this thesis. The presented framework is expected to be a valuable tool for the next generation of water decision-making and, the two applications demonstrate novel integrations of metrics and methods valuable for UWS analysis and future work.

RESUM

En el marc de desenvolupament sostenible, els gestors dels sistemes de sanejament han d'incloure factors socials, econòmics, tècnics i ambientals durant la presa de decisions. Donada l'elevada complexitat en la presa de decisions, la incertesa relacionada amb aquests factors pot esdevenir rellevant i per tant s'ha d'incorporar de forma adequada. La incertesa està present no només en la forma d'interpretar el sistema sinó també en el seu comportament natural i canviant. Necessitem processos de decisió que siguin adaptatiu i iteratiu per tal d'assegurar que el sistemes de sanejament mantenen la seva eficàcia durant la seva vida útil. Tot i que existeixen en la bibliografia alguns marcs d'ajuda a la presa de decisions cap d'ells inclou de forma efectiva tots aquests aspectes. Per tant, necessitem disposar d'un marc que ajudi en la gestió dels sistemes de sanejament sota el paradigma de gestió adaptativa al mateix temps que consideri aspectes de sostenibilitat i incertesa de diferent origen. Aquest és l'objectiu principal d'aquest treball, conjuntament amb dos aplicacions de demostració.

La tesis descriu aquest marc conceptual que pot ser utilitzat per avaluar aspectes ambientals i socio-econòmics de la gestió de sistemes de sanejament sotmesos a diferents reptes presents i futurs. Això s'aconsegueix mitjançant la definició d'un marc per l'ajuda a la presa de decisions de gestió adaptativa que avaluï i compari diferents alternatives. En aquest marc, els aspectes socio-econòmics juntament amb les externalitats es tenen en compte, al mateix temps que altres criteris tradicionals. A més els anàlisis de robustesa, fiabilitat i resiliència posen a prova l'eficàcia del sistema davant la variabilitat present i futura. L'ànàlisis d'incertesa incorpora diferents fonts d'incertesa en el procés de presa de decisions (per exemple, la variabilitat deguda al canvi climàtic o la incertesa dels paràmetres d'un model).

Aquest marc conceptual s'aplica al sistema de sanejament de la conca del riu Congost a Catalunya, Espanya, per demostrar la seva aplicabilitat en un repte real que han d'afrontar els gestors dels sistemes de sanejament: una millora dels rendiments de depuració, un increment de la població i/o una legislació més estricta. L'aplicació s'ha basat en: i) un anàlisis que inclou els beneficis i els danys ambientals monetitzats; ii) un anàlisis de robustesa de l'eficàcia del sistema en front a canvis futurs; iii) un anàlisis de fiabilitat i resiliència del sistema donada la variabilitat contextual; i iv) un anàlisis d'incertesa dels paràmetres del model utilitzat. Es van avaluar diferents ampliacions del volum reactor, i

utilitzant projeccions en el futur del preu de l'electricitat, del creixement de la població i dels escenaris de canvi climàtic. Els resultats il·lustren que l'ampliació de la depuradora amb volums del reactor més grans coincideix amb un augment del valor actual net (VAN), així com amb una major robustesa i resiliència del sistema de sanejament. L'increment poblacional és el factor que té major influència en el VAN, seguit dels preus de l'electricitat i dels escenaris de canvi climàtic.

Posteriorment es presenta una aplicació alternativa de la part de sostenibilitat dins el marc conceptual proposat. S'avaluen els aspectes de sostenibilitat per quatre mesures proposades per la renovació del sistema de sanejament d'Eindhoven i del riu Dommel a Holanda, en comparació amb el cas de referència "no fer res". Les mesures tenen per objectiu reduir l'impacte ambiental del sistema de sanejament, mitjançant la disminució del dèficit d'oxigen i dels pics d'amoni al riu, la reducció de descàrregues del sistema unitari de col·lectors i millorar l'eliminació de nutrients. Les mesures s'avaluen utilitzant la tècnica d'Anàlisis de Cicle de Vida, incloent el riu dins els límits del sistema. Es va demostrar que un model integrat del sistema de sanejament és una eina potent per analitzar i avaluar les mesures proposades. També es va realitzar un anàlisis d'incertesa dels impactes estimats per donar més rellevància als resultats. L'estudi també aplica el concepte econòmic de preus ombra per assignar pesos relatius a aspectes socio-econòmics a les categories d'impacte ambiental estudiades. Aquesta integració innovadora d'eines complementa l'assessorament del sistema de sanejament ja que incorpora aspectes ambientals a llarg termini i la investigació dels *trade-offs* entre els diferents impactes ambientals, utilitzant una única unitat monetària.

Els objectius d'aquesta tesis s'han assolit. El marc conceptual d'ajuda a la decisió desenvolupat pot ser una eina molt valuosa per la pròxima generació de gestors de sistemes de sanejament. A més, les dues aplicacions del marc conceptual en sistemes reals han demostrat integracions innovadores de mètriques i mètodes valuosos per l'anàlisis de sistema de sanejament.

RESUMEN

Con el objetivo de desarrollo sostenible los gestores de los sistemas de saneamiento tienen que considerar factores sociales, económicos, técnicos y ambientales durante la toma de decisiones. Dada la complejidad en la toma de decisiones, la incertidumbre relacionada con estos factores puede ser importante y por lo tanto se tiene que abordar de forma adecuada. La incertidumbre está presente no solamente en la forma de interpretar el sistema de saneamiento sino también en su comportamiento natural variable. Los procesos de decisión debe ser adaptativos e iterativos para asegurar que el sistema mantiene su eficacia durante su vida útil. Aunque existen en la bibliografía algunos marcos de toma de decisión ninguno de ellos aborda de forma efectiva todos estos aspectos. Por lo tanto, es necesario disponer de un marco que ayude en la gestión de los sistemas de saneamiento bajo el paradigma de gestión adaptativa al mismo tiempo que se consideren aspectos de sostenibilidad e incertidumbre de distinta índole. Este es el objetivo principal de este trabajo, juntamente con dos aplicaciones de demostración.

La tesis describe dicho marco conceptual que puede ser utilizado para evaluar aspectos ambientales y socio-económicos de la gestión de sistemas de saneamiento bajo distintos retos presentes y futuros. Esto se consigue mediante la definición de un marco de toma de decisones de gestión adaptativa que evalúa y compara distintas soluciones. Los aspectos socio-económicos juntamente con externalidades se tienen en cuenta, juntamente con otros criterios tradicionales. Además, los análisis de robustez, fiabilidad y resiliencia ponen a prueba la eficacia del sistema ante la variabilidad presente y futura. El análisis de incertidumbre incorpora distintas fuentes de incertidumbre en el proceso de toma de decisiones (por ejemplo, la variabilidad debida al cambio climático o la incertidumbre de los parámetros de un modelo).

Este marco conceptual se aplica al sistema de saneamiento de la cuenca del río Congost en Cataluña, España, para demostrar su aplicabilidad en un reto real que deben afrontar los gestores de los sistemas de saneamiento: una mejora de los rendimientos de depuración, un incremento de la población y/o una legislación más exigente. La aplicación se basó en: i) un análisis que incluye los beneficios y los daños ambientales monetizados; ii) un análisis de robustez de la eficacia del sistema ante cambios futuros; iii) un análisis de fiabilidad y resiliencia del sistema dada la variabilidad contextual; y iv) un análisis de incertidumbre de los parámetros del modelo. Se evaluaron distintas ampliaciones del volumen del reactor,

utilizando proyecciones en el futuro para los precios de la electricidad, del crecimiento de la población y del cambio climático. Los resultados ilustran que una ampliación de la planta depuradora con volúmenes del reactor de mayor capacidad coincide con un aumento del valor actual neto (VAN), así como con una mayor robustez y resiliencia del sistema. El incremento poblacional es el factor con mayor influencia en el VAN, seguido de los precios de la electricidad y las proyecciones de cambio climático.

A continuación se presenta una aplicación alternativa de la evaluación de sostenibilidad dentro del marco conceptual propuesto. Se evalúan los aspectos de sostenibilidad para cuatro medidas propuestas para la renovación del sistema de saneamiento de Eindhoven y del río Dommel en Holanda, en comparación con el caso de referencia "no hacer nada". Las medidas tienen por objetivo reducir el impacto ambiental del sistema de saneamiento de Eindhoven mediante la disminución del déficit de oxígeno y los picos de amonio en el río, la reducción de vertidos de los sistemas unitarios de saneamiento y mejorando la eliminación de nutrientes. Las medidas se evalúan utilizando la técnica de Análisis de Ciclo de Vida e incluyendo el río en los límites del sistema. Se demostró que un modelo integrado del sistema de saneamiento es una herramienta potente para analizar y evaluar las medidas propuestas. También se realizó un análisis de incertidumbre de los impactos estimados para dar más entidad a los resultados. El estudio también utiliza el concepto económico de precios sombra para asignar pesos relativos a aspectos socio-económicos a las categorías de impacto ambiental estudiadas. Esta integración novedosa de herramientas complementa la evaluación de este sistema de saneamiento ya que incorpora impactos ambientales a largo plazo y la investigación de *trade-offs* entre los distintos impactos ambientales, utilizando una única unidad monetaria.

Los objetivos de esta tesis se han cumplido. El marco conceptual de ayuda a la decisión desarrollado puede ser una herramienta muy valiosa para la próxima generación de gestores de sistemas de saneamiento. Además, las dos aplicaciones el marco conceptual en sistemas reales han demostrado integraciones novedosas de métricas y métodos valiosos para el análisis de los sistemas de saneamiento.

LIST OF PUBLICATIONS

Journal publications related to this doctoral thesis:

Hadjimichael, A., Morera, S., Benedetti, L., Flameling, T., Corominas, L., Weijers, S., Comas, J., 2016. Assessing urban wastewater system upgrades using integrated modeling, life cycle analysis and shadow pricing. Environ. Sci. Technol. doi:10.1021/acs.est.5b05845

Hadjimichael, A., Corominas, Ll., Comas, J.. Do artificial intelligence methods enhance the potential of decision support systems? A review for the urban water sector. AI Communications [Special Issue] (Accepted for publication)

Hadjimichael, A.; Corominas, Ll. ; Belia, E. ; Comas, J. ; *Adaptive management for sustainable decisions in UWSs in the face of uncertainty – Part 1: The framework*. (Under review)

Hadjimichael, A.; Corominas, Ll. ; Belia, E. ; Comas, J. ; *Adaptive management for sustainable decisions in UWSs in the face of uncertainty – Part 2: Application on UWS upgrading.* (Under review)

1. INTRODUCTION

Redrafted from:

Hadjimichael, A., Corominas, Ll., Comas, J.

Do artificial intelligence methods enhance the potential of decision support systems? A review for the urban water sector.

AI Communications [Special Issue] (Accepted for publication)

1. INTRODUCTION

1.1. THE URBAN WASTEWATER SYSTEM (UWS)

The urban water cycle constitutes part of the natural water cycle (Figure 1-1). Water is extracted from natural or artificial reservoirs (surface water, groundwater, etc.) and is treated for use and consumption. It is then transported to an urban agglomeration - an urban area where population and/or economic activities are sufficiently concentrated for water to be conducted to and collected from. After its various uses (municipal, industrial, agricultural), it is collected again through a system of conduits (sewer system) responsible for the transportation of wastewater to a treatment facility. At the wastewater treatment plant (WWTP) it is treated and then either reused or discharged back to the environment, most commonly a receiving water body. The urban wastewater system (UWS) typically comprises four main components: i) the agglomeration, ii) the sewer system, iii) the WWTP, and iv) the receiving water body.

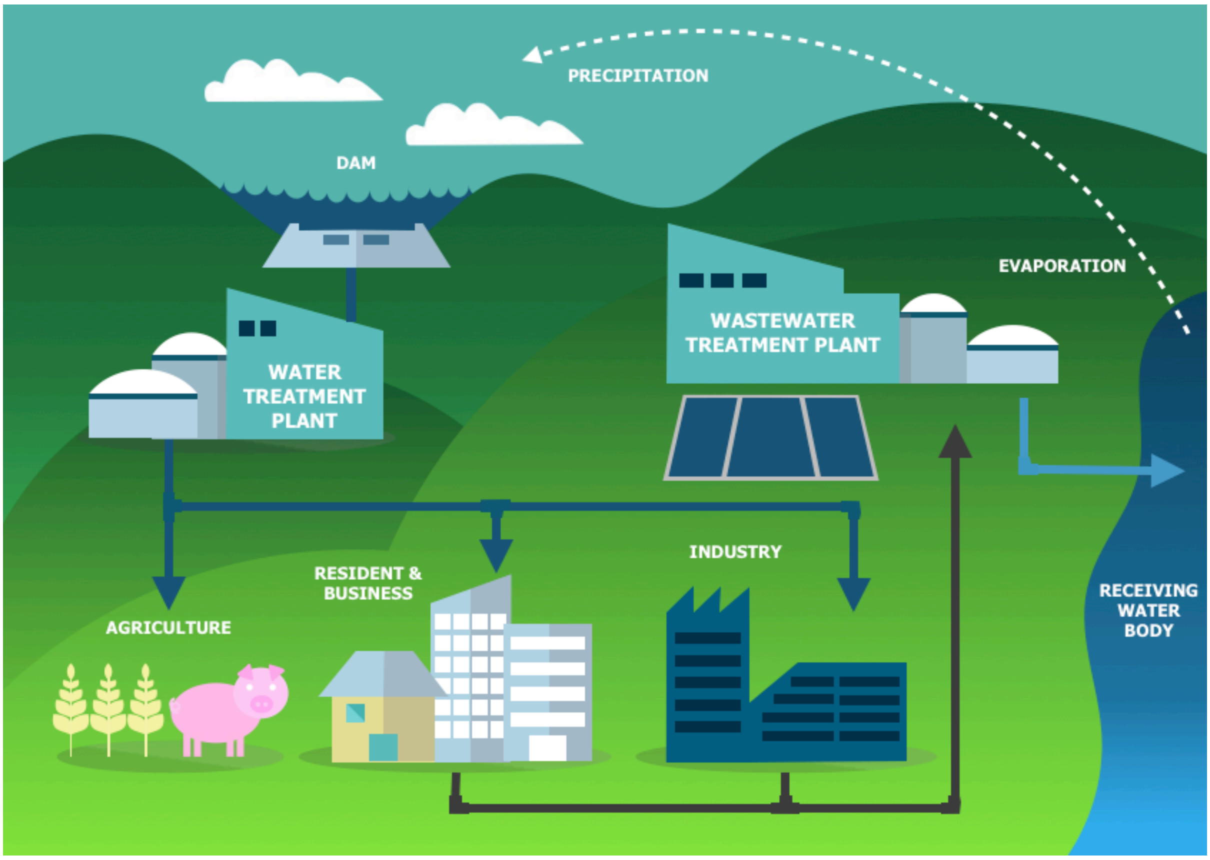


Figure 1-1 - The urban water cycle and the urban wastewater system (UWS)

As Sedlak (2014) and van Loosdrecht and Brdjanovic (2014) explain and Rauch and Kleidorfer (2014) and Neumann *et al.* (2015) exemplify, the primary concerns of UWSs have evolved through the years: from being primarily about sanitation and hygiene (early 20th century) to focussing on water pollution and the removal of organic matter and nutrients (mid - late 20th century) to the removal of emerging pollutants and other concerns (late 20th - present). As we advance our understanding of the complicated interconnected relationships between society and environment the goals of water and wastewater management also become more complex and multifaceted.

There is great eco-toxicological concern regarding the presence of various emerging pollutants, such as nanomaterials and persistent chemical compounds found at trace concentrations (namely micro-pollutants) in our water bodies. WWTPs represent one of the main pathways for their discharge, yet municipal WWTPs are generally not equipped for their removal (Metz and Ingold, 2014). In effluent-dominant rivers with low dilution capacity (often the case with Mediterranean river basins) the problem is magnified (Comoretto and Chiron, 2005). Excessive nutrient discharge is not only a main cause of eutrophication but also a waste of resources; especially with regards to phosphorus, as phosphate rock is a limited and critical raw material (Cornel and Schaum, 2009). Nutrient recovery is now more feasible with the development of new technologies that allow for valuable products (such as fertilisers) to be generated from wastewater. Wastewater is also a source of organic matter, which can be used for the production of biogas, a potent renewable energy source. Efficient recovery of biogas is now possible for supplying energy to the WWTP or other uses (Daigger, 2007). These opportunities are increasingly more important in the face of increased energy dependency on energy imports and scarce energy resources in Europe and beyond. Energy efficiency is a progressively important topic, including in the management of the UWS, in the process of reducing primary energy consumption and greenhouse gas (GHG) emissions. Water resource efficiency also comes in question, especially in the Mediterranean region facing dire climate change impacts. Water scarcity is reported in nearly all Mediterranean river basin districts (Angelakis *et al.*, 1999). Institutions and governing bodies, including the European Union, encourage wastewater reclamation for various applications. Apart from protecting the quantity and quality of water resources, the sustenance of the ecosystems they bear is also of increasing concern. A large number of factors put ecosystems at risk in the “anthropocene”. Insufficiently treated discharges and sewage overflows are only part of the damage, excessive use of fertilisers and pesticides in agriculture, diffuse pollution and urbanisation are some others. Around the world, societies aim to restore, maintain and improve ecological status of aquatic ecosystems to ensure the provision of goods and service that contribute to human well-being. Surface waters provide services for recreational activities (swimming, fishing, rowing and other water sports). Agricultural practices, such as irrigation, benefit from good quality of surface and groundwater. The industrial sector also benefits from water of sufficient quality to be used for industrial practices. Healthy aquatic ecosystems do not only benefit the direct users of water services, but also everyone that uses a service indirectly (e.g. consumer of agricultural products) or merely values the existence of the service (e.g. knowing that beautiful natural environment is in proximity). Improved qualities for aquatic ecosystems and the associated increase of biodiversity and environmental assets could therefore have important socio-economic benefits, including for public health and resilience towards future environmental pressures (OECD, 2014).

1.2. UWS AND THE PILLARS OF SUSTAINABILITY

These concerns and challenges are mirroring a general socio-political concern that took hold in the late 20th century. Various United Nations agencies, along with many individual nations, local governments and corporations have adopted sustainable development as an overarching goal of all economic and social development (OECD, 2006a). Definitions of the concept of sustainable development vary, however based on the Brundtland Report (WCED, 1987), sustainable development is the *"development that meets the needs of the present generation without compromising the ability of future generations to meet their own needs"*. Decisions based on economic, social and environmental conditions of the present and the future will accordingly be necessary across sectors, including the UWS. A new metric – that of sustainable development – will have to be adopted (Daigger, 2007) and the UWS will need to measure up to new standards of economic, social and environmental sustainability. This does not just entail impacts of the UWS to the three pillars of sustainability, but equally the impacts of changes in these three pillars to the system.

For instance besides the direct impacts discharges have on water quality, secondary impacts that were previously overlooked are now into consideration. Methane (CH_4) build-up is observed in sewer systems as well as nitrous oxide (N_2O) formation during the wastewater treatment process. With CH_4 representing 25 carbon dioxide (CO_2) equivalents and N_2O representing 298, they pose a problem that can no longer be ignored in evaluations (Kampschreur *et al.*, 2009). Conversely, the impacts of climate change on the UWS through rainfall intensification, temperature increase and other effects have not yet been adequately assessed (Astaraie-Imani *et al*., 2012; Langeveld *et al.*, 2013b). Furthermore, as climate change can impact all the components of the UWS, considering the whole system when assessing these effects is now necessary. Extremely relevant are also the issues of wet-weather flow management and drought, as the intensification of rainfall is sure to cause significant disturbances to the system hindering the efficient operation of the system and increasing the treatment needs (Hall *et al.*, 2012).

Article 9 of the EU Water Framework Directive (WFD) states that *"Member States shall take account of the principle of recovery of the costs of water services, including environmental and resource costs, […] and in accordance in particular with the polluter pays principle"* (European Council, 2000). Full cost recovery of water and sanitation services is a major component of the strive towards improving UWSs and rehabilitating the receiving water bodies (OECD, 2003). Full cost recovery implies the accounting of externalities such as water pollution and over-extraction.

Failure to recognise the total value of water assets has been identified as one of the factors that can set in motion a vicious cycle of underfunding in water and wastewater related infrastructure and management activities. Underestimation of the potential societal benefits arising from improved water quality can hinder the mobilisation of funding and the securing of the required investment. This can lead to inadequate maintenance of water infrastructure, poor management of the water resource and low-quality services and thus detriment of water quality and further reduction of the value perceived by society (OECD, 2010). Appraisal of the positive and negative externalities of sanitation services is of increasingly greater importance as it can compromise the success of the decision-making process and the resulting outcome (Nancarrow *et al.*, 2009; Scott *et al.*, 2012). Despite the growing interest in this concept, very few investment analyses are taking into account public valuation of these services as they are difficult to quantify, qualify and assign tangible monetary values to (Fan *et al.*, 2013). Apart from full cost recovery, efficient use of resources is also necessary in times of economic instability, such as the economic crisis experienced in Europe since 2008. With limited financial resources, integrated system analyses are necessary to work out the UWS interventions that would offer the best "value-for-money".

The urban water cycle is large in many aspects, not least in terms of geography (Figure 1-1). Water sanitation, water supply and watershed authorities, environmental agencies, municipalities as well as the industrial and agricultural sectors affect and are affected by the urban water cycle. Involvement of these stakeholders and integration of their concerns and interests are necessities for a sustainable UWS management. The contrasting visions and priorities between these stakeholders make their integration into the decision-making process a complex endeavour. For example, public priorities might lie in other aspects of governance or other types of environmental pollution. How is the administration meant to allocate funds for UWS when budget is restricted and the local society would rather see other issues addressed?

Perhaps what really makes this situation an incredibly complicated problem is that there is no panacean answer; no solution can be the most socially acceptable, most environmentally beneficial and have the highest financial return. A trade-off is almost always established and some of the aspects need to be compromised – preferably with the end-goal of sustainable development. The trade-off relationships are certainly not linear and the variables are many. Dynamic elements of socio–ecological systems, such as biophysical relationships, human behaviour, and feedbacks between them are poorly understood (Polasky *et al.*, 2011). There is no "one fits all" solution (or rather compromise) to be found either. Each decision-making challenge is specific to its characteristics and concerns. Be that as it may, due to the globalisation of issues and interrelationships of systems, the consequences of a wrong policy decision can now be more daunting than ever (Walker *et al.*, 2003). Finally, these interrelationships are not static. How the future unfolds is always uncertain and with global change it is highly uncertain (Polasky *et al.*, 2011).

Challenge:

UWS planning needs sustainable development as its principal metric and UWS decisions need to be accounting for social, environmental, financial, technical and other issues relating to their operation

1.3. Uncertainty in the UWS

Uncertainty is one of the most important concerns in any type of decision-making. It affects UWSs in various ways and compromises their efficacy (Walker *et al.*, 2003). Its presence may have formidable consequences when a wrong or simplified picture of the system is perceived. Alas, uncertainty is a fact of life and decision-makers have indicated more willingness to trust models if they are presented to them accompanied by the appropriate uncertainty analyses (Borowski and Hare, 2006). This is largely due to the liability and responsibility they assume for their decisions. A better understanding of the types of uncertainty would also potentially lead to more trust in these tools (Walker *et al.*, 2003).

Confusion among definitions impedes the inclusion of uncertainty concepts in decision-making practice. Even though Walker *et al.* (2003) have provided a typology of uncertainties, a considerable amount of ambiguity still surrounds the terms describing its effects. Numerous and various definitions of the concepts of: robustness, reliability, resilience, flexibility, functionality, stability, sensitivity, predictability, controllability and vulnerability can be found. These definitions are not always in agreement, especially in the literature between different disciplines (social science, engineering, ecology, urban planning, etc.), and are often used interchangeably by authors. This further complicates the task of decision-makers to adequately address uncertainty in decisions. The general notion captured by most is the idea of **satisficing** (or not) over the many plausible states a system might be found in (Jim W. Hall et al., 2012). Satisficing refers to the idea of **achieving**

acceptable (satisfactory) outcomes rather than optimal solutions (Stakhiv, 2011). Satisficing is hindered by uncertainty. Based on (Herder and Verwater-Lukszo, 2006) and (Walker *et al.*, 2003) we define two types of uncertainty in regard to their origin: **context** and **valuation** uncertainty.

Differentiating between types of uncertainty can help decipher what the causes of discrepancy are and who should bear the burden of proof. Different types of uncertainty require different responses: for example retrofitting infrastructure or devoting more research efforts. Decision makers can also separate between reducible and irreducible uncertainty and thus prioritise efforts and allocate funds (Walker *et al.*, 2003). Finally, properly categorised uncertainty along with precise definitions of how it is impacting the system (e.g. robustness, resilience, etc.) can facilitate better communication among decision-makers, policy analysts, scientists and other stakeholders.

Context factors are related to the **socio-political, environmental, financial and technical aspects that are subject to variability**. These might include climate, population, market prices and other factors that affect UWSs and ultimately shape their efficacy (Zhang and Babovic, 2011). Great uncertainty especially surrounds future changes and their combined effects. For example, issues of wet-weather flow management and drought pose important threats, as the intensification of rainfall is expected to cause significant disturbances to the system hindering its efficient operation and increasing the treatment needs. Factors besides climate change, such as population growth and urbanisation can also have a great impact on the future of the quality and quantity of water in urbanised catchments. However, the combined or relative effects of future changes such as climate change, urbanisation and population growth on the UWS have not been given extensive research focus thus far (Astaraie-Imani *et al.*, 2012). Finally, the evolution of important economic factors – principally, water and energy prices – can also potentially increase the uncertainty regarding the operational equilibrium of UWSs. Variability is an intrinsic quality of these factors. As such, this type of uncertainty is deemed inherent and irreducible - in contrast to epistemic uncertainty that can be reduced by more research and empirical efforts (e.g. measurement errors) (Belia *et al.*, 2009; Walker *et al.*, 2003). Context uncertainty may affect the system in different ways, from operational disturbances to infrastructural damage, and different concepts are used to describe the system's capacity to deal with these disturbances: robustness, resilience, reliability, functionality, etc. Nonetheless, literature on the terms can often be ambiguous and contradicting.

Valuation uncertainty emerges when **one attempts to describe and assess the system**. For example, if the system is to be modelled, the modelling process itself involves a relative inaccuracy in the representations and predictions. This discrepancy is intrinsic to the process of modelling, i.e. when interpreting and attempting to represent natural processes using mathematical models. Even when a “good fit” is found between a measurement of a concentration and its modelled estimate for example, it is not necessarily the case that the model has adequately provided the right answer for the right reason (Olsson and Andersson, 2006). Valuation uncertainty, though always present, is often neglected (Belia *et al*., 2009; Refsgaard *et al.*, 2006). This is also evident in the literature review discussed later (Section 1.5). Other valuation uncertainties arise with choices regarding assessment, for example which criteria are considered the most appropriate by the decision-makers (Dominguez *et al.*, 2011) or values of factors such as the discount rate in a Cost Benefit Analysis (CBA) or simply the choice of the assessment tool itself.

Challenge:

Uncertainty for the UWS has different origins: in the physical world (context) and during modelling for decision-making (valuation). It manifests itself and affects the system in various ways and as such appropriate uncertainty analyses are an indispensable part of the decision-making process

1.4. Adaptive Management – a new decision-making paradigm

Authors have remarked on the fact that stationarity and its associated implications for UWS management are dwindled under the weight of rapid and unpredictable changes (as detailed in previous sections) (Brown, 2010; Milly *et al.*, 2008). This suggests that current static design and upgrading practice is unsuitable as it is based on the premise that the future can be effectively predicted (Brown, 2010; Dominguez and Gujer, 2006). Unless current management regimes undergo a transition towards a more adaptive approach sustainable management of water and wastewater resources cannot be realised (Herrfahrdt-Pähle, 2013; Kashyap, 2004; Pahl-Wostl *et al.*, 2007). Williams *et al.* (2007) defined adaptive management as *“a systematic approach for improving resource management by learning from management outcomes”*. It is based on the notion that cooperative learning-by-doing coupled with scientific progress supports sustainable development (den Uyl and Driessen, 2015). The idea of adaptive management has already been discussed in the field of ecosystem and resource management for quite some time now (for example, in Holling (1978) and Walters (1986)). Accordingly literature has been suggesting a shift in UWS management to a more adaptive and flexible approach to ensure operation under fast changing socio-economic and environmental conditions (Giacomoni and Berglund, 2015; Meire *et al.*, 2008; Pahl-Wostl, 2007). With regards to improving the ecological resilience of the water bodies of UWSs, adaptive approaches seem the most promising (Kopprio *et al.*, 2014). Adaptive management is thus widely considered to be the best available approach for managing biological systems in the presence of uncertainty (Pahl-Wostl, 2007; Westgate *et al.*, 2013). Furthermore, in current UWS management in the industrialised world there is little room for the conventional "Planning-Design-Operation" paradigm. Already existing infrastructure needs to be retrofitted to respond to the new emerging challenges rather than be designed "from scratch". So far adaptation measures have been mostly technical (e.g. extending systems) but authors suggest that governance adaptability should also be increased (Pahl-Wostl *et al.*, 2008a). Increasing adaptability is expected to be one of the major future challenges for the water sector and water governance (Herrfahrdt-Pähle, 2013). This transition to adaptive governance is particularly complicated for systems with large-scale infrastructure with a life-span of decades (such as the UWS) as there are few opportunities for learning and lock-in situations are more likely (Pahl-Wostl *et al.*, 2008a). Adaptive management at the operational level is usually less troublesome.

Adaptive management relies strongly on a decision-making process that is participatory and has active stakeholder involvement. Stakeholder participation allows for the inclusion of a wide range of different perspectives rather than decision-making by specialists and experts in isolation – something that is particularly important in early design and planning stages (Pahl-Wostl *et al.*, 2008a). There is also an apparent disparity between experts, researchers, stakeholders and policy makers, which often leads to an inadequate application of models and other support tools in the decision-making process (Janssen *et al.*, 2008). Early involvement of stakeholders in the development and application of support tools can help bridge that gap and spread the application of model-based tools in the decision-making process (Brandon, 1998). In terms of the adaptive capacity of a system, a broad range of perspectives can facilitate adaptation by recognising new challenges and needs for institutional change (Pahl-Wostl *et al.*, 2008a). For these reasons, the European WFD encourages that *"stakeholders are invited to contribute actively to the process and thus play a role in advising the competent authorities"* (European Council, 2000).

Challenge:

Adaptive management is considered to be the best available approach for managing UWSs in the presence of uncertainty. The decision-making process and the tools it employs should therefore embrace an adaptive planning and operation process

1.5. Literature review of decision-support tools

In this seemingly dire decision-making setting, one must not overlook the fact that the tools and methods made available by research and technology are now more advanced and all encompassing than ever. The focus of research and technology development to support these challenges is demonstrated by the ever-increasing literature addressing these issues. Searching though the

Scopus database for the terms "sustain*" (to include variations of sustainability, sustainable, etc.), "uncertain*" (to include variations of uncertainty, uncertain, etc.) and "adapt*" (to include variations of adaptation, adaptive, etc.) demonstrates an increased scientific output in these topics, mirroring socio-political concerns of recent decades. Figure 1-2a presents the number of documents published per year containing each of the terms along with the terms "wastewater" or "water" in the title, abstract and keywords.

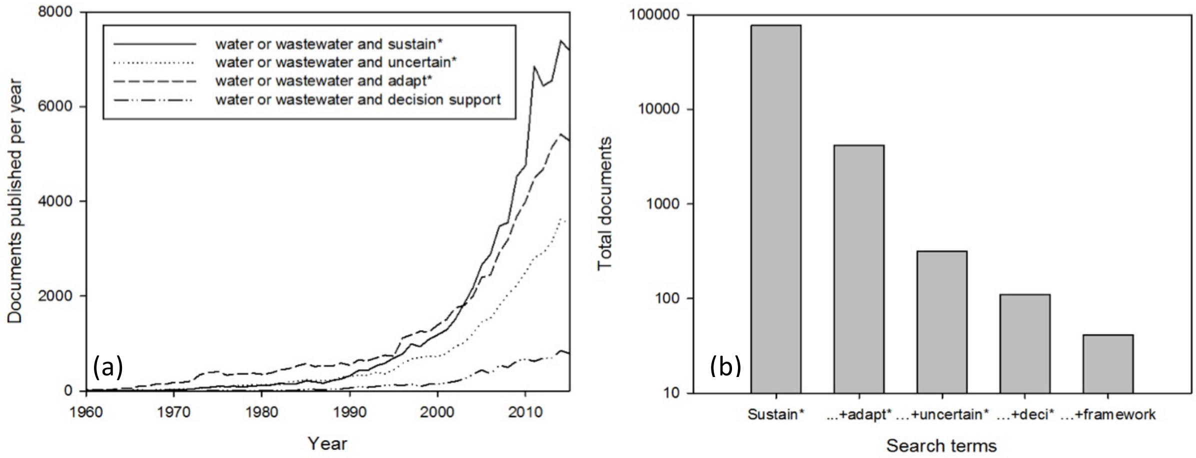


Figure 1-2 - Results of search through the Scopus database for (a) number of documents published for each search term per year; and (b) number of documents published for combinations of terms

Technologies are now available for the conservation of water resources; reduction of water consumption, reclamation and reuse of wastewater; the management and extraction of energy from the wastewater stream; the recovery of nutrients; the separation of wastewater sources; and not least in instrumentation, control and automation. At the same time tools for data processing, information processing and assessment have advanced significantly. From statistical analyses, numerical algorithms and mathematical modelling to large-scale simulations and data mining techniques, the capacity to predict and address environmental management problems has advanced significantly (Poch *et al.*, 2004). The complexity of UWS decision-making however requires more elaborate approaches than the mere application of conventional numerical models (Garrido-Baserba, 2013). For this purpose integrative approaches of expert systems, rule-based systems and other tools also started appearing, giving rise to advanced tools (such as environmental decision-support systems) for multi-criteria decision-making (Guariso and Werthner, 1989; Poch *et al.*, 2004; Rizzoli and Young, 1997).

There has been an increased application of decision-support tools (DSTs) in UWS literature: the results of a search in the Scopus database for articles published per year containing the term "decision support" along with the terms "wastewater" or "water" in their title, abstract or keywords are also presented in Figure 1-2a. Their increased study can be attributed to the multiple benefits they appear to offer for UWS management. At higher levels of executive decision-making, DSTs offer the ability to incorporate qualitative knowledge from different agents; the ability to integrate various tools, analyses and metrics; can summarise expertise from different fields (ecology, hydrology, engineering and others); facilitate in the communication between scientific outcomes and decision-making; and provide easily communicable outcomes. For technical, mid and low level management, DSTs can support decision-making by incorporating various monitoring technologies (e.g. data acquisition, data validation and analysis); integrating expert knowledge with models and other tools; providing both online and offline responses; helping the user to formulate and diagnose the problem. Given this magnitude of abilities, DSTs can solve problems of high complexity; can cope with situations where experience is essential for finding a solution; reduce the time need to identify the problem and make a decision; and improve the consistency, quality and argumentation of decisions (Hamouda *et al.*, 2009; Poch *et al.*, 2012). This approach to decision-support is conducive to effective decision-making for sustainable development as advocated by authors (Bradley *et al.*, 2002; Daigger, 2007; Daigger and Crawford, 2004).

Looking at the literature on these terms (and their variations) in the fields of water and wastewater, the extensive focus placed on them individually is clearly apparent (Figure 1-2a). Studies looking at combinations of the terms are reduced in number, despite the strong interrelations of the terms for water and wastewater management (as previously elaborated) (Figure 1-2b). Searching the Scopus database with all the terms ("(water or wastewater") and sustain* and adapt* and uncertain* and deci* and framework") results in 41 documents of which 27 are research articles. The most notable frameworks appearing as well as other noteworthy examples are presented in Table 1-1 and discussed below.

Kwakkel *et al.* (2015) presented an integrated assessment meta-model for the optimisation of possible pathways for decision-making. Both context and valuation uncertainty were taken into account in the analysis, however they were not explicitely differentiated and their impacts on robustness were mixed. Girard *et al.* (2015) presented an innovative "bottom–up meets top–down" integrated approach by integrating sustainability goals with adaptive management. The framework considers local socio-economic criteria as well as climate change impacts to produce a set of adaptation measures. This approach however does not include other possible future changes (such as urbanisation and population growth) nor is the uncertainty of the valuation assumptions taken into account. A modelling framework was presented by Giacomoni and Berglund (2015) to assess the effects of management strategies to municipal water demands and the sustained storage in surface water supply reservoirs. Economic considerations of these managements were not taken into account, nor was valuation uncertainty. A back-casting methodology was presented by van Vliet and Kok (2015) to identify management actions that are robust against various future scenarios encompassing socio-economic and environmental changes. While a wide range of context uncertainty was evaluated, parametric and other valuation uncertainties of the scenarios and response models was not considered. Poff *et al.* (2015) presented a decision-support framework that explores the trade-offs in ecological and engineering performance metrics as defined by stakeholders under future scenarios. A conceptual framework was presented by Butler *et al.* (2014) to link emerging threats to their consequences on social, economic and environmental recipients and facilitate mitigation and adaptation. An integrated framework to value investments in urban water systems under uncertainty was presented by Deng *et al.* (2013). The uncertainty studied was through the application of future scenarios (context uncertainty) but no valuation uncertainty was taken into account nor impacts to society. Starkl *et al.* (2013) presented a planning-oriented sustainability assessment framework facilitating participatory planning and the evaluation of complex interactions between environmental and social systems. Hattermann *et al.* (2011) proposed a model-based participatory framework for measure planning. Proposed measures are evaluated using developed scenarios and stakeholder participation. Management objectives and applicable scenarios include various sustainability aspects, however valuation uncertainty is not taken into account despite the model-heavy nature of the framework. Mahmoud *et al.* (2009) presented a formal scenario development framework for future scenarios (context uncertainty) and even though they discuss at length the uncertainty regarding their development (valuation uncertainty) they do not include any qualitative or quantitative estimations of it. To the best of the author's knowledge, no decision-support framework has been presented in literature that includes the impacts of both context and valuation uncertainty and explicitly differentiates between the two. This comes in agreement with observations by other authors (e.g. Belia *et al*. (2009) and Refsgaard *et al.* (2006)) that valuation uncertainty and its consequences are often neglected in evaluations and DSTs based on modelling. Another omission made apparent is that only few of the available decision-support frameworks truly address both the social, environmental and financial consequences (T_S, T_E, T_F in Table 1-1) of decisions as well as the consequences of changes in these domains to the system (F_S, F_E, F_F in Table 1-1). This appears to be due to the fact that quantifying these aspects is often challenging - an endeavour further complicated when single or aggregated metrics are pursued in the name of simplicity. **The lack of a DST, and specifically a framework, to support in all the aforementioned challenges is therefore clear.**

Reference	*Type of tool*	*U*	*U_C*	*U_V*	*U_{Dif}*	*T_S*	*T_E*	*T_F*	*F_S*	*F_E*	*F_F*	*AM*	*Tool description*
Kwakkel *et al.* (2015)	MM	x	x	x			x	x	x	x		x	Meta-model for optimisation
Girard *et al.* (2015)	FW	x	x			x	x	x		x		x	"Bottom–up meets top–down" integrated framework for decision-support
Giacomoni and Berglund (2015)	FW	x	x				x	x	x	x		x	Adaptive system modelling framework assessing the effects of strategies to municipal water demand and sustained water availability
van Vliet and Kok (2015)	M	x	x			x	x	x	x	x	x	x	Back-casting methodology for identifying robust actions under future scenarios
Poff *et al.* (2015)	FW	x	x			x	x	x		x	x		Eco-engineering decision-scaling framework exploring stakeholder-defined metrics of engineering and ecological performance
Butler *et al.* (2014)	FW	x	x			x	x	x	x	x	x	x	Framework linking emerging threats to social, environmental and economic consequences to facilitate adaptation
Deng *et al.* (2013)	FW	x	x				x	x		x	x		Integrated framework to value investments in urban water management systems under uncertainty
Starkl *et al.* (2013)	FW					x	x		x	x		x	Planning-oriented sustainability assessment framework for participatory planning
Hattermann *et al.* (2011)	FW	x	x			x	x	x	x	x	x	x	Model-based framework for participatory measure planning
Mahmoud *et al.* (2009)	FW	x	x		x				x	x	x		Framework for a formal approach to scenario development

Table 1-1 - Notable decision-support frameworks and other tools appearing in literature. U: Uncertainty; U_C: Context uncertainty; U_V: Valuation uncertainty; U_{Dif}: Type of uncertainty explicitly differentiated; T_S, T_E, T_F: Social, environmental and financial impacts (respectively); F_S, F_E, F_F: Impacts from changes in the social, environmental, financial (economic) domains (respectively); AM: Adaptive management; M: Methodology; FW: Framework; MM: Meta-model

2. Thesis Objectives

2. THESIS OBJECTIVES

Given the defined challenges and the present state of the art, the main objective of this thesis is to describe a decision-support framework for management decisions in the UWS that:

- Is based on the principles of adaptive management
- Can incorporate socio-economic, environmental and technical aspects of operation
- Addresses context and valuation uncertainty explicitly and separately

The secondary objectives of this thesis are to apply said adaptive management framework to a real UWS case study and:

- Demonstrate its real-world applicability and utility
- Demonstrate how environmental externalities can be included in sustainability assessments through monetisation
- Apply a sustainability assessment of costs and environmental benefits
- Provide metrics for robustness, reliability and resilience
- Assess the impacts of context uncertainty on the operation of the system
- Assess the impacts of valuation uncertainty on the decision taken

And to demonstrate an alternative approach to sustainability assessment by:

- Applying a novel assessment method that integrates various tools
- Estimating the life-cycle impacts of the studied system
- Including the functions of the receiving medium in the assessment
- Applying monetised weighting to estimated impact

3. MATERIALS AND METHODS

3. Materials and Methods

3.1. Studied catchments

3.1.1. The Congost Catchment

The Besòs River catchment is located in Catalonia, in north-eastern Spain, with most of its surface within the Vallès region. The Besòs catchment is one of the most populated in the area, with about two million people connected to the system. The basin is under the management of the Besòs River Basin Authority, the Consorci per la Defensa de la Conca del Besòs (CDCB).

The Congost River is one of the main tributaries of the Besòs River, with a length of 43 km. It is characterised by a typical Mediterranean hydrological pattern and average flow of 0.73 m^3/s. The river flow during summer months is mostly WWTP effluent, which makes the effluent quality a very critical aspect of the ecological well being of the river. The Congost River receives the discharges of five municipalities: La Garriga and L'Amettla del Vallès, with their own sewer system and WWTP, and Les Franqueses del Vallès, Canovelles and Granollers, which all share the sewer system and WWTP (Figure 3-1). There is high industrial activity in the area, all of which is connected to the urban sewer systems. The major part of the network of both sewer systems is combined (jointly collecting wastewater and rainfall runoff in the same conduits). The La Garriga WWTP is located at the town of La Garriga and treats domestic and industrial wastewater of about 27,000 m^3/day. The Granollers WWTP is located within the town of Granollers and treats domestic and industrial wastewater. The WWTP was constructed in 1992 with a primary physico-chemical treatment. In 1998 it was extended with a biological process based on an activated sludge system with a modified Ludzack-Ettinger configuration at 25,000 m^3/day and anaerobic digestion.

With the establishment of the EU WFD regional water authorities of member states were required to develop and publish River Basin Management Plans (RBMPs) to improve or maintain the ecological state of all water bodies for each river basin district. Accordingly, the Catalan Water Agency (ACA) has set out a RBMP for each river basin, including a set of measures to be implemented, with emphasis on the most threatened rivers of each river basin. During the 2000-2004 monitoring period the mean ammonium (NH_4^+) concentration in the Congost River was 26.95 g/m^3, the highest measured concentration in the lower Besòs River basin (Devesa, 2006). This was primarily due to the fact that about 60% of the river flow was WWTP effluent and that the main plant of the catchment (Granollers WWTP), with a conventional activated sludge system, was not performing any nitrogen removal at the time due to its limited capacity. In addition, the local population had been steadily increasing putting further pressure on the sanitation system. For these reasons, the water authorities had included interventions on the Granollers UWS in the list of measures for the first cycle of the RBMPs (year 2007). The framework presented in Chapter 4 was applied in this decision-making problem, presented in more detail in Chapter 5.

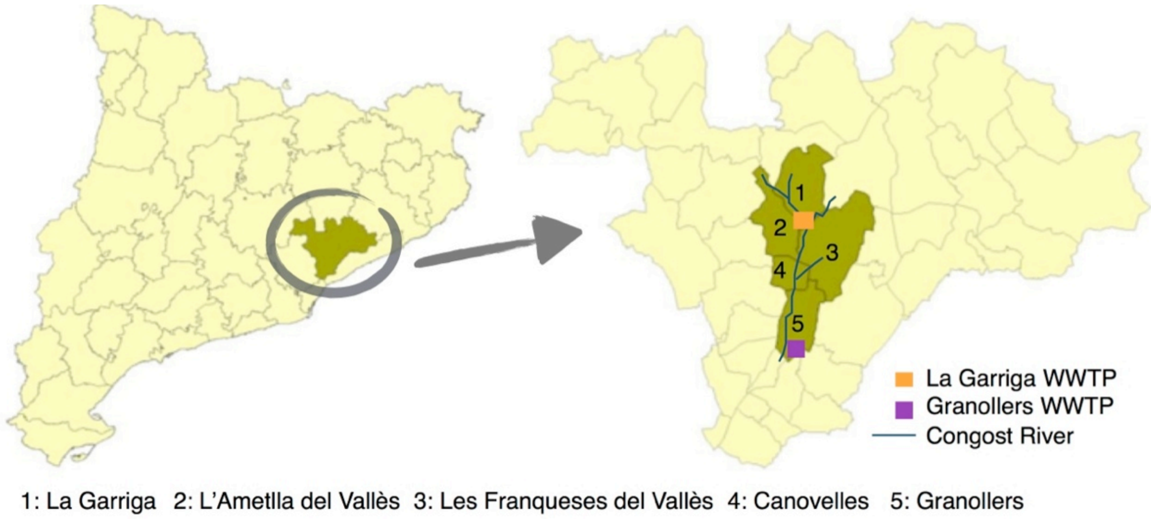


Figure 3-1 - The Congost River catchment with main discharging municipalities

3.1.2. *The Eindhoven Catchment*

The Eindhoven catchment area is located in the southern Netherlands. Its water barriers, waterways, surface waters and wastewater system are under the jurisdiction of the Dommel Waterboard (Waterschap de Dommel - WdD), the regional water authority. The Eindhoven WWTP is the largest plant managed by the WdD with its catchment area located in and around the city of Eindhoven (Figure 3-2). Despite it being a relatively small river, the Dommel River is the largest in the catchment area. It originates in north-eastern Belgium and flows in a northerly direction, through the city of Eindhoven. Through its 150 km length it receives discharges from the Eindhoven WWTP and 200 combined sewer overflows (CSOs). The collection system drains 4,000 ha of impervious area. The WWTP treats wastewater from ten municipalities (Son en Breugel, Nuenen, Eindhoven, Geldrop-Mierlo, Veldhoven, Waalre, Heeze-Leende, Eersel, Bergeijk and Valkenswaard) which are scattered over a large area requiring an extensive wastewater collection and transport system. It has a design capacity of 750,000 population equivalents (PE) with a design load of 136 g Chemical Oxygen Demand (COD)/day/PE making it the third largest in the Netherlands. Between 2003 and 2006 the WWTP was renovated to comply with nutrient removal standards. The received wastewater is treated in three parallel lines, each consisting of a primary settler, a biological tank and four secondary settlers. The plant has a modified UCT (University Cape Town) configuration for biological COD, N and P removal. The Dommel River's relatively low flows (1-10 m^3/s most of the year) mean that during summer time the base flow is made up of to 50% of Eindhoven WWTP effluent. Strict water quality standards apply in the river to protect fish populations. As a result, during dry months effluent that meets its quality standards can nevertheless exceed surface water quality standards. The WdD has therefore launched a series of comprehensive studies and analyses to identify the most cost-effective set of measures to meet the necessary standards (Benedetti *et al.*, 2013b; Langeveld *et al.*, 2013a, 2013b; Weijers *et al.*, 2012). Chapter 6 presents an alternative sustainability analysis (as part of the framework) applied to the Eindhoven UWS.

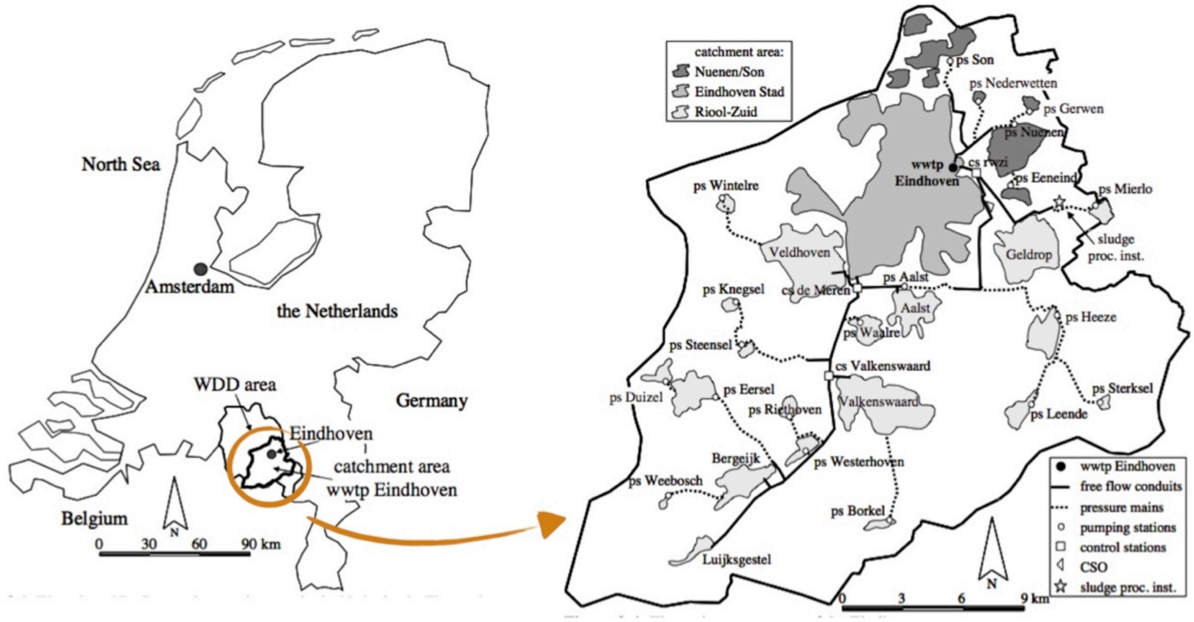


Figure 3-2 - The Eindhoven catchment and surrounding municipalities. Adapted from Schilperoort (2011)

3.2. INTEGRATED MODELLING AND THE WEST SIMULATION SOFTWARE

Mathematical models can be seen as compilations of the knowledge available about a process or a system. Mathematical modelling of UWSs has a long history, especially process-specific models of individual subsystems. Integrated models are made up of a compilation of several sub-models in a single model and their application has been increasing in the past decade. This is due to the fact that an integrated model allows for a better understanding of the role of each individual component as part of a larger system, the interactions between the components and the function of the system as a whole. Multiple models of varying complexity have been presented in literature for the various components of an UWS.

Various methods can be employed to obtain dynamic influent flow rate data, either by means of simple equations or through more sophisticated models. Model-based influent scenario generation has proved to be a valuable tool to obtain dynamic influent flow rate data as it can provide a significant reduction to the time and cost necessary for measuring campaigns, fill missing-data gaps in the generated profiles, and be used as a basis for the development of additional scenarios (Gernaey *et al.*, 2011; Snip, 2015). For an overview of the available influent generation methods, the reader is directed to the review by Martin and Vanrolleghem (2014) and to Talebizadeh *et al.* (2016) for discussion on advantages and shortcomings of the various methods. Apart from municipally generated wastewater, surface runoff also needs to be accounted for when modelling a catchment. Surface runoff is created when excess rainfall cannot infiltrate the impermeable surface area in a catchment and therefore ends up in the sewer system. When modelling surface runoff several phenomena need to be taken into account, including soil infiltration and evapotranspiration. There are various available models of varying complexity describing surface runoff (Butler and Davies, 2010). When it comes to the mathematical modelling of sewer systems, deterministic and conceptual approaches are available (Butler and Davies, 2010). Deterministic models are generally more computationally intensive and require detailed information about the system, making simplified conceptual models more useful in that regard. Most conceptual models describe the flow through the sewer using series of linear reservoirs, with the output of a previous reservoir being the input of the next (Benedetti, 2006).

WWTP models are a compilation of the processes occurring in an actual WWTP including activated sludge biodegradation, oxygen transfer, sedimentation and hydraulics. WWTP models have been widely developed and applied both in academia and practice for the design, assessment, prediction and control of plant-wide operations and processes. The first "standard" activated sludge model (Activated Sludge Model No. 1 – ASM1) was introduced in the 1980s, describing carbon removal, nitrification and denitrification (Henze *et al.*, 1987). ASM2 (Henze *et al.*, 1995) and ASM2d (Henze *et al.*, 1999) were introduced a decade later to include the biological removal of P, along with ASM3 correcting some ASM1 defects and including storage of organic substrates (Gujer *et al.*, 1999). (Gernaey *et al.*, 2004) presented a comprehensive review of the state of the art and application of these models.

The USEPA QUAL family of models was the standard in river water quality modelling in the 1980s and 1990s. QUAL2E is a multi-constituent river ecosystem model that was developed as a series of modifications to the previous QUAL models (Brown and Barnwell, 1987). The model includes various processes and interactions of dissolved oxygen (DO), organic matter, N, P and algae. Due to some mass-balance problems with QUAL2E (Shanahan *et al.*, 1998), the River Water Quality Model No. 1 (RWQM1) was developed to be compatible with the activated sludge family of models (Reichert *et al.*, 2001). RWQM1 introduced additional processes that were not included in QUAL2E and bacterial biomass as a model component. Furthermore, Lijklema *et al.* (1996) presented the DUFLOW water quality model simulating river hydraulics and biochemical processes, including DO, organic matter and ammonia dynamics.

Simulation environments are software that allow the representation and simulation of (in this case) UWS configurations. They usually contain extended libraries of predefined process unit models (such as the ASM models) that can be integrated to simulate the desired system. Various simulation packages are available including, but not limited to, SIMBA (Alex *et al.*, 1999), AQUASIM (Reichert, 1994), and WEST® (Vanhooren *et al.*, 2003) (www.mikebydhi.com). For critical reviews of integrated urban wastewater modelling and the various modelling methods and software, the reader is directed to Bach *et al.* (2014) and Rauch *et al.* (2002). The integrated models of both the Congost and Eindhoven UWSs were developed using the modelling software WEST®, as an integration of three separated models for the catchment and sewer system, WWTP and river. To model surface runoff and sewer, the simplified conceptual KOSIM model (ITWH, 2000) available in WEST® was used (Solvi, 2006). Both WWTPs were modelled using ASM2d (Henze *et al.*, 1999) and both Congost and Dommel Rivers were modeled using DUFLOW (Lijklema *et al.*, 1996).

3.2.1. Congost catchment: Model development and calibration

The Congost integrated model was developed to include both catchments for La Garriga and Granollers, both WWTPs and the river section between the two. As the application in Chapter 5 is only concerned with the Granollers part of the model, the following description is focused only on that. The model layout as developed in the WEST® platform is provided in the Annex, Figure 10-1. The model was developed as an integration of three separated models for the catchment, WWTP and river (Figure 3-3). The KOSIM modelling tool contains model descriptions of various processes of the sewer system elements (urban catchment, pipes, storage tanks), listed in Table 3-1. The processes considered in the modelling application have been calibrated to the study; processes not considered were left to their default values.

At the catchment, both domestic and industrial flows were represented, from the towns of Les Franqueses del Vallès, Canovelles and Granollers. Modelled pollutants generated at the catchment are soluble and particulate COD, ammonia, total nitrogen (TN) and total phosphorus (TP). Using population density, water consumption per inhabitant and total catchment area, the dry weather flow is determined. The catchment module was also used to introduce the rain series as input for the wet weather flow. The spatial distribution of rainfall is assumed to be uniform over the catchment. The runoff entering the sewer system depends on the area connected to the drainage system and the magnitude of impervious area in the catchment. The influent data generator was used to produce catchment data (surface rainfall runoff and influent WWTP flowrate and pollutant concentrations), based on demographic, meteorological and influent measurements. As wastewater production is not the same throughout the day, daily, weekly and seasonal influent profiles of volume and NH_4^+ concentration were produced by use of patterns in the model. Flow, pollution and infiltration patterns can be found in Table 3-2.

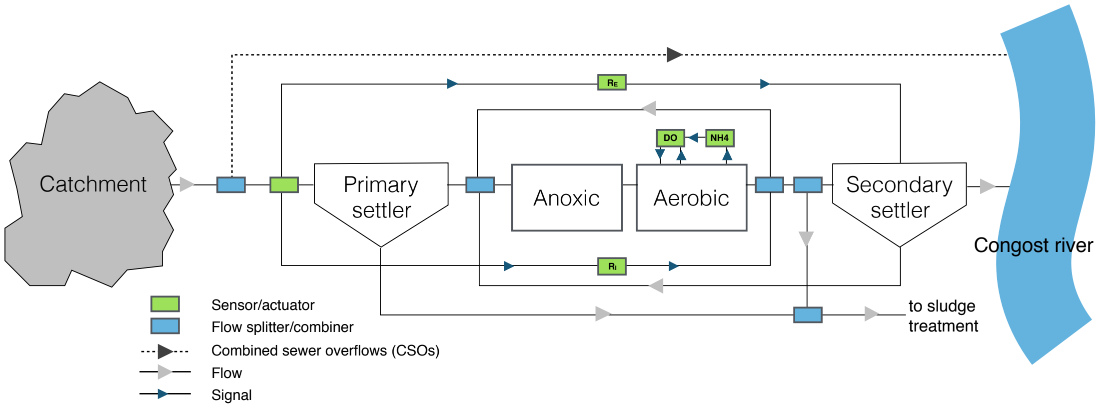


Figure 3-3 - Graphic description of the mathematical model for the studied UWS

Subsystem	**Water**	**Pollutants**
Atmosphere	Evaporation*/Rain	
Surface	Runoff generation	Accumulation*/Washoff*
Sewer network	Flow Storage* Combination and splitting	Pollutant generation Pollutant transport Sedimentation* Resuspension*

Table 3-1 - Urban drainage processes represented in KOSIM. Adapted from Solvi (2007). * indicates processes not considered in this application.

Element	Type of profile	Type of values	Pattern values
Flow	Daily	Hourly	0.96, 0.87, 0.83, 0.67, 0.62, 0.59, 0.59, 0.68, 1.10, 1.31, 1.34, 1.33, 1.31, 1.29, 1.22, 1.28, 1.29, 1.24, 1.18, 1.15, 1.21, 1.44, 1.38, 1.29
Pollution	Yearly	Monthly	0.98, 1.33, 1.34, 0.86, 0.75, 0.94, 1.07, 0.78, 1.05, 0.91, 0.89, 1.09
	Daily	Hourly	0.82, 0.77, 0.56, 0.49, 0.39, 0.45, 0.45, 0.52, 0.99, 1.50, 1.55, 1.46, 1.34, 1.26, 1.21, 1.20, 1.21, 1.07, 1.09, 1.10, 1.09, 1.14, 1.16, 1.17
Inflitration	Yearly	Monthly	1.24, 0.75, 1.10, 1.62, 1.82, 1.20, 1.00, 0.44, 1.00, 1.78, 0.01, 0.005

Table 3-2 - Flow, pollution and inflitration patterns used by influent generator model. Calibrated using measurement data from the Granollers catchment and WWTP for 2007.

Calibrated parameter values of the catchment module can be found in Table 3-3; unmodified parameters were kept to the default values by WEST®. As wastewater production differs during the weekend, a multiplication factor was used according to measurements. For the surface runoff, rain events at the catchment were classified by magnitude (mm) and related to influent measurements reaching the WWTP over a four-hour time frame. Details on the sources of data used for calibration can be found in Table 3-4.

Name	Value	Unit
Total area	600	ha
Infiltration	0.05	l/s/ha
Inhabitants		-
Wastewater per inhabitant	0.2	m^3/day
Weekend factor	0.9	-

Table 3-3 - Calibrated parameter values for the catchment module of the model

Data	Use in model	Year	Source
Demographic: Town populations	Population equivalents	Historical and current	Statistical Institute of Catalonia (http://www.idescat.cat)*
Meteorological: Rainfall data from three stations in the catchment	Surface runoff calibration	Historical and current	Meteorological Service of Catalonia (http://www.meteocat.cat)*
WWTP Measurements: influent volume, pollutant concentrations	Influent generator calibration WWTP calibration	2007, 2011	Catalan Institute for Water Research (ICRA) (http://www.icra.cat/) Consorci per a la Defensa de la Conca del Riu Besòs (CDCB) (http://www.besos.cat)

Table 3-4 - Data used during model calibration, along with their sources. * indicates publically available data

Figure 3-4 (a, b & c) presents summarising results of the calibration procedure for inflow and NH_4^+ concentration. Figure 3-4 (d) presents summarising results of the calibration of the surface runoff model for two weeks. No chemical processes (e.g. degradation) in the sewer system or backwater effects are represented. Any volume of water reaching the WWTP that is beyond hydraulic capacity overflows through a bypass into the Congost River.

The primary settler (4623 m^3 volume) was modelled using the primary Otterpohl Freund model (Otterpohl and Freund, 1992). The biological reactor is divided into two compartments, an anoxic (of 650 m^3 volume) and an aerobic (of 6500 m^3 volume). Internal and external recirculations are set by a proportional controller based on the influent flowrate. There is a cascade controller for DO and NH_4^+ with a proportional-integral (PI) algorithm and proportional controllers for internal and external recirculations. Following the biological reactor there is deoxygenation tank of 10 m^3, from where sludge is wasted. An ideal point settler model was used to describe the secondary clarifier. All control and operation settings are were determined using plant data and adjusted accordingly during the calibration procedure, as provided in Table 3-5. All kinetic and stoichiometric parameters for the biological reactor and settlers were left at default and are given in Table 10-1 of the Annex.

The equations of the implemented model are described by ordinary differential equations and are solved with the use of the numerical variable step size Runge-Kutta method included in WEST®.

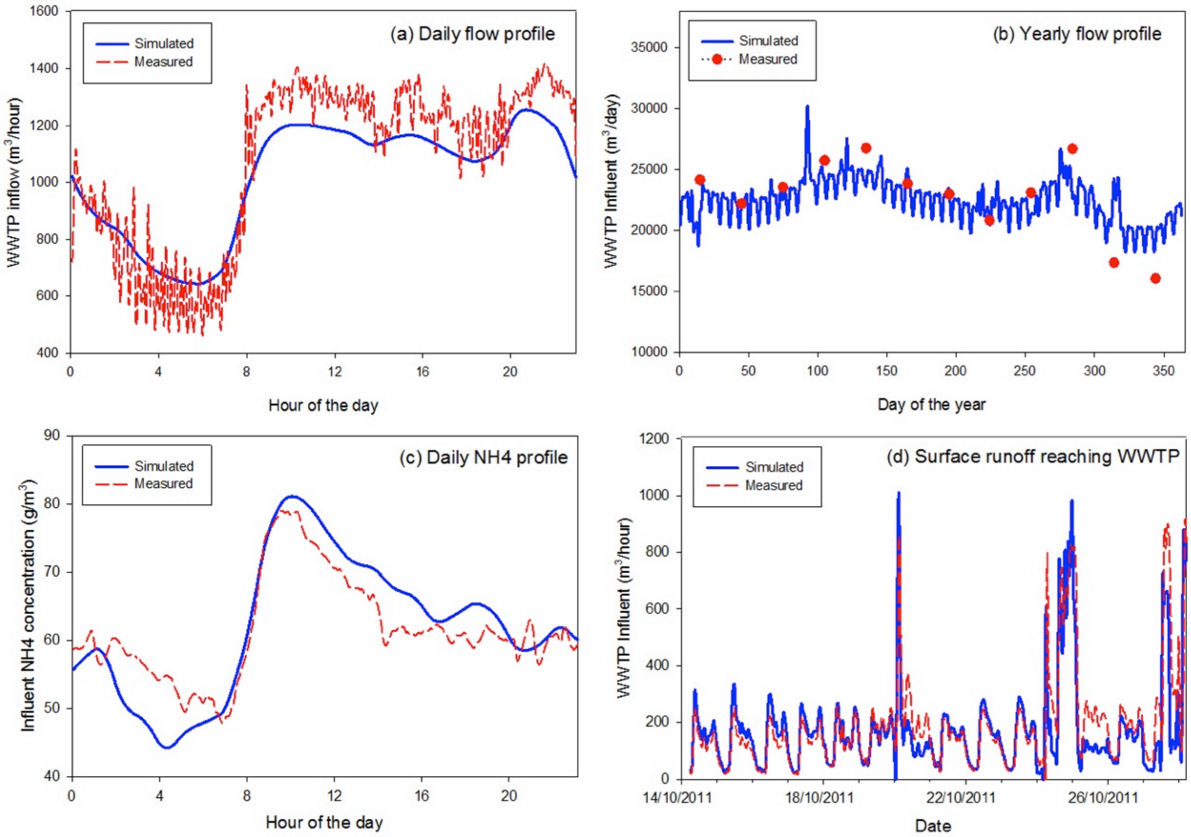


Figure 3-4 - Model estimates compared against (a) a typical daily influent flow-rate profile from year 2007; (b) the yearly influent WWTP flow-rate profile for 2007; (c) a typical daily WWTP influent NH_4^+ profile from year 2007; and (d) two weeks of surface runoff from October 2011.

DO Control			
Reading: DO concentration in aerobic reactor			
Name	Value	Unit	Description
K_P	500.00		Factor of proportionality
T_I	0.02	d	Integral time
u0	60.00		No error action
u_Max	1000.00		Maximum control action
u_Min	0.00		Minimum control action
y_S	2.00		Setpoint value for controlled variable
NH_4^+ **control**			

Reading: NH_4^+ concentration in aerobic reactor			
Name	Value	Unit	Description
K_P	-1.00		Factor of proportionality
T_I	0.10	d	Integral time
u0	0.50		No error action
u_Max	5.00		Maximum control action
u_Min	0.10		Minimum control action
y_S	3.33		Setpoint value for controlled variable
Internal recirclulation control			
Reading: WWTP inflow			
Name	Value	Unit	Description
ConstantRatio	2.67		Ratio between measured value and controller output
External recirculation control			
Reading: WWTP inflow			
Name	Value	Unit	Description
ConstantRatio	2.25		Ratio between measured value and controller output

Table 3-5 - Control and operational settings for the WWTP

3.2.2. Eindhoven catchment: Model development

The Eindhoven integrated model used is made up of the integration of three separate models – for the catchment and sewer system, the WWTP and the river. The hydraulics of the sewer system were represented as tanks-in-series in a simplified hydrodynamic sewer model based on InfoWorks version 9.5 (www.innovyze.com). A conceptual catchment model based on empirical relationships is used to generate the influent water quality, as water quality modules in sewer models are still not considered sufficiently reliable (Bertrand-Krajewski, 2007; Langeveld *et al.*, 2013a). Event mean concentrations were applied for the CSO outputs into the river, derived from two years of monitoring data of CSOs. The NH_4^+ loads were given by the model, whereas the P loads were estimated based on empirical measurements. The WWTP was modelled using the ASM2d biokinetic model modified by Gernaey and Jørgensen (2004) and Takács *et al.* (1991) for settler modelling. A surface water model has been set up to represent the Dommel River and its main tributaries as tanks-in-series, as a simplified version of the Duflow Modelling Tool (Stowa/MX.Systems 2004). The DUFLOW model is based on the one-dimensional partial differential equation that describes nonstationary flow in open channels and allows for the construction of 1D-hydrodynamic models including substance transport and processes. 70 river sections and 34 discharge points, representing (clusters of) CSOs and the WWTP effluent are combined together to describe the Dommel River system. The processes in the river model with DO and NH_4^+ as state variables include BOD decay, re-aeration, algae production and respiration, nitrification and settling of particulate organic matter (Langeveld *et al.*, 2013a). The equations of the implemented tanks-in-series model are described by ordinary differential equations and are solved with the use of the numerical Variable-coefficient Ordinary Differential Equation (VODE) solver included in WEST®.

More details on the model development for the Eindhoven catchment can be found in Benedetti et al. (2013b), Langeveld et al. (2013a, 2013b) and Weijers et al. (2012). The model layout as developed in the WEST® platform is provided in the Annex, Figure 10-2.

3.3. Scenario generation

Parson *et al.* (2007) have defined scenarios as "descriptions of potential future conditions developed to inform decision-making under uncertainty". Scenarios about the future are employed to investigate how the system might respond to social, economic and environmental changes. Decision-makers face a big variety of plausible futures, but they "often have limited cognitive bandwidth" (as put by Lempert (2013)) so they need a concise summary of the futures they might face. Scenarios are thus very useful in that respect as they use a small number of plausible values for key planning variables to create storylines for future conditions in a system (Kasprzyk *et al.*, 2013). Key scenario variables (population and precipitation, for example) are then related to variables of the integrated UWS model to study any potential change.

Future projections of climate and population spanning 30 years were used for the Congost case study. The climate projections used are based on the Fifth Assessment Report (AR5) of the Intergovernmental Panel on Climate Change (IPCC) (IPCC, 2014). Population size, economic activity, lifestyle, energy use, land use patterns, technology and climate policy are the main drivers for anthropogenic GHG emissions. Based on these factors the AR5 presents four representative concentration pathways (RCPs) that describe four different 21st century pathways of GHG emissions and are used for developing projections. They include one stringent mitigation scenario (RCP2.6), two intermediate scenarios (RCP4.5 and RCP6.0) and one scenario with very high GHG emissions (RCP8.5). Each RCP has an associated group of models that have been downscaled to the Catalonia region using regionalised information points from observatories in various Spanish regions by the Spanish meteorological agency (AEMET, 2013).

The projected changes to precipitation in Catalonia where then applied on the observed average yearly precipitation and number of rain events of the reference period (1961-1990) in the Congost catchment. This resulted to a total of 27 projections (ten projections for RCP 8.5, three for RCP 6.0, and eleven for RCP 4.5) for each total precipitation and number of rain events. Figure 3-5 summarises the projections for (a) the total annual precipitation, and (b) the total number of rain events. The shaded areas present the range (min to max) of the projections per RCP, the bold lines represent the average estimate per RCP. The projections see the total yearly rainfall remaining similar to current levels (Figure 3-5a), and the total number of rainfall events per year decreasing (Figure 3-5b), intensifying therefore the received rainfall. To develop the rainfall scenarios the R statistical software was employed. Average estimates of the intermediate and extreme projections were extracted. A series of rainfall measurements from the catchment for 2007 was used as the baseline yearly rainfall series. The series was fitted to a gamma distribution, and its scale ($= variance/mean$) and shape ($= (mean/standard\ deviation)^2$) parameters were calculated. The values were then converted into quantiles (value-points dividing the range of the distribution into contiguous intervals with equal probabilities). Using these quantiles, a rain distribution can then be resampled with the scale and shape parameters of the distribution changed to give the projected total rainfall and number of rain events for a year. The simple code used to perform these calculations is now provided in Annex Section 10.1.1. Two rainfall projections were then generated based on moderate and extreme precipitation predictions. The generated rainfall series (displayed in Firgure 3-6 in relation to the 2007 series) were then used as input to the model.

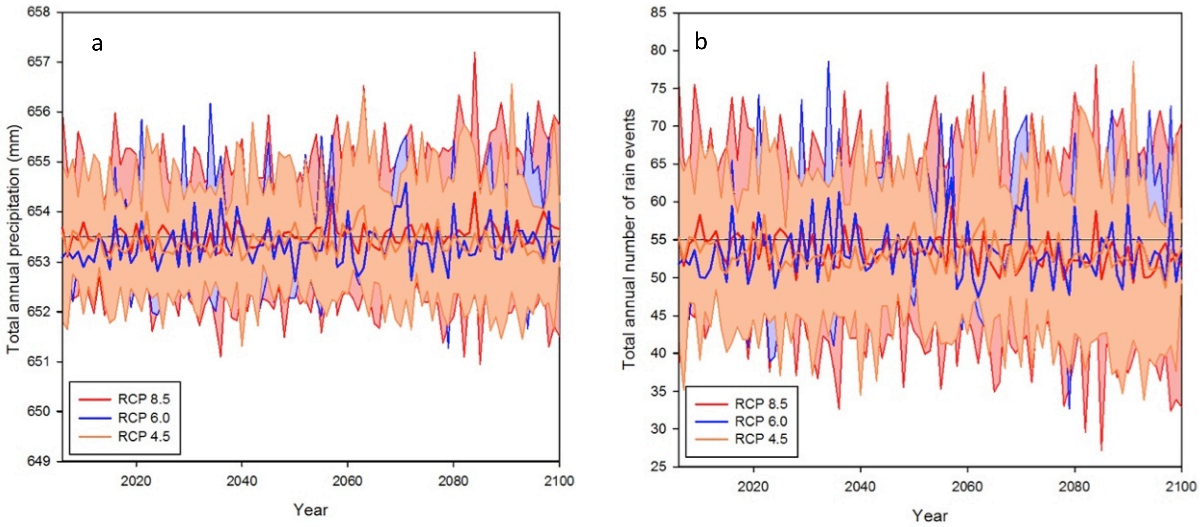


Figure 3-5 - Reference period for projections was 1961-1990. Three representative concentration pathways (RCPs) were used (4.5, 6.0, 8.5). Each RCP has an associated group of models downscaling the projections to Catalonia, Spain (shaded areas). The projections were estimated based on projected % of change from the reference period. (a) is the projected total annual precipitation; and (b) is the total annual number of rain events for the Congost catchment. Projections for Catalonia are publically available through AEMET (www.aemet.es)

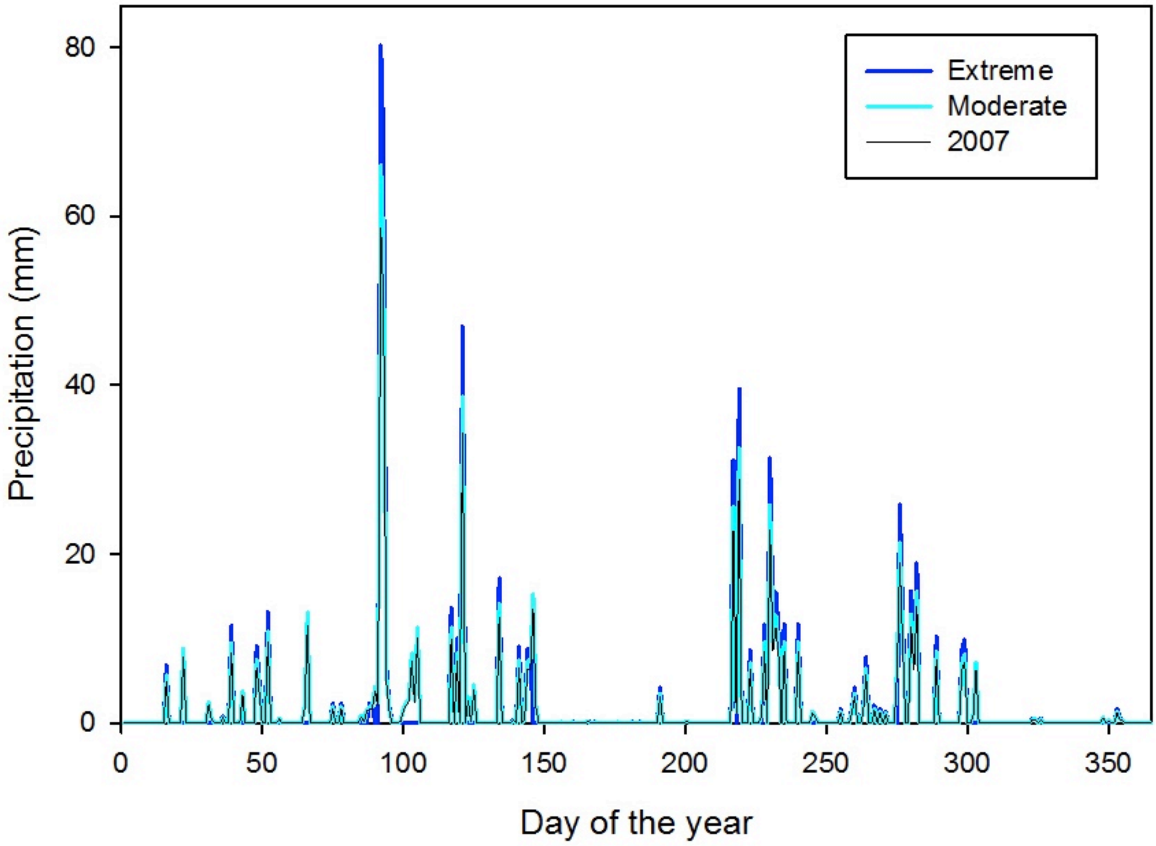


Figure 3-6 - Moderate and extreme precipitation scenarios for the Congost catchment. Developed based on precipitation measurements from 2007.

The Statistical Institute of Catalonia (IDESCAT) (http://www.idescat.cat) has developed population estimates for the municipalities of Granollers, Canovelles and Franqueses del Vallès. Three population projections were then extracted from the 30-year period – low, moderate and high. Projections of future electricity prices were also used, as developed by the Directorate General for Energy of the European Commission (European Commission, 2010). Two projections are included in the report (*2009 Baseline* and *Reference*) determining the development of the EU energy system under population and market trends and policies. The *Reference* scenario is based on the same macroeconomic assumptions as the *2009 Baseline*, but also includes 2009 energy policies and assumes that national targets on renewables and GHG emissions will be achieved by 2020. Regarding the scenarios of population growth and electricity prices, their projected values were input to the model directly.

3.4. Cost-Benefit Analysis (CBA)

CBA is a rational and systematic approach used in public and private decision-making to evaluate whether the long-term benefits of an action outweigh the costs in monetary terms. Based on the principles of CBA, a project should be supported only if the benefits for the gainers are sufficiently greater than the costs for the losers, so they could - in principle - compensate the losers and still be better off (OECD, 2006a). Besides the obvious operational and capital financial expenses, the concept of externalities is central in a CBA. In reality and despite the growing interest in this concept, very few CBA analyses are taking into account environmental externalities as they are difficult to quantify, qualify and assign tangible monetary values to (Fan *et al.*, 2013; Hernández-Sancho *et al.*, 2010). A CBA was used for sustainability assessment during the framework application to the Congost catchment. To take into account environmental externalities, shadow prices developed by Hernández-Sancho *et al.* (2010) were applied; more details on this are provided below (Section 3.6).

3.5. Life Cycle Analysis (LCA)

Life Cycle Analysis (LCA) is a technique to quantify the impacts associated with all the stages of a product, service or process from cradle-to-grave. It is designed to evaluate - and even possibly reduce - the environmental impact for the entire life cycle of said product, service or process (ISO, 2006). There are four main phases in a LCA analysis: *Goal & Scope Definition*, *Inventory Analysis*, *Impact Assessment* and *Interpretation*.

The *Goal & Scope Definition* is the first stage, where the basic decisions that will determine the entire LCA procedure are made. The goal of the analysis has to be defined, regarding the exact question being answered, the target audience and the intended application of the study. The scope has to be specified in terms of its geographical, temporal and technological extend. At this stage, the product, service or process that is under study also needs to be defined in terms of its function, functional unit and reference flows. The next phase is the *Inventory Analysis* where the process (or service or product) system is described by defining its boundaries, designing the flow diagrams with unit processes, collecting the data for each of these processes, performing allocation steps for multifunctional processes and completing the final calculations (Guinée *et al.*, 2001). The product of this step is an inventory table with all the quantified inputs from and outputs to the environment, using the functional unit selected in the previous step. The third step of an LCA study, the *Impact Assessment* stage, comprises compulsory (classification and characterisation) and voluntary (normalisation and weighting) elements. Here, the inventory table produced at the previous step is initially processed and interpreted in terms of environmental impacts. This is done using a defined list of impact categories and characterisation factors to relate the environmental effects to the suitable categories. Normalisation of results is optional, but it serves to calculate "the magnitude of indicator results relative to reference information" (ISO, 2006) by comparing all the environmental impacts on the same scale. Weighting is also an optional step, but it can be used to include societal preferences of the various impact categories and convert and aggregate the results into a single indicator. Finally, the *Interpretation* phase is the stage where all the assumptions made during the course of the analysis and the results are evaluated for their soundness and overall conclusions are formulated. This includes mainly an assessment of the consistency, completeness and sensitivity of the resulting outcomes and the derivation of conclusions, limitations and recommendations stemming from the LCA study (ISO, 2006).

3.5.1. Software and calculation method

A LCA was applied for the purposes of the sustainability assessment part of the framework on the Eindhoven UWS (presented in detail in Chapter 6). The impact assessment phase has been carried out by means of computational software, specifically SimaPro 8.0.3.. SimaPro was developed by PréConsultants (www.pre-sustainability.com) and is a user-friendly tool that combines various impact databases and calculation methods. It allows for the modelling and analysis of complex systems and the estimation of impacts at the midpoint and endpoint categories. The calculation method used, ReCiPe, is a method created by the combination of two other methods, CML and Eco-indicator 99 (Goedkoop *et al.*, 2009). The ReCiPe method differentiates between two levels of impact categories, midpoint and endpoint, and can be used to calculate impacts from three perspectives, individualist, hierarchist and egalitarian. The impact categories used in this analysis are provided in Table 3-6. The estimated environmental impacts were then weighted using shadow prices estimated by de Bruyn *et al.* (2010). Additional details on this application can be found in Chapter 6.

Impact category	**Description**	**Unit**
Climate change (CC)	Release of greenhouse gases causing alteration of global temperature	kg CO_2 eq.
Terrestrial acidification (TA)	Increase in acidity and potential impacts on terrestrial ecosystems	kg SO_2 eq.
Freshwater eutrophication (FE)	Accumulation of nutrients in freshwater aquatic systems	kg P eq.
Marine eutrophication (ME)	Accumulation of nutrients in marine aquatic systems	kg N eq.
Human toxicity (HT)	Toxic effects of chemicals on humans	kg 14DB eq.
Freshwater ecotoxicity (FET)	Toxic effects of chemicals on a freshwater ecosystem	kg 14DB eq.

Table 3-6 - Impact categories taken into account in the LCA of the Eindhoven UWS, with short description and units of equivalence

3.6. Shadow prices for the consideration of externalities

Valuation aims to express the value society puts on a good in monetary terms for purposes of assessment and internalisation. By attaching a value on an emission the estimated environmental damage (or 'cost') can be an indicator of the environmental losses for the local society regarding its present and future emission goals (Howarth *et al.*, 2001; Vos *et al.*, 2007). This a largely vague and ambitious undertaking, mainly hindered by the fact that in many cases no market exists for elements such as water quality or pollution (OECD, 2006a). Various methods for valuation of externalities have been developed in the field of economic theory (Harmelen *et al.*, 2007) with the most common technique appearing in water resources literature being the Contingent Valuation Method (CVM). CVM is considered by many authors as a consolidated method, given its numerous practical applications (Bateman et al., 2006; Birol et al., 2006; del Saz-Salazar et al., 2009). Nonetheless, there is no consensus on the validity of this methodology in the scientific community as a tool for the valuation of environmental goods (Boyle *et al.*, 1994; Diamond and Hausman, 1994; Molinos-Senante *et al.*, 2010; Shabman and Stephenson, 2000). One of the most common criticisms of the CVM and other survey-based methods is that people are often responding to a survey and not a budget constraint, which tends to positively bias their support of abatement costs (Färe *et al.*, 2001).

An alternative is offered through the economic concept of shadow prices. Shadow pricing offers a way to put a value on goods that do not carry a market price. There is a variety of methods to estimate shadow prices for pollutant externalities in scientific literature. They are most often based on an estimation of the damages caused by the release of a pollutant (damage costs) or by calculating the costs associated to its avoidance and removal (abatement costs). The "desktop" shadow price estimation has very low costs, compared to traditional surveying methods and for applications in a policy context, decision-makers have found approaches based on the premise that government represents society more promising (Harmelen *et al.*, 2007). In this manner, shadow prices can provide valuable input into the decision-making process in various ways:

- to determine the possible income that can be gained in case of privatisation of some resources (Molinos-Senante *et al.*, 2010);
- used by authorities as informational means to set rates for the use of environmental services and compare the current rates with the marginal generated revenue (Färe *et al.*, 2001);
- to analyse the social effects of an investment project and compare environmental impacts alongside other monetised costs and benefits by assigning them a monetary value (de Bruyn *et al.*, 2010);
- to help the public recognise the benefits generated as a result of environmental improvement programs (Molinos-Senante *et al.*, 2010);

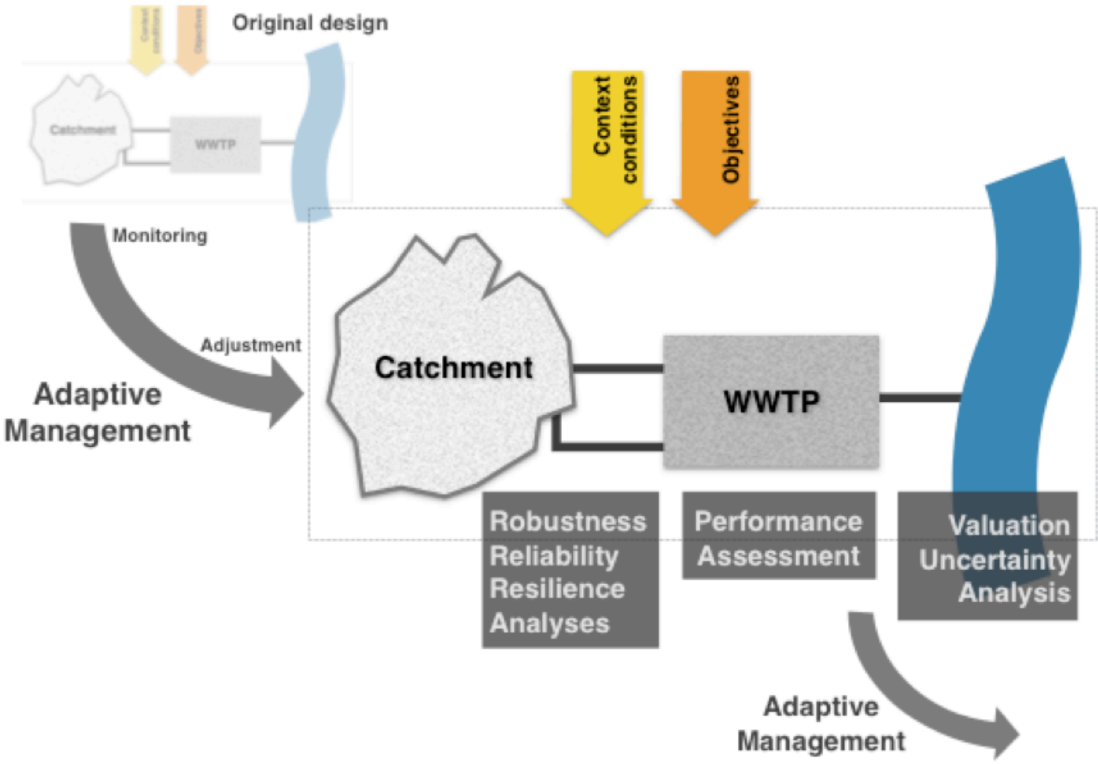
- using shadow price models, provide economists with an additional examination of the estimated values of environmental goods calculated though other models, such as the CVM or capitalisation methods (Färe *et al.*, 2001); and

- to assign a relative weight to each of the environmental impacts identified in environmental analyses, such as LCAs, Environmental Impact Assessments and benchmarking exercises (de Bruyn *et al.*, 2010).

Shadow prices are currently being used as a tool in various decision-making processes, representing hypothetical prices for valuable environmental goods. The concept has also been used in empirical economic analyses including environmental and resource management problems (Liao *et al.*, 2009). In principle, valuation of environmental externalities should take place in every social CBA, providing a means to obtain a comprehensive assessment of all the impacts stemming from a decision (de Bruyn *et al.*, 2010). Empirical applications of the shadow prices method can be found in studies such as Coggins and Swinton (1996), where shadow prices for sulphur dioxide emissions resulting from electrical appliances manufacturing are calculated; McClelland and Horowitz (1999) where they estimated the marginal cost of water pollution abatement for pulp and paper plants; Reig-Martínez *et al.* (2001), where shadow prices of waste produced by the Spanish ceramic industry are calculated; and Van Ha *et al.* (2008) where they estimated shadow prices of pollutants from household paper-recycling units in Vietnam. Shadow prices have also been employed in social CBAs on offshore wind power (Verrips *et al.*, 2005) and to estimate the impact of the European REACH Directive (Witmond *et al.*, 2004). During the development of this thesis shadow prices were applied in a CBA for the inclusion of externalities (Chapter 5) and in a LCA for the weighting of estimated impacts (Chapter 6).

4-6. Results

4. Results I: Adaptive management for sustainable decisions in UWSs in the face of uncertainty: The framework



Redrafted from:

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Adaptive management for sustainable decisions in UWSs in the face of uncertainty – Part 1: The framework

Submitted

4.1. Motivation and Objectives

The Introduction chapter detailed that UWS management needs to be:

- Based on the principles of adaptive management
- Able to incorporate socio-economic, environmental and technical aspects of operation
- Addressing context and valuation uncertainty explicitly and separately

Reviewing the literature, no DST appeared to support UWS decisions while addressing all. The aim of this chapter is to provide a framework to fill this gap.

4.1.1. UWS management in the face of uncertainty

These interrelated factors often have great uncertainty accompanying them, especially in the future, and they represent a problem of formidable complexity for decision-makers. The term 'uncertainty' appears to be interpreted very differently in scientific literature. Even though Walker *et al.* (2003) have provided a typology of uncertainties, a considerable amount of ambiguity still surrounds the terms describing its effects. These definitions are not always in agreement, especially in the literature between different disciplines (social science, engineering, ecology, urban planning, etc.), and are often used interchangeably by authors. The general notion captured by most of the terms is the idea of ***satisficing*** (or not) over the many plausible states a system might be found in (Jim W. Hall et al., 2012). Satisficing refers to the idea of achieving acceptable (satisfactory) outcomes rather than optimal solutions (Stakhiv, 2011). Satisficing is hindered by uncertainty. Based on Herder and Verwater-Lukszo (2006) and Walker *et al.* (2003) we define two types of uncertainty in regard to their origin: context and valuation uncertainty.

4.1.1.1. Context uncertainty

Context factors are related to the socio-political, environmental, financial and technical aspects that are subject to variability. Variability is an intrinsic quality of human and natural systems induced by variations in the social, economic, environmental and technological spheres. As such, this type of uncertainty is deemed inherent and irreducible (Belia *et al.*, 2009; Walker *et al.*, 2003) and great uncertainty is especially expected in future changes of these factors and their combined effects. These context uncertainty factors might include climate, population, market prices and others.

4.1.1.2. Valuation uncertainty

Valuation uncertainty emerges when one attempts to describe and assess a system. If the system is to be modelled, i.e. when interpreting and attempting to represent natural processes using mathematical models, the process involves an intrinsic inaccuracy in its representations and predictions. This type of uncertainty, though always present, is often neglected in evaluations and DSTs based on modelling (Belia *et al*., 2009; Refsgaard *et al.*, 2006). Valuation uncertainty is used here as a broad term to cover all assumptions made regarding the conceptualisation (or model) of the system and can include model structure, model parameters and inputs among others. Other valuation uncertainties arise with choices regarding assessment, for example the criteria and weights considered most appropriate by the decision-makers (Dominguez *et al.*, 2011) or simply the choice of the assessment tool itself. A conceptual visualisation of the two types of uncertainty is presented in Figure 4-1. Figure 4-1(a) presents the defined context; figure 4-1(b) presents variability in the context (context uncertainty); in figure 4-1(c) the same system is under valuation; and finally in figure 4-1(d) the valuation of the system is uncertain.

Some notable decision-support frameworks taking into account these concepts of uncertainty are discussed in the Introduction (Section 1.5). To the best knowledge of the authors, no decision-support framework has appeared yet that explicitly includes both context and valuation uncertainties in its assessment.

4.1.2. Adaptive management for sustainable UWS management

To support in these challenges for UWS sustainable management adequate methods and tools are needed, with a leading role to be played by science and research. Such tools should acknowledge the relevance and unpredictability of the development of the aforementioned factors, the presence of which has not been very apparent in UWS literature (Dominguez and Gujer, 2006). In addition, the tools presented in scientific literature are not always in a meaningful format and language for managers looking to implement them into real-life decisions. Approaches are thus needed to facilitate this communication by integrating and synthesising various metrics and indicators via a structured process into easily understandable and transferable output for decision-makers. In view of the unpredictability of future changes (Section 1.3), authors have asserted that current design and upgrading practice is unsuitable (Brown, 2010; Dominguez and Gujer, 2006); and that, unless current management regimes undergo a transition towards a more adaptive approach, sustainable management of water and wastewater resources cannot be realised (Pahl-Wostl *et al.*, 2007). Notable examples of adaptive management frameworks that have appeared in the literature are discussed in the Introduction (Section 1.5). To the best knowledge of the authors no decision-support framework has appeared in UWS literature for adaptive management after various types of environmental and market changes while supporting a sustainability assessment and various uncertainty analyses.

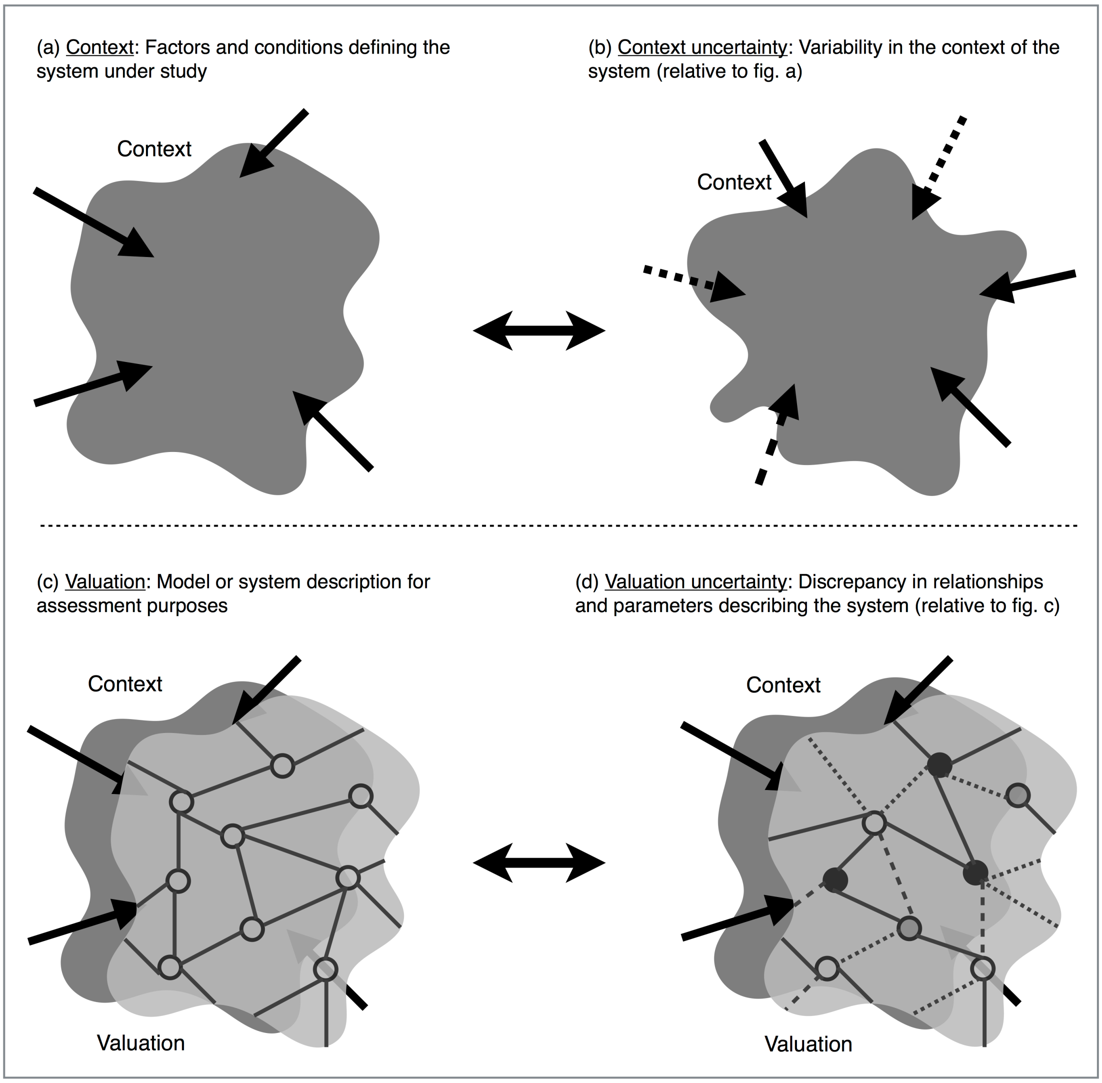


Figure 4-1 - Conceptual visualisation of context and valuation uncertainty, adapted from Walker *et al.* (2003)

Traditionally, the principal purpose of UWSs has been the protection of man and the environment from the adverse effects of wastewater discharges. The intricate and interconnected relationships between society, economy and the environment however further complicate the goals of UWS management. Adaptive management relies strongly on a decision-making process that is participatory and has active stakeholder involvement. Stakeholder participation allows for the inclusion of a wide range of different perspectives rather than decision-making by specialists and experts in isolation – something that is particularly important in early design and planning stages (Pahl-Wostl *et al.*, 2008a). In terms of the adaptive capacity of a system, a broad range of perspectives can facilitate adaptation by recognising new challenges and needs for institutional change (Pahl-Wostl *et al.*, 2008a). For these reasons, the European WFD encourages that *"stakeholders are invited to contribute actively to the process and thus play a role in advising the competent authorities"* (European Council, 2000).

More discussion on these concepts and the identified gaps in literature is provided in the Introduction of this thesis (Chapter 1).

4.1.3. Objectives of this chapter

Looking at the necessity of decision-support frameworks and the lack of a framework encompassing all the aforementioned, the objective of this chapter is to present a structured and coherent conceptual framework to:

- evaluate and compare options for their socio-economic, environmental, and technical performance with sustainable development as an overarching goal;
- address relevant context and valuation uncertainties; and
- support UWS decision-making through an adaptive management approach;

while allowing for flexibility in the criteria and objectives selected by UWS decision-makers.

4.2. Conceptual framework for Adaptive Management of UWSs

Adaptive management can be achieved by a variety of methods as appropriate for the system under study and different authors propose different implementations. Based on the set of steps identified by Westgate *et al.* (2013) we define the following steps for this framework for the adaptive management of the UWS:

i. Identification of current and future context conditions of the system and management objectives by the group of stakeholders (if they differ from the ones previously defined)

ii. Specification of multiple management interventions (solutions), one of which can be 'do nothing' (the current state of operation)

iii. Application of a sustainability assessment of the proposed solutions including:

a. Performance Assessment (given criteria based on management goals)

b. Robustness, Reliability and Resilience Analyses (given present and future context conditions)

c. Valuation Uncertainty Analysis

iv. Deliberation and interpretation of assessment outputs and implementation of selected intervention

v. Monitoring of system response to the selected intervention

vi. Adjustment of management practices in response to the monitoring outcomes

In the proposed conceptual framework these steps have been structured within three principal stages of assessment for UWS decision-making (Figure 4-2).

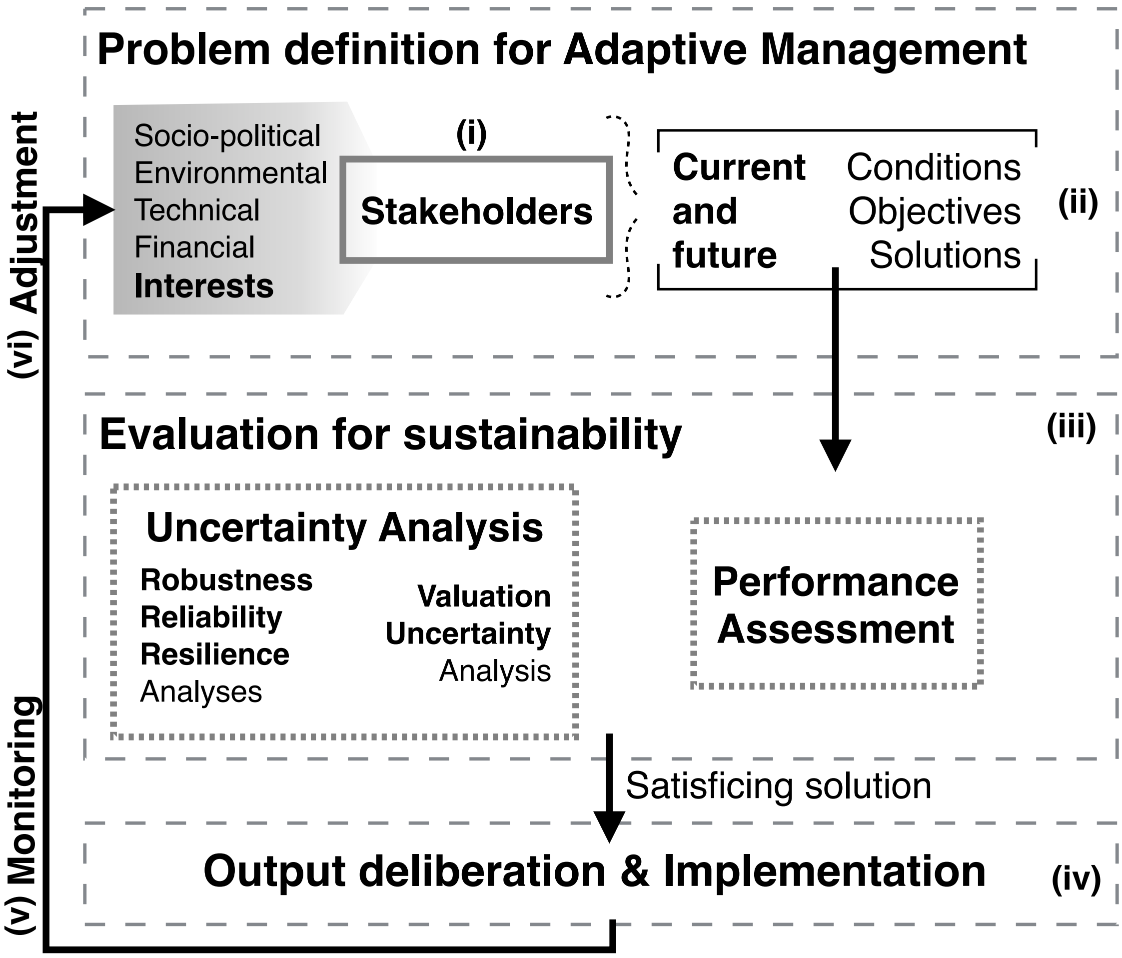


Figure 4-2 - Proposed framework and methods of each stage with steps i-vi of adaptive management

Below we explain in more detail the proposed stages of the framework and the methods that should be applied in each one. Various tools and analyses appear in the literature evaluating different aspects of a decision and providing different indicators. It is not however within the scope of this work to present a review of said tools, but to rather guide in the selection of the most appropriate for each stage of assessment.

4.2.1. *Problem Definition for Adaptive Management*

The *Problem Definition* stage covers the application of steps (i) and (ii). At this stage a group of stakeholders examines the current socio-political, environmental, technical and financial management objectives of the system as well as the current context of the system and its expected future changes (e.g. in climate, markets and society). These are then compared with the context conditions and management objectives that had been occurring during the initial system design. Based on how these have changed, a set of management interventions (solutions) is defined to adapt to the changes one of which can be a "do-nothing" solution.

Socio-political, environmental, technical and financial future changes should be examined at this stage: for example precipitation, temperature, population, energy and water prices, urbanisation, industrial activity, legislation and others as appropriate for the studied system. Scenarios about the future are employed to investigate how the system might respond to future changes and are used here to set up the necessary conditions under which the system should be robust, reliable and resilient. Parson *et al.* (2007) have defined scenarios as *"descriptions of potential future conditions developed to inform decision-making under uncertainty"* and they present a concise summary of the futures decision-makers might face (Lempert, 2013). Scenarios are thus very useful in that respect as they use a small number of plausible values for key planning variables (population and precipitation, for example) to create storylines for future conditions in a system (Kasprzyk *et al.*, 2013). UWS executives have also indicated more willingness to accept scenario-based assessments offering a range of possible outcomes rather than predictive models that are very challenging to build (Borowski and Hare, 2006). In addition, management objectives (criteria for solution) should be set out at this point.

Based on the assessment of how objectives and context conditions have and will change the outcome of this stage is a set of solutions for management intervention. During the next stage (Evaluation for Sustainability) the proposed solutions will be evaluated according to the objectives (Sustainability Assessment), for the determined context conditions (Robustness, Reliability and Resilience Analyses) and for their valuation uncertainty.

4.2.2. Evaluation for Sustainability

This stage is an application of step (iii). The proposed solutions need to conform to the objectives set by the stakeholders under current and future conditions.

4.2.2.1. Performance Assessment

Within the presented framework, the Performance Assessment we propose is performed by accounting for socio-political and environmental costs along with the technical and financial costs of the provision of the service. This includes externalities and opportunity costs (Rogers *et al.*, 1998) - concepts which admittedly are often very difficult to quantify and qualify. Opportunity costs are defined as the value lost from the possible alternative uses of a particular good. Externalities are side effects of an activity that influence the welfare of others. Public appraisal of the sanitation services is of increasing importance, since it can compromise the success of the decision-making process and the resulting outcome (Nancarrow *et al.*, 2009; Scott *et al.*, 2012). Where applicable and relevant, the decision-making process should aim to include them as much as possible. In general, when selecting the most appropriate tool for this assessment the user of the framework should ensure sufficient evaluation of social, economic and environmental impacts. This can be done either through the application of aggregate monetary methods, e.g. CBA; or the application of environmental-impact accounting techniques, e.g. LCA; or combinations of them as the stakeholders deem appropriate.

4.2.2.2. Robustness, Reliability and Resilience Analyses

As already discussed, definitions of uncertainty and the various ways it can affect a system and decision-making are unclear in literature and often contradicting (Chapter 1 and Section 4.1.1). It is not within the scope of this work to produce a comprehensive review of how various terms are used to describe the impacts of uncertainty on UWSs. The reader is directed to Refsgaard *et al.* (2007) for a review of methodologies of uncertainty assessment. Definitions are proposed though based on some supporting literature (Table 4-1).

Reliability is defined as **the ability of the system to fulfil its requirements (i.e. to be satisficing) under its design conditions**, with a common metric being the frequency or probability of failure. The analysis is using the conditions provided by the *Problem Definition* stage to evaluate the reliability of each proposed measure. Regular variability to the operation (storm events, daily and seasonal variability, for example) should be investigated. Resilience often has varying definitions as applied in different fields, particularly in ecology and engineering. UWS management is often occupied with both fields and combining resilience indicators can be difficult. As the framework and the concepts of robustness and reliability used here are mainly involved with engineering (operational) performance, we define ***resilience*** as **the speed with which the system recovers from failure to a satisficing state under its design conditions and once a failure has occurred** and as such it will be used in the subsequent framework application. We believe however that the ecological definition of resilience is also compatible with this conceptual framework and can be applied where relevant. Based on various literature definitions, the term ***robustness*** in this conceptual framework is defined as **the ability of a system to maintain a satisficing state under undesirable (future) conditions**. Undesirable changes (or shock events) are conditions different from those the system was designed for. The Robustness Analysis explores the long-term robustness of the investment project under the established future context conditions, summarised using scenarios developed during the *Problem Definition* stage. Illustration of the types of events that determine robustness, reliability and resilience is provided in Figure 4-3. The occurrence and extent of failure events (non-acceptable performance) diminishes a system's robustness, reliability and resilience: under design conditions and for standard loading, the frequency of failure events defines the reliability of the system – their duration defines its resilience. The extent of failure under a wide range of unexpected, stressful (future) conditions defines its robustness.

	Definition	**Supporting literature**
Reliability	The ability of the system to fulfil its requirements under its design conditions	Butler *et al.* (2014), Fowler *et al.* (2003), Hashimoto *et al.* (1982b), Ryu *et al.* (2012), Zhang *et al.* (2012)
Resilience	The speed with which the system recovers from failure under its design conditions once a failure has occurred	Fowler *et al.* (2003), Hashimoto *et al.* (1982b), Karamouz *et al.* (2003), Nazif and Karamouz (2009), Zhang *et al.* (2012)
Robustness	The ability of a system to maintain a satisficing state under undesirable (future) conditions	Hashimoto *et al.* (1982a), Moody and Brown (2013), Nazif and Karamouz (2009), Scott *et al.* (2012), Zhang *et al.* (2012)

Table 4-1 - Definitions of reliability, resilience and robustness along with supporting literature

4.2.2.3. Valuation Uncertainty Analysis

A Valuation Uncertainty Analysis accompanies this stage safeguarding the decision from both types of uncertainty (context and valuation). The Valuation Uncertainty Analysis explores uncertainties stemming from modelling and valuation assumptions. These include disparities in modelling assumptions made during the procedure of modelling, for example influent fractionation, selected bio-kinetic model parameters or other evaluation parameters (e.g. shadow prices for pollutants). The purpose of the Valuation Uncertainty Analysis in this stage is to assess the uncertainty of achieving the selected outcome. A prior Sensitivity Analysis often accompanies an uncertainty analysis of this type so as to determine the most significant parameters and factors likely to affect the desired outcome.

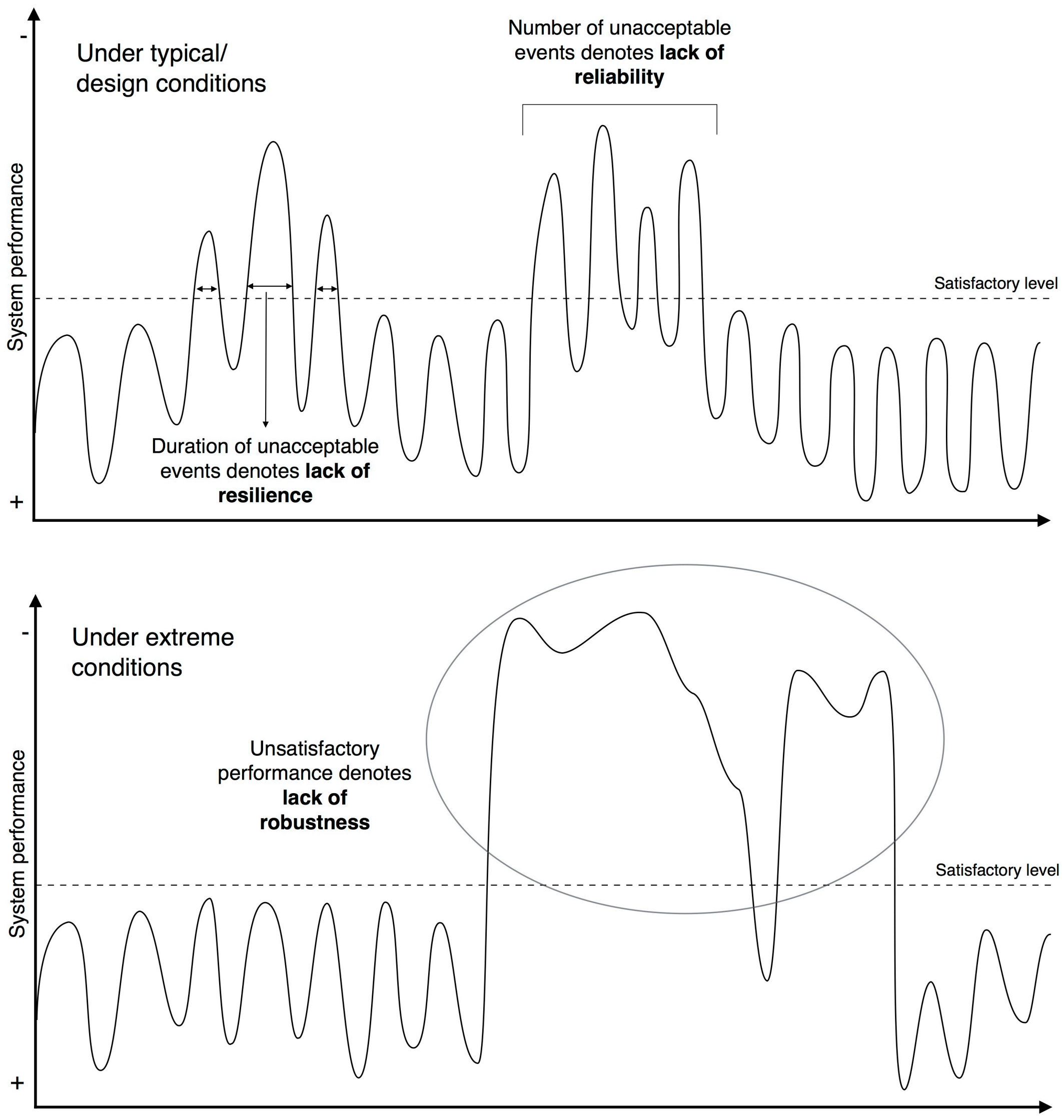


Figure 4-3 - Types of events used to determine the system's robustness, reliability and resilience. Lack of robustness is demonstrated under unexpected conditions. Under typical conditions, the frequency of failure denotes lack of reliability, whereas the length of failure denotes the lack of resilience.

The application of this stage results in a (set of) satisficing solution(s). The solution(s) adhere(s) to the pre-defined objectives and is/are deemed to be sufficiently robust, reliable and resilient despite context uncertainty, i.e. to be satisficing. Proposed solutions that are not satisficing are rejected. By revealing the solutions least affected by uncertain valuation parameters, the results of the Valuation Uncertainty Analysis support the selection of a solution.

4.2.3. *Output deliberation & Implementation*

The outputs of all analyses are then deliberated upon and interpreted. In multi-criteria problems it is common to assign a weight to each criterion describing its importance. These weights are a proxy of the decision-makers' preferences and their application would result in a single "best" solution. There are many methods in literature to collect and apply such weights, based on utility functions, rankings, ratios and others. However no "correct" set of weights exists that is applicable across a range of different procedures, as the obtained weights depend on the procedure used (in this case the framework) (Lahdelma *et al.*, 2014). In the case of multiple decision-makers or stakeholders aggregating conflicting weights into a single weight set poses another complication. In fact, Lahdelma *et al.* (2014) maintain that the overall preferences of a group of stakeholders cannot, in general, be summarised by any single weight set. Due to a variety of reasons it might also not be possible to collect preference information from the decision-makers, e.g. due toi time constraints, decision-maker unwillingness to either state or confine themselves to a preference due to political concerns and others. Consequently in this framework we present how the proposed solutions perform according to the criteria and based on that a reduced set of solutions remains. The final selection is left to the decision-maker or the group of stakeholders.

To ensure coherence with the principles of adaptive management and guarantee effective operation and continuous improvement under the ever-changing conditions of the system, system monitoring (step (vi)) and, accordingly, adjustment of management (step (vii)) need to follow. Large-scale infrastructures with a life-span of decades allow for only few opportunities for adaptation and learning (Pahl-Wostl, 2002; Tillman *et al.*, 2005). Adaptive management is therefore often limited to retrofitting and operational-level decisions (Pahl-Wostl *et al.*, 2008a).

4.3. Discussion

This chapter provided a framework to support decision-making for adaptive management in UWSs. Water managers and policy makers have indicated their need for methods that integrate ecology, economics, social, physical, chemical and biological impacts of water and wastewater (Borowski and Hare, 2006). The proposed framework represents a coherent method for decision-support and allows for the integration of such aspects. The uncertainty surrounding each decision is also addressed by investigating the variability of factors that affect the system's satisficing state (context uncertainty), and including an uncertainty analysis of valuation assumptions (valuation uncertainty).

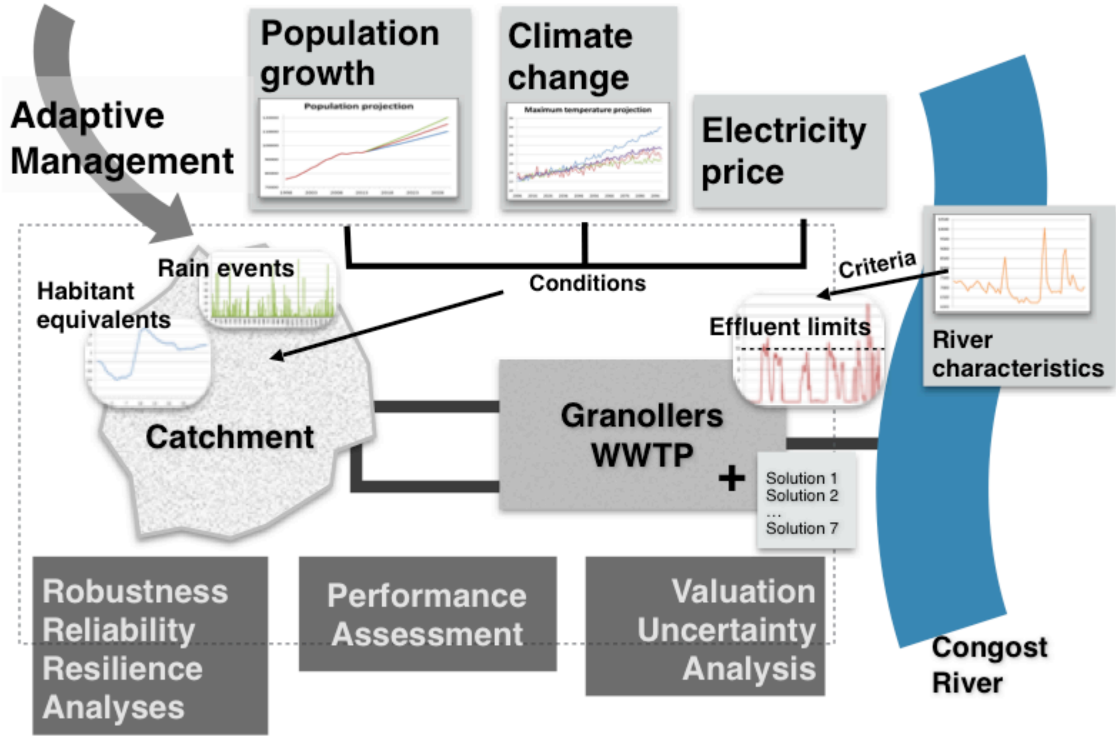
Uncertainty plays an important role in the framework and is classified in two types based on their origin – context and valuation. To tackle context uncertainty, Robustness, Reliability and Resilience Analyses are assessing the response of each proposed solution in the face of present and future variability in environmental, social and other conditions that might affect the efficacy of an UWS. The Valuation Uncertainty Analysis is also performed to assess the effect valuation assumptions have to the estimated performance of each solution.

The concepts of robustness, reliability and resilience are often not explicitly defined and applied in water and wastewater management, mainly due to the general ambiguity of the terms. Many authors neglect to explicitly define them and often use them interchangeably. A search through the literature of water and wastewater management has not revealed any studies that explicitly define and investigate all these concepts for a system. On this basis, we consider this a novelty of the presented framework. An additional novelty of this framework is the combination of these concepts with valuation uncertainty, with their intrinsic difference clearly defined and separated. Evaluating both in the same framework and also separately facilitates the process of tackling and adapting to each type, as the methods and procedures for each one differ.

A concept not addressed in this framework is system vulnerability. Vulnerability refers to the ability of the system (or lack thereof) to minimise failure, i.e. the extent of a failure once it occurs in a system. It is a concept closely related to reliability (frequency of failure) and especially to resilience (speed of recovery from failure). It was therefore deemed redundant to include all three in the framework, in addition to the robustness and valuation uncertainty analyses. Additional analyses can be added should there be an interest of exploring system vulnerability, as the framework is laid out with great flexibility as already discussed in this chapter. Fowler *et al.* (2003) and others have provided metrics of vulnerability that are compatible with the other metrics of the framework and can be included in the assessment.

To the best knowledge of the authors no decision-support framework has appeared in the literature that evaluates planning and intervention decisions for UWSs through the lens of adaptive management. The few notable examples (Brooks, 2003; Grayman *et al.*, 2010; Pahl-Wostl *et al.*, 2008b; Wang *et al.*, 2011) focus on adaptation to climate change impacts or do not allow for other types of uncertainty or sustainability analyses. For this, the flexibility of this framework with respect to the tool selection plays an important role. Flexibility has also been identified as an important factor in the utilisation of DSTs in real-world problems. Tools should be flexible enough in order to meet the users' requirements and to allow them to be used however the users see fit, both in terms of objectives but also in terms of authority and expertise (McIntosh *et al.*, 2011). The decision-support framework presented in this chapter allows for the application of tools deemed suitable by the user for the purposes of sustainability and uncertainty analyses, as long as they conform to the adaptive management principles of the framework.

5. RESULTS II: ADAPTIVE MANAGEMENT FOR SUSTAINABLE DECISIONS IN UWSS IN THE FACE OF UNCERTAINTY: A CASE STUDY



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Adaptive management for sustainable decisions in UWSs in the face of uncertainty – Part 2: Application on UWS upgrading

Submitted

5.1. Motivation and Objectives

In much of the industrialised world urban water and wastewater infrastructure was built during the first half of the 20th century and replacement work has often been neglected (OECD, 2006b). Upgrade, replacement and modernisation of existing systems is overdue; they are deteriorating and in doing so they are creating problems (Global Water Intelligence, 2014; OECD, 2006b). At the same time, most of these systems have not been constructed for the same purposes. As has been illustrated in the Introduction, the primary concerns of UWSs have evolved through the years. The more we advance our understanding of the complicated interconnected relationships between society and environment, the goals of water and wastewater management become more complex and multifaceted. There is a need for assessments that integrate the various facets of UWS management to support decision-making for present and future challenges.

A conceptual framework for adaptive management of UWSs has been presented Chapter 4. The framework represents a decision-support method to support planning through an adaptive management approach. It focuses on incorporating methods and metrics for socio-political, environmental, technical and financial assessment of the UWS. The framework also incorporates the investigation of the factors and parameters that affect the efficacy of the system (context uncertainty), and includes an uncertainty analysis of valuation assumptions (valuation uncertainty). To demonstrate its utility the framework is applied in this chapter for the adaptive management of a specific UWS, as an example of a common situation in most parts of the industrialised world.

5.2. Materials and Methods

5.2.1. Case study

In Europe, with the establishment of the EU WFD regional water authorities of member states are required to develop and publish RBMPs to improve or maintain the ecological state of all water bodies for each river basin district. Accordingly, ACA has set out a RBMP for each river basin, including a set of measures to be implemented, with emphasis on the most threatened rivers of each river basin. Among them is the river basin of the Congost River, a tributary of the Besòs River with a typical Mediterranean hydrological pattern and significant rainfall variability. The Congost River receives the discharges of five municipalities: La Garriga and and L'Ametlla del Vallès, with their own sewer system and WWTP, and Les Franqueses del Vallès, Canovelles and Granollers, which all share the sewer system and the

WWTP (Granollers WWTP). There is high industrial activity in the area, all of which is connected to the sewer systems. The major part of the network of both sewer systems is combined (jointly collecting wastewater and rainfall runoff in the same conduits). There is also a bypass from where WWTP influent above capacity overflows to the river. During the RBMP development the mean NH_4^+ concentration in the Congost River was 26.95 g/m^3, the highest measured concentration in the lower Besòs River basin (Devesa, 2006). Interventions in the Congost catchment were therefore decided in order to improve the river water quality. More details on catchment characteristics are provided in Chapter 3.

In this chapter we apply the conceptual framework presented in Chapter 4 in an attempt to replicate the decision-making process in identifying the most appropriate solution for the intervention. We aim to demonstrate its applicability and utility for real problems faced by UWS decision-makers through the lens of adaptive management.

5.2.2. Conceptual Framework

In the first stage of the framework (Figure 5-1), *Problem Definition for Adaptive Management*, the current management objectives of the system are set out by a group of relevant stakeholders, in this case to improve river water quality. The current context of the system and its expected future conditions are also investigated (population growth, climate change, energy prices) (Section 5.2.4). They are compared with the context conditions and management objectives during the initial system design and, based on how they have changed, a set of management interventions (solutions) is proposed, in this case WWTP reactor volume extensions. The evaluations are conducted using a detailed model of the Granollers UWWS (Section 5.2.3). Then, during the *Evaluation for Sustainability* stage a rigorous assessment of how the system is expected to respond to the proposed solutions is performed. This stage includes a Performance Assessment (given management objectives), Robustness, Reliability and Resilience Analyses (given context conditions), and a Valuation Uncertainty Analysis (given uncertain valuation assumptions) (Section 5.2.5). Finally, during the *Output Deliberation and Implementation* stage the stakeholder group evaluates the assessments' outputs for the measure to be implemented. Following the principles of adaptive management, the system will then be continuously monitored and adjusted as necessary by re-applying the presented stages. The framework has been set out to facilitate various types of administration settings and decisions. It is adaptable in terms of being applied insofar as the basic necessary conditions are fulfilled for sustainable decision-making: taking into account current and future social, economic and environmental impacts; and sufficiently incorporating context and valuation uncertainties (Chapter 4).

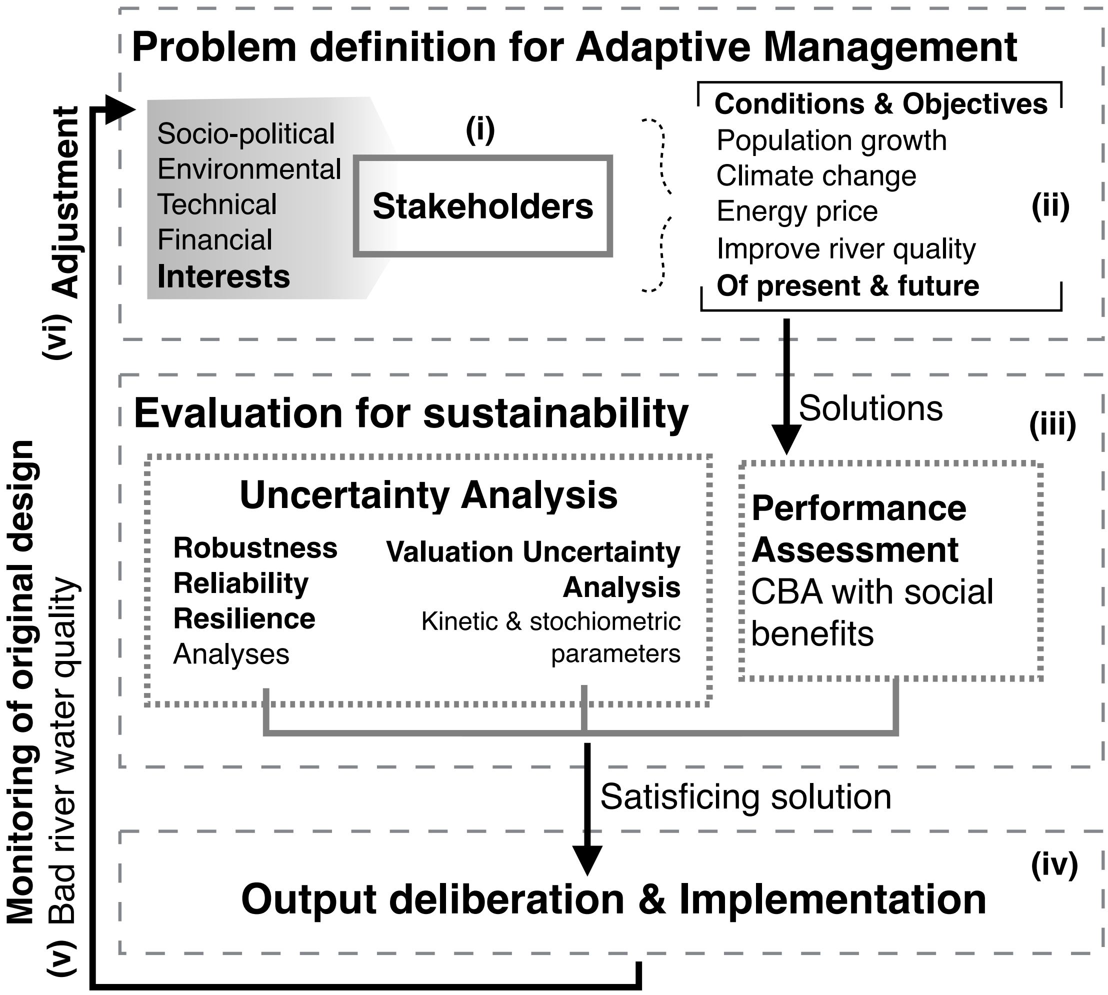


Figure 5-1 - Conceptual framework as presented in Chapter 4 and applied to the adaptive management of this UWS

5.2.3. Modelling and data for calibration

A mathematical model has been developed to describe the Granollers UWS. The model was developed using the modelling software WEST® (www.mikebydhi.com), as an integration of three separated models for the catchment, WWTP and river. More details on model development and calibration are provided in Chapter 3.

5.2.4. Future conditions

Future scenarios were developed based on projections of rainfall intensity, population growth and electricity prices. In the case of rainfall intensity projections, three RCPs (4.5, 6.0, 8.5) were used, as presented in the Fifth Assessment Report (AR5) of the IPCC (IPCC, 2014). The projections have been downscaled to Catalonia using various models by the

Spanish meteorological agency (AEMET, 2013). The projections were used to obtain estimates of the future evolution of total yearly rainfall, number of rainfall events (rainfall intensification) and yearly temperature. The regionalised projections see the total yearly rainfall levels approximately equal to current levels, but the total number of events is expected to decrease. Figure 3-3 (a & b) of Chapter 3 summarises all the rainfall projections for the area. Two rainfall projections were generated based on moderate and extreme precipitation predictions. Three population projections were extracted from the 30-year period – low, moderate and high. Two electricity price projections were also used. Section 3.3 of Chapter 3 details the development of the scenarios used as presented and applied in Section 5.3.1.

5.2.5. Tools and indicators used in evaluation

To assess the performance of each of the proposed solutions the framework makes use of several system analysis tools and indicators presented in this section.

5.2.5.1. Cost-Benefit Analysis (CBA)

CBA is a rational and systematic approach used in public and private decision-making to evaluate whether the long-term benefits of an action outweigh its costs in monetary terms. Besides the obvious operational and capital expenses, the concept of externalities is central in a CBA. The valuation of environmental externalities (such as an improved river water quality) aims to express the value society puts on said quality in monetary terms for purposes of assessment and internalisation. However this is a largely ambiguous undertaking, mainly hindered by the fact that no market exists for elements such as water quality or pollution (OECD, 2006a). A promising approach to this issue is through the use of so-called **shadow prices**. Shadow prices are constructed proxy prices for externalities for which real market prices do not exist, such as emissions, pollution, environmental impacts and environmental quality (Hernández-Sancho *et al.*, 2010; Molinos-Senante *et al.*, 2010). The main externality in the context of UWSs is the detrimental effects resulting from the release of pollutants in a receiving water body (OECD, 2010). Authors have used the valuation methodology of distance function to estimate the shadow prices of the pollutants released into the receiving water body and therefore estimate the avoided cost provided by their removal (Hernández-Sancho *et al.*, 2010; Molinos-Senante *et al.*, 2010). The employment of this method to include externalities allows for the demonstration of the social and environmental benefits of each of the proposed solutions. This can be useful especially in cases where public expenditure needs to be justified – as it was for this case study. Typically, net present value (NPV) after 20 years of operation higher than a reference scenario is used as indicator of socio-economic benefits. NPV is defined as the net sum of all future net benefits (i.e. benefits minus costs) after they have been discounted to their “present value”. Based on the premise that money available at the present time is worth more than money available in the future, future cash flows have to be discounted to more accurately represent their value at the present time. This is usually done by use of discount rates, which vary and depend on the state of the economy and other factors. The NPV is given by the formula:

$$NPV = \sum_{T=0}^{N} \frac{Benefits_T - Costs_T}{(1+i)^T} \quad (eq. 1)$$

where N is the total number of periods (most commonly years), T is the time period (most commonly a year), i is the discount rate (in %s), and $Benefits_T$ and $Costs_T$ are the benefits and costs being accounted for during a given time period T.

A CBA was applied during the second stage of framework application as a performance assessment of the system.

5.2.5.2. Reliability Analysis

Reliability is the ability of a system to fulfil its requirements under its design conditions (Chapter 4). Many indicators appear in the literature. In this application we apply the following indicator of reliability, presented by Fowler *et al.* (2003) and Hashimoto *et al.* (1982b) using effluent TN as the metric of system performance:

$$RL_C = Prob\{X_t \in S\} \quad (eq. 2)$$

where X_t is the measured concentration, C is the legal limit and

$$if\ X_t \leq C\ then\ \ X_t \in S, else\ X_t \in U$$

RL_C is therefore the probability that the performance of a system at time t yields a concentration X_t that is below the required limit C. When the concentration is below the effluent limit, then it is deemed to be satisfactory (success) and $X_t \in S$, otherwise it is unsatisfactory (failure) and $X_t \in U$. This indicator indicates the probability that at a given moment (t) the operation is satisfactory.

5.2.5.3. Resilience Analysis

As already discussed in Chapter 4, resilience is a slightly more complicated concept to address in the realm of UWS management as it can pertain to both engineering and ecological resilience. As the rest of the framework primarily focuses on the engineering aspects of performance, resilience is defined here as the speed with which the (engineered) system recovers from a failure under its design conditions. The indicator used in this application has been presented by Fowler *et al.* (2003) and Hashimoto *et al.* (1982b), with effluent TN as the metric of system performance:

$$RS_C = \frac{Prob\{X_t \in U \ \ and \ X_{t+1} \in S\}}{Prob\{X_t \in U\}} = Prob\{X_{t+1} \in S | X_t \in U\} \quad (eq. 3)$$

RS_C is therefore the probability that given a failure of X_t (as metric) at time t (i.e. $X_t \in U$), at the next time-step $(t+1)$ X will be satisfactory ($X_{t+1} \in S$). In other words, it denotes the probability of the system recovering within a defined time period.

5.2.5.4. Robustness Analysis

Robustness is defined as the ability of a system to withstand undesirable changes, maintaining its original state (Chapter 4), with undesirable changes (or shock events) being conditions different than those the system was designed for. Many indicators of system robustness have appeared in literature (Berggren, 2008; Moody and Brown, 2013), but they vary according to the evaluated conditions and the performance indicator used. In this study we apply a relative indicator for robustness developed by Hashimoto *et al.* (1982a) with NPV (including environmental externalities) used as the metric of system performance. By defining robustness in terms of economic performance the indicator can complement and be combined with other economic analyses, such as CBA (Moody and Brown, 2013). In addition, operational parameters can be sensitive to future or shock conditions and this may not necessarily involve large financial or externality costs (Hashimoto *et al.*, 1982a). The presented indicator can however be applied using other metrics of performance as relevant to each case study (e.g. CSO volume). The relative robustness of each proposed measure is thus measured by:

$$RB_\beta = Prob\{P(q|D) \geq (1-\beta) * L(q)\} \quad (eq. 4)$$

where RB_β is the robustness of a solution, $P(q|D)$ is the performance of solution D under condition q, $L(q)$ is the maximum performance among solutions ('best-case') for a given metric, and β is the fraction of $L(q)$. If the performance metric used is positive (e.g. environmental benefits) then $L(q)$ is the maximum performance among the proposed solutions, in the case where the metric is negative (e.g. total pollutant loads), then $L(q)$ denotes the minimum. β can be any fraction of the 'best-case' solution ($L(q)$) deemed acceptable by the decision-maker (or the public) due to impressions or relative unconcern. This therefore indicates the probability that a proposed solution will be within $(1-\beta)$ of the best possible outcome. For example, if the performance between assessed options was measured in CSO volume, and the best performance resulted in 50,000 m^3 of CSO volume being released per year, then $RB_{10\%}$ of an option denotes the probability it results in a CSO volume of 55,000 m^3.

The indicators and metrics chosen for this application were deemed as most relevant and appropriate for the specific case study and are used as an example of possible evaluation criteria than can be applied. Other indicators for reliability, resilience and robustness exist in literature, however only few are appropriately and distinctly capturing the concept definitions and relate to UWSs. Some more discussion on the terms is provided in Section 4.2.2.2. It is the opinion of the author that the indicators by Fowler et al. (2003), Hashimoto *et al.* (1982a) and Hashimoto *et al.* (1982b) are relevant enough for UWSs, they adhere to the definitions and avoid any double-counting in their estimates. In other applications ad hoc evaluations can be performed for specific metrics and criteria of interest to the decision-makers.

5.2.5.5. Valuation Uncertainty Analysis

A Valuation Uncertainty Analysis was also performed to assess the uncertainty of the proposed solutions. This Valuation Uncertainty Analysis is aimed at assessing discrepancies between valuation (model) parameters and reality. The method for Valuation Uncertainty Analysis adopted in this study makes use of Monte Carlo simulations (McKay, 1988) to propagate uncertainties surrounding the values of model parameters into the model predictions. Table 5-1 lists the kinetic and stoichiometric uncertain parameters assessed at this stage along their probability density functions (PDFs) and ranges. These parameters and statistical properties were selected based on previous studies (Benedetti, 2006; Benedetti *et al.*, 2008; Rousseau *et al.*, 2001). For each measure, 100 parameter combinations were sampled from the parameter space using Latin Hypercube Sampling (McKay, 1988) to perform the analysis.

Symbol	Full name	Probability density function (PDF)	Mean	Min.	Max.	Standard deviation
f_{S_F}	Fraction of soluble COD becoming fermentable substrate	Triangular	0.375	0.3	0.45	–
f_{X_S}	Fraction of particulate COD becoming slowly biodegradable particulate matter	Triangular	0.69	0.552	0.828	–
μ_H	Maximum growth rate of heterotrophs	Normal	6	4.8	7.2	0.4
μ_{AUT}	Maximum growth rate of autotrophs	Normal	1	0.8	1.2	0.067
b_H	Rate constant for lysis and decay	Uniform	0.4	0.2	0.6	–
b_{AUT}	Rate constant for decay of autotrophs	Uniform	0.15	0.075	0.225	–
η_{NO3_Hyd}	Anoxic hydrolysis reduction factor	Triangular	0.6	0.48	0.72	–
η_{NO3_Het}	Reduction factor for denitrification	Triangular	0.8	0.64	0.96	–
K_{O_A}	Half saturation constant for oxygen of autotrophs	Triangular	0.5	0.25	0.75	–
$\eta_{NO3\_Het\_d}$	Anoxic decay rate reduction factor for heterotrophs	Triangular	0.5	0.4	0.6	–
$\eta_{NO3\_Aut\_d}$	Anoxic decay rate reduction factor for autotrophs	Triangular	0.33	0.264	0.396	–

Table 5-1 - Uncertain parameters studied in the VUA and their respective statistical properties

5.3. RESULTS

5.3.1. PROBLEM DEFINITION FOR ADAPTIVE MANAGEMENT

The Congost catchment falls under the jurisdiction of CDCB. For the RBMP development, the CDCRB had to prioritise measures in the Congost catchment through a series of stakeholder meetings involving the local municipalities and CDCRB experts. The bad river water quality was attributed to the fact that about 60% of the river flow was WWTP effluent and that the main WWTP of the catchment (Granollers WWTP), with a conventional activated sludge system, was not performing any N removal at the time due to its limited capacity. In addition, the local population had been steadily increasing pressure on the sanitation system. For these reasons, the stakeholder group decided on upgrades to the Granollers WWTP. The proposed upgrades were evaluated under expected future conditions with effluent quality, volumes of bypass overflows and economic benefits as main metrics of performance.

Using expected future changes in population and precipitation in the area for the next 30 years and expected future electricity prices (i.e. changes in the context of the system), 12 future scenarios were developed and are presented in Table 5-2. The applied conditions are assumed to be independent from each other. The duration of each scenario is one year. The moderate and extreme rainfall scenarios for the catchment are presented in Figure 3-4 of Chapter 3. The developed scenarios will be used as future conditions to evaluate the UWS during the following steps of the framework. The assumed probabilities of each projection occurring are given in parentheses. The probability of each of the 12 scenarios occurring is the joint probability of all three (population, rainfall and electricity price).

Scenario	Population (P.E.)	Number of rain events	Electricity price (€/MWh)
Current	*95,000*	*71*	*124*
A.1 (a/b) A.2 (a/b)	**A:** Low – 110,125 (0.25)	**1:** Moderate – 71 (0.6) **2:** Extreme – 48 (0.4)	**a:** Moderate – 139 (0.6) **b:** Extreme – 159 (0.4)
B.1 (a/b) B.2 (a/b)	**B:** Moderate – 115,432 (0.5)	**1:** Moderate – 71 (0.6) **2:** Extreme – 48 (0.4)	**a:** Moderate – 139 (0.6) **b:** Extreme – 159 (0.4)
C.1 (a/b) C.2 (a/b)	**C:** High – 120,210 (0.25)	**1:** Moderate – 71 (0.6) **2:** Extreme – 48 (0.4)	**a:** Moderate – 139 (0.6) **b:** Extreme – 159 (0.4)

Table 5-2 - Future scenarios for the Congost catchment. First column indicates the scenario code (e.g. B.2.b is scenario with moderate population growth, extreme rainfall intensification and extreme electricity price growth). Values in parentheses next to each projection indicate the assigned probability of each projection. Number of rain events refers to rain events of precipitation >1mm.

Size	Anoxic (m^3)	Aerobic (m^3)	Hydraulic capacity (m^3/d)
Base-case (2007)	*650*	*6500*	*25000*
S. Min	2070	5330	35520
S.1	2320	5980	39840
S.2	2800	7210	48050
S.3	3540	9100	60670
S.4	4020	10090	68930
S.5	5000	12870	85780
S. Max	5740	14760	98400

Table 5-3 - Proposed sizes (solutions) for assessment. The base-case denotes the reactor sizes already installed at the time of the evaluation (2007).

Based on the new management objectives (to improve river water quality) and seeing the expected future conditions, the following size extensions (solutions) are proposed for upgrading the Granollers WWTP (Table 5-3). Design guidelines for activated sludge removal of organic carbon and nitrogen by Metcalf & Eddy (Tchobanoglous *et al.*, 2003) were applied to obtain design volumes. Design guidelines of the activated sludge process, the standard wastewater treatment technology, consist of a set of equations that are computed in a sequential manner and, using a set of design inputs, result in a set of design outputs (Flores-Alsina *et al.*, 2012). Influent characteristics, operational settings, safety factors, kinetic and stoichiometric parameters and effluent requirements make up the design inputs. When applied, the equations produce design volumes for aerobic, anoxic and anaerobic reactors, DO demand, internal and external recycle flow-rates, settling areas and dosage of chemicals.

5.3.2. *Evaluation for Sustainability*

Using the outcomes of the *Problem Definition* stage (criteria, conditions -in the form of scenarios- and proposed solutions) the *Evaluation for Sustainability* step is performed to evaluate the solutions. The selected solution must perform well during the Performance Assessment, and be robust against expected future scenarios, reliable and resilient in its operation in spite of the system's inherent variability.

5.3.2.1. Performance Assessment

A CBA including social benefits was used for the Performance Assessment. It allowed for the assessment of the socio-economic benefits of the proposed investment (solution) on the Granollers WWTP. The applied shadow price values were estimated by Hernández-Sancho *et al.* (2010) for the nearby region of Valencia: 0.098 €/kg COD, 0.005 €/kg TSS and 16.353 €/kg TN. The shadow prices were used to monetize the environmental impacts of released bypass overflows and the avoided impacts (benefits) from improved nutrient removal. Investment, operation and maintenance costs were accounted for in the CBA, using formulas based on literature and expert opinion that are provided in Table 5-4.

Table 5-5 provides a summary of the performance of the proposed solutions under the moderate scenario (B.1.a). The NPV after 20 years of operation is provided for each, along with the yearly bypass overflow volume and the TN concentration at the effluent, averaged over a period of six months. Environmental benefits (nutrient removal) and damages (bypass overflows) are included in the NPV. The estimated benefits brought about by the improved performance seem to far outweigh the costs for the local society after 20 years – indicated by the NPV – justifying therefore the investment in terms of social benefits. This is due to the fact that the increased sizes allow for improved nutrient removal and thus increased environmental benefits. Detailed results of NPV, volume of bypass overflows and effluent TN under all 12 scenarios are provided in Table 10-2 of Annex Section 10.1.

5.3.2.1. Robustness, Reliability and Resilience Analyses

Table 5-6 lists the results of the Robustness, Reliability and Resilience Analyses for all proposed solutions. The indicator of robustness, $RB_{5\%}$, indicates the probability that each solution will have an acceptable outcome (within 5% of the best performance) under the future scenarios. The estimated probability is the cumulative probability of all scenarios that do not prevent each solution from achieving an acceptable outcome. The metric of performance used here was the NPV of all measures, including monetised environmental impacts and benefits. NPV was deemed an appropriate robustness indicator as it is able to capture the effects of all three future changes and is in accordance with Hashimoto *et al.* (1982a). Reliability and resilience were assessed at the effluent TN level, 10 mg/L, for the duration of nine months. Reliability and resilience were estimated for the moderate and most probable scenario (B.1.a). RL indicates the probability of a satisfactory effluent (below legal limit) and RS the probability that, given an unsatisfactory state, the system will return to a satisfactory state within the next time-step (in this case half a day). Expectedly, system reliability improves with increased volumes.

Investment costs (formula)		Reference
Reactor cost	10^(0.806*LOG10(35.315*[Volume]/1000)+0.306)*4820.8	Dorr-Oliver Incorporated; DiGregorio, D. *Cost of wastewater treatment processes*; U.S. Department of the Interior - Federal Water Pollution Control Administration: Cincinnati, Ohio, 1968.
Primary treatment operational costs (formulas)		Reference
Maintenance	(0.7126*25*[Influent flowrate]/3)+666	Manual para la implementación de sistemas de depuración en pequeñas poblaciones. Ministerio de medio ambiente y medio rural y marino. Research centers: CEDEX and CENTA. 2010
Other operational costs	0.002*3600*[Influent flowrate]*[Price per kWh]	
Secondary treatment operational costs (formulas)		Reference
Maintenance	(1.5706*[Influent flowrate)+80.654	EDSS-PSARU 2002: DEVELOPMENT AND IMPLEMENTATION OF A DECISION SUPPORT SYSTEM FOR THE SELECTION OF WASTEWATER TREATMENT SYSTEMS FOR COMMUNITIES WITH LESS THAN 2000 EQUVALENT PEOPLE IN CATALONIA. Universities involved: UdG, CSIC, UB, UAB and UPC.
Other operation costs	(309.44*25*[Influent flowrate]/3^(-0.389))*25*[Influent flowrate]/3	
Other		Reference
Discount rate	4.5%	Publically available as provided by the European Commission at: http://ec.europa.eu/competition/state_aid/legislation/reference_rates.html
Electricity price	0.124€/kWh	European Commission, 2010. EU energy trends to 2030 - Update 2009. European Commission, Luxembourg.

Table 5-4 - Investment and operational costs, electricity price and discount rate used in the estimation of the net present value (NPV)

Size	NPV (€)	Bypass overflows (m^3/year)	TN (mg/L)	Investment (€)
Base-case (2007)	83,665,201	540,777	17.52	-
S. Min	99,702,794	219,001	10.30	974,593
S.1	100,765,052	5,511	9.97	1,069,019
S.2	101,868,694	-	9.44	1,243,288
S.3	103,004,552	-	8.86	1,500,559
S.4	103,590,652	-	8.56	1,641,005
S.5	104,171,245	-	8.19	1,983,534
S. Max	104,471,865	-	7.97	2,215,717

Table 5-5 - Net Present Value (NPV), yearly bypass overflow volume and average effluent TN concentrations of keeping the base-case WWTP compared to the proposed solutions. NPV after 20 years of operation including environmental costs and benefits. Context conditions were assumed to be those of the moderate scenario (B.1.a).

Size	$RB_{5\%}$	$RL_{10mg/L}$	$RS_{10mg/L}$
S. Min	0	0.35	0.57
S.1	0.3	0.41	0.60
S.2	0.3	0.53	0.65
S.3	0.3	0.73	0.69
S.4	0.3	0.81	0.68
S.5	0.6	0.90	0.68
S. Max	0.6	0.93	0.68

Table 5-6 - Results of Robustness ($RB_{5\%}$), Reliability (RL) and Resilience (RS) Analyses for all proposed solutions. Values indicate probabilities (0-1).

Figure 5-2 summarises the impacts of all 12 scenarios on the NPV and the effluent TN of the proposed solutions. The biggest relative effects in operation are brought about by the population growth projections (A, B and C in Figure 5-2, as defined in Table 5-2). The effects of an increased electricity price are also visible looking at the NPV (a and b in Figure 5-2, as defined in Table 5-2), with about 1M€ lower expected returns with a higher energy price. The impacts of climate do not seem to be significant with respect to effluent TN and NPV (1 and 2 in Figure 5-2, as defined in Table 5-2). There is a clear positive effect of the increased population on the NPV. This is attributed to the fact that the socio-economic benefits from removal outweigh the operational costs, despite the increase of the latter. Measures S.3, S.4, S.5 and S.Max, perform best for the future effluent limit.

5.3.2.1. Valuation Uncertainty Analysis

Figure 5-3 presents the effects of valuation (model uncertainty) on effluent TN. The Valuation Uncertainty Analysis was performed as described in section 5.2.5 for the parameters and statistical properties listed in Table 5-1.

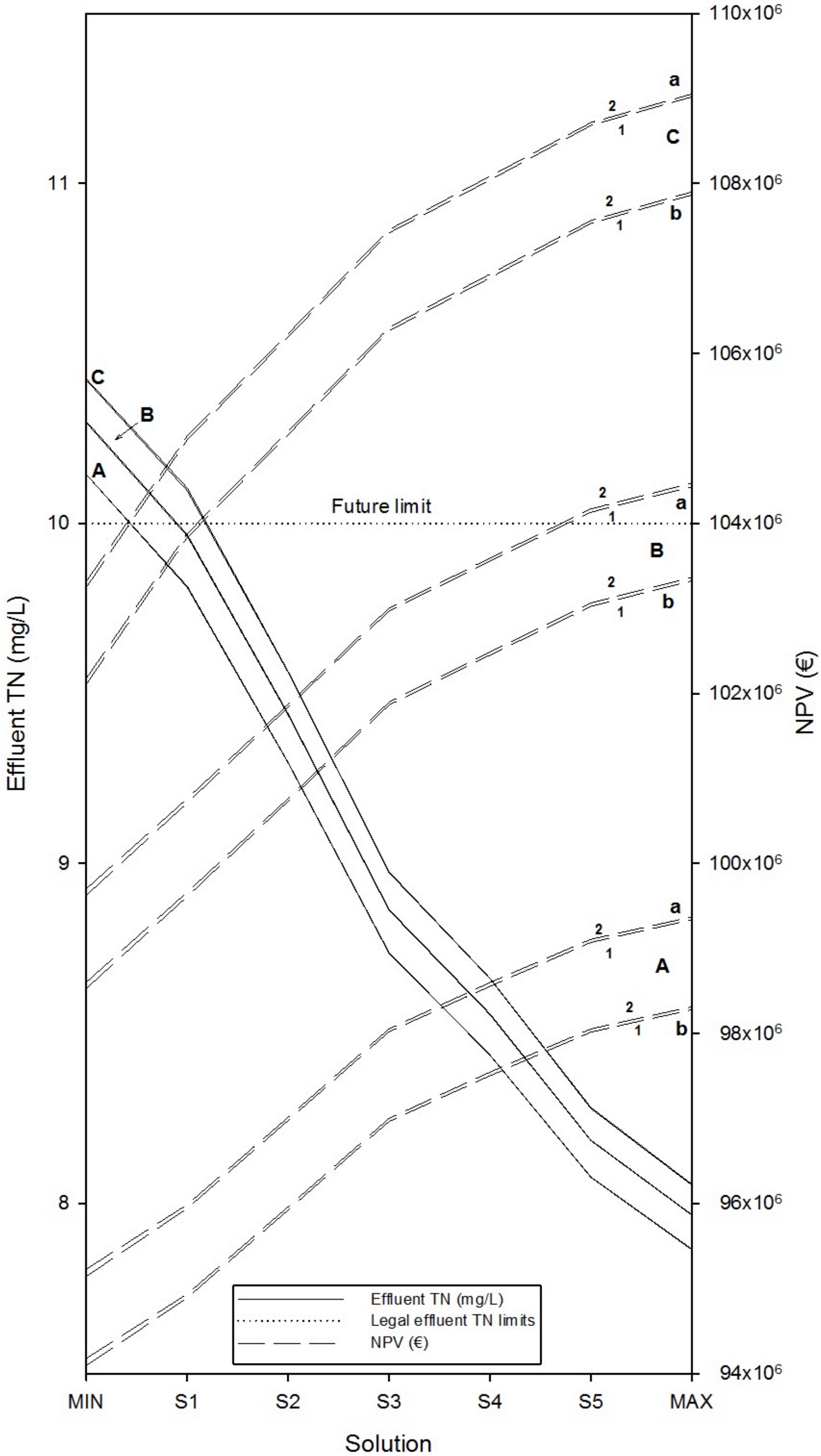


Figure 5-2 - Impacts of all 12 future scenarios on effluent TN and NPV for all proposed solutions. Current and future (with respect to P.E.) legal effluent limits of TN are given. A, B and C represent population projections, a and b electricity price projections, and 1 and 2 climate change projections. Each line represents a projection combination to form a scenario (e.g. A.2.b).

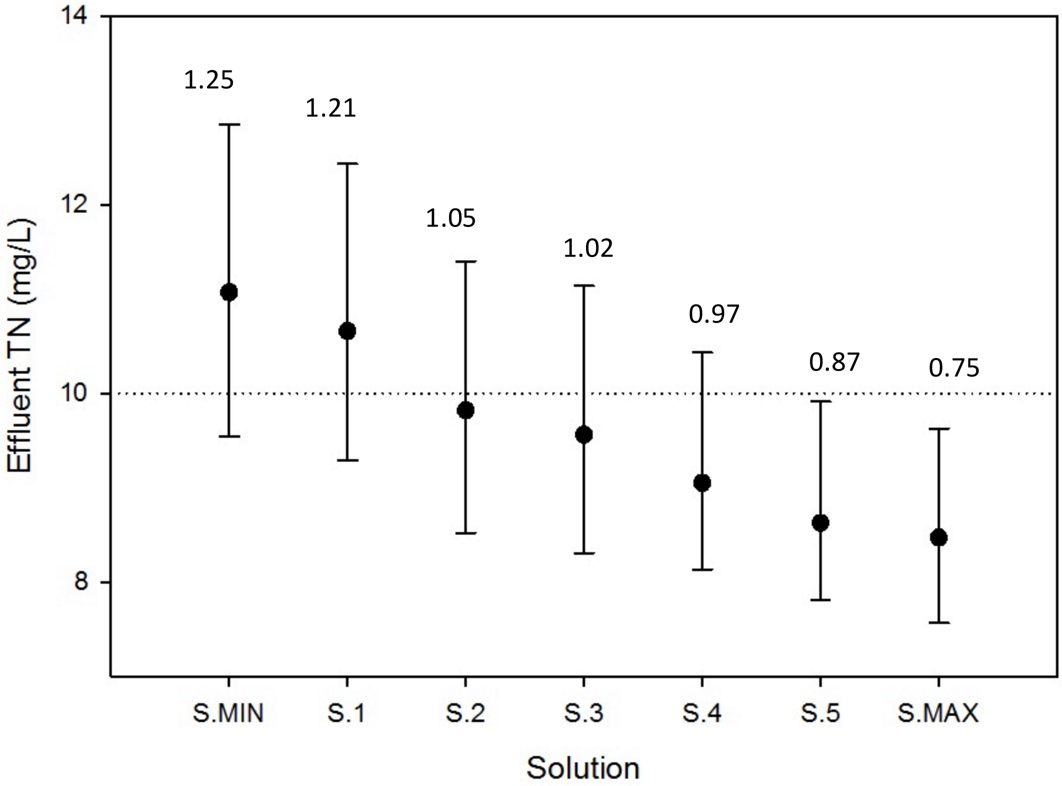


Figure 5-3 - Impacts of valuation uncertainty on effluent TN for all proposed solutions. The error bars indicate the 10th and 90th percentiles. Values above each bar indicate the standard deviation of the set of estimated values.

Even though the assumed uncertainty was the same for all solutions, the larger systems displayed a reduced sensitivity to these variations. Furthermore, S.5 and S.Max remain below the legal threshold despite the assumed model uncertainty.

5.3.3. Output Deliberation & Implementation

The outputs of the analyses are now to be deliberated upon for selection. Figure 5-4 (a and b) summarises the performance and the uncertainty of all assessed alternatives. The relative increase in operational costs brought about by each measure appears to be lesser than the increase caused by the future scenarios. The improvement in effluent TN and bypass overflow volume by the larger sizes is clear, with sizes S.2-S.Max having acceptable effluent quality (under 10 mg/L) and no bypass overflows. Regarding uncertainty, Figure 5-4b indicates the relative uncertainties of each of the estimated effluent concentrations (X_t) for each measure. A bigger segment area is a sign of lower reliability and resilience. A "leap" in relative reliability and resilience can be observed when increasing from S.2 to S.3 (also seen in more detail in Table 5-6) which can also be observed in terms of performance (Figure 5-4a). The selection is now a matter of stakeholder preference according to their objectives and priorities. It is clear however that sizes S.Min and S.1 are inadequate by all metrics of both performance and uncertainty. Measures S.5 and S.Max also clearly outperform all other sizes, ensuring the most robust, reliable and resilient performance (Table 5-6) as well as maintaining an acceptable performance despite valuation uncertainty (Figure 5-3). The biggest relative leap in both performance (Figure 5-2) and total uncertainty (Figure 5-4b) is observed by S.3. Depending on stakeholders' preferences, S.3 might prove to be a satisfactory selection.

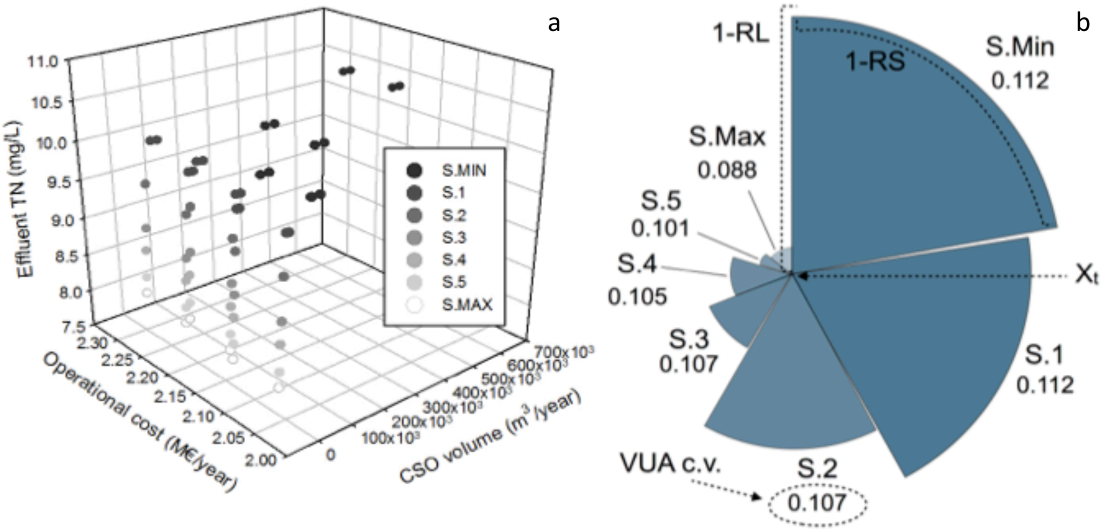


Figure 5-4 - Summarising the assessment of all sizes for their performance (a) and their uncertainty (b). (a): The performance (effluent TN, bypass overflow volume, operational costs) of all measures under all scenarios. (b): Given estimated effluent concentration X_t (as defined in eq. 2): the radius of each segment indicates the lack of reliability (1-RL); the length of the circumference indicates the lack of resilience (1-RS); the value of each segment indicates the coefficient of variation (c.v.) of each measure under valuation uncertainty.

5.4. Discussion

Chapter 4 presented a framework to support in UWS decision-making based on the principles of adaptive management. To demonstrate its real-world applicability and utility the framework is being put into practice in this chapter to support in the decision-making for the adaptive management of an UWS. For this application specific tools and metrics were selected as deemed to be most appropriate and relevant for the decision-makers acting in this case.

UWS managers have often mentioned the need for meaningful tools that can act as supporting material for talking with the public and stakeholders (Borowski and Hare, 2006). Using a monetary indicator to demonstrate social benefits is an easily communicable tool for public decision-making. In this case, during the Performance Assessment, the concept of shadow prices was utilised to demonstrate benefits of the decision for the environment and society. As evident in Table 5-5, the benefits provided to society by the increased capacity for treatment far outweigh the capital and operational costs for providing it. The shadow prices used were estimated by Hernández-Sancho *et al.* (2010) based on abatement costs per pollutant. They are also subject to uncertainty, especially considering future legislation becoming more stringent. It was deemed out of the scope of the application to address these uncertainties in the present study seeing as they are extrinsic to the UWS system. Likewise, the climate and population projections also carry uncertainties in the assumed values that were regarded as secondary to the present application.

Developed scenarios allowed for a more in-depth investigation on the capacity of the system to deal with future changes. The population growth projections appear to affect the UWS operation the most. This is seen both at the effluent TN concentrations (where it has a negative effect), but also in the estimated NPVs where there is an increase. This of course does not imply that population growth would be "better" for the system, but rather that given population growth A, B or C the benefits for society having applied one of the solutions will be such. Future changes in energy prices seem to also affect system performance in terms of costs. Electricity prices are projected to increase in the next 30 years in Europe (by 10-30%) (European Commission, 2010). This naturally will be an additional pressure to the operational equilibrium of the UWS. The effects of rainfall intensification (induced by climate change) on the metrics used are less apparent, seemingly confirming the hypothesis by Hashimoto *et al.* (1982a) that sensitive operational conditions might have little externality costs. This could also be due to the predicted rainfall intensification for this region, which is not as extreme as expected for other regions. Other studies looking into more extreme changes in rainfall intensification (e.g. Astaraie-Imani *et al.* (2012)) have found more significant impacts. These results highlight the importance of considering a wide range of future changes.

Regarding the selection of future projections, population, climate and energy were selected for being independent (at least within the UWS context) so as to avoid overestimating effects on the system. Undoubtedly, other factors such as urbanisation and industrialisation of the area, might affect this particular UWS in the future, but as this was an application of multiple assessments it was deemed unnecessary to study further projections at the present time. Regarding the metrics of system performance applied in the robustness, reliability and resilience analyses, they were selected as being the most relevant to the problem at hand and to serve in the demonstration of the framework. The framework itself has been designed with a wide array of UWS concerns in mind and can be applied with multiple ad hoc metrics and tools.

The results of the application do suggest that larger systems would improve system reliability. It is important to note however that high reliability does not necessarily mean a “good” performance, but rather a more stable operation (Oliveira and Von Sperling, 2008), which is desirable in its own right. WWTP operation can be reliable, i.e. stable in its performance, but not performing “well” which will depend on the set criterion for a good performance, e.g. a legal limit. With respect to resilience, the indicator we used in this demonstration indicated the probability that given a failure (effluent above limit) the operation would return to a desirable state within the determined time-step (0.5 days). In this case the bigger systems did not appear to be necessarily more resilient, even though some improvement was observed (Table 5-6).

The results of the Valuation Uncertainty Analysis displayed a diminishing impact on increasing system volumes (Figure 5-3), despite all solutions being based on the same models and having the same assumed model uncertainty. Compared to the impacts of context uncertainty on effluent TN (Figure 5-2), valuation uncertainty proved to be more significant, especially for the smaller volumes. In the case of an appraisal between solutions modelled in a different manner (e.g. two different treatment technologies), the implicit valuation uncertainty of each will be different and might possibly exacerbate the significance of this type of uncertainty in the comparison. The results of the valuation uncertainty suggest that the estimated performance of the smaller investments is accompanied with larger uncertainty compared to the bigger investments. Should the smaller measures be preferred (e.g. due to their lower costs), stakeholders are aware that there is less confidence in their predicted performance. In other words, when robustness, resilience and reliability are estimated they are predictors of how the system will respond to change; when valuation uncertainty is estimated it is an indicator of the confidence that should be put in said predictions.

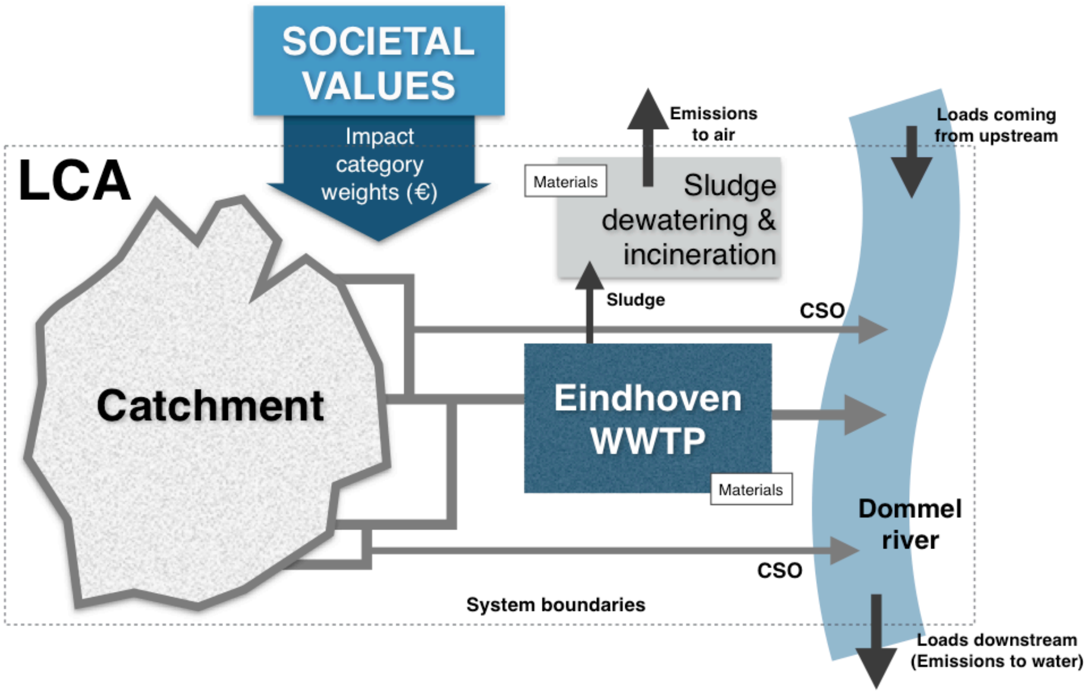
The valuation uncertainty analysis in this application has been admittedly rather limited. The parameters used were kinetic and stoichiometric WWTP parameters, chosen due to the fact that the measures tested were at the WWTP and they are more commonly studied making their uncertainty ranges more accessible and reliable. Besides, the main purpose of this analysis has been demonstration and as such, simplicity in application was preferred. This means that any valuation uncertainty on parameters from other parts of the model has not been accounted for and could potentially affect the results. Future applications taking into account variable interdependencies could also be beneficial. Additional studies including a wider range of future scenarios as well as possible sources of valuation uncertainty would allow confirming these findings for this system and support in the prioritisation of subsequent research efforts.

When the actual decision was made for this system, the WWTP was upgraded with an anoxic reactor of 3248m^3 and an aerobic reactor of 6514m^3, a lesser capacity than the S.2 solution evaluated here. Solutions S.Min and S.1 demonstrated an unsatisfactory performance by all standards of this framework application. Especially when tested for their robustness, reliability and resilience, S.Min and S.1 were shown not to be robust against the expected future changes and to be the least resilient and reliable compared to the other measures. They were also the solutions most sensitive to valuation uncertainty. An upgrade larger than S.1 (such as the one actually implemented) could therefore have an acceptable performance. Looking however at the performance of the other solutions, the results suggest significant improvement on both resilience and reliability as well as effluent quality with the implementation of S.3 and larger sizes (Table 5-6 and Figure 5-2).

On these grounds, we believe that UWS decision-making can benefit from an application of this novel adaptive framework for a more thorough system assessment against uncertainty. With this application we demonstrated that for the expected future conditions and the assumed uncertainty, an extension of a size of at least S.2 would have been more appropriate with even bigger sizes demonstrating larger benefits. That is not to suggest that bigger volumes are always better, as issues arising with the use of bigger volumes (e.g. ease of operation) have not been accounted for. In addition, as the mass of pollutants removed by treatment is finite (in contrast to the costs of construction and operation), as volumes are increased the costs will eventually outweigh the benefits, making thus the investment unjustifiable. This is also suggested in Figure 5-3 where the increase in NPV tends towards a plateau. Having demonstrated increased benefits with larger volumes, the actual construction of additional volumes is still left to WWTP managers' preferences (e.g. done over two lines) so that the estimated environmental benefits remain but more flexibility in operation is allowed for. Qualitative criteria related to the operation of bigger vs. smaller volumes can still be taken into account during stakeholder deliberation and affect the decision. As a final point, environmental benefits were calculated using shadow prices (provided in Section 5.3.2.1) that have been the research output of other studies at a nearby region with similar characteristics. The results of this application could possibly benefit by the use of shadow prices (or other such values) estimated for the catchment at hand as they would more accurately represent environmental benefits in the area and better justify any proposed investments.

To that end, other inputs to this evaluation have their sources in literature and were aimed to be as closely applicable to the case study as possible. For example, the price for electricity and its projections were EU-wide. Spain or Catalonia-specific prices and projections would result in more accurate estimations. Same holds for the discount rate (i) used in the NPV calculation. It was selected to be 4.5%, which is the rate applicable to Spain as reported by the European Commission. Again, if a case-specific study were to be conducted to determine a discount rate according to the nature of the investment and the political and financial climate of the region, the new rate would be possibly more accurate. All this has been however out of the current scope of the study and thus data that was already available and applicable has been used.

6. Results III: Assessing urban wastewater system upgrades using integrated modelling, life cycle analysis and shadow pricing



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6.1. MOTIVATION AND OBJECTIVES

Sustainable development has now been adopted as an overarching goal of all economic and social development by multiple United Nations agencies, individual nations, local governments and corporations (OECD, 2006a). Prerequisites for this are decisions encompassing technical economic, social and environmental considerations (Macleod and Haygarth, 2010). For UWS management this integration can be achieved through the integration of state-of-the-art tools that support decision-making.

Water management agencies are already conducting several studies to ensure the operation of the UWS adheres to these principles (for example the KALLISTO project (Benedetti *et al.*, 2013b; Langeveld *et al.*, 2013a, 2013b; Weijers *et al.*, 2012) by the WdD). These studies have evaluated the cost-effectiveness and the technical performance of various proposed UWS upgrades, often looking at the ecological improvement of the receiving water body. An important tool in these analyses has been integrated modelling, encompassing the whole UWS and receiving medium and enabling dynamic assessments of the systems at hand (Benedetti *et al.*, 2013b; Lluís Corominas *et al.*, 2013; Devesa *et al.*, 2009; Langeveld *et al.*, 2013a, 2013b; Weijers *et al.*, 2012). Despite the great strides made during these assessments, some aspects often remain unaddressed – primarily global and long-term environmental impacts.

LCA is a technique to quantify the impacts associated with all the stages of a product, service or process from cradle-to-grave, in order to evaluate the environmental impact of its entire life cycle (ISO, 2006). The application of LCA allows for the assessment of secondary, global impacts brought about by the proposed measures for UWS upgrading. There have been multiple examples of the LCA method being applied to the UWS (Ll. Corominas *et al.*, 2013; Loubet *et al.*, 2014), several of which have expanded on the conventional WWTP boundaries to include other parts of the UWS (for example, El-Sayed Mohamed Mahgoub *et al.* (2010), Meneses *et al.* (2010), Morera *et al.* (2015)). To the best knowledge of the authors, no LCA on the UWS has yet employed a deterministic integrated model taking into account hydraulics as well as biochemical processes of the system (in this case WWTP and receiving water body). For this purpose we employ an integrated model of the UWS - already used in previous studies (Benedetti *et al.*, 2013b; Langeveld *et al.*, 2013a, 2013b; Weijers *et al.*, 2012) - which has shown to be a powerful tool to analyse and evaluate the proposed measures. This allows for a more integrative analysis as well as studying the effects of climatic and seasonal variations in the influent composition. The river model allows for the consideration of its functions and its capacity to dilute and uptake the discharged loads. Integrated modelling also provides the ability to investigate the dynamic effect of operational changes and upgrades on the assessment of the LCA impact categories.

Weighting is an optional step during a LCA and it can be used to include a prioritisation of the various impact categories and convert and aggregate the results into a single indicator. It is thought to be subjective and has therefore always been a controversial step of the LCA technique as it reflects personal values in the social, ethical and political fields (Fougerit et al., 2012; Huppes and Oers, n.d.; Wu et al., 2005). Despite their subjectivity, these value-choices may be more relevant to the decision-making process as they can simplify and aggregate impacts. Weighting can be either quantitative or qualitative (Guinée *et al.*, 2001; Harmelen *et al.*, 2007) and authors have suggested that methods based on monetary values or the judgement of an expert panel are the most promising (Finnveden, 1999). Currency appears to be a unit that can be easily integrated by decision-makers in the decision process and be contrasted against other indicators (Ahlroth *et al.*, 2011; Harmelen *et al.*, 2007). It is also a unit that is easily understandable and communicable by a wide range of decision-makers (Eldh and Johansson, 2006). Valuation aims to express the value society puts on them in monetary terms for purposes of assessment and internalisation. By attaching a value on an emission the estimated environmental damage (or 'cost') can be an indicator of the environmental losses to the society regarding its present and future emission goals (Howarth *et al.*, 2001; Vos *et al.*, 2007). Shadow pricing offers an interesting approach to the valuation of environmental externalities (in this case pollution), among other purposes (Molinos-Senante *et al.*, 2010; Pearce, 1993). Some commentary on other available valuation methods is offered in Section 3.6, as well as discussion on the advantages of this approach and the various ways it can be used in assessments. Among various other uses, authors have proposed the use of shadow prices to assign relative weights to environmental impacts identified in environmental analyses such as LCA (de Bruyn *et al.*, 2010). The values used in this study were derived from abatement costs related to each environmental impact based on Dutch policy.

The objective of this chapter is to employ a novel approach to assess measures for UWS upgrading, as an alternative to the Performance Assessment methods applied in the previous chapter. The measures are evaluated by integrating a deterministic model with LCA and the application of weights of relative abatement prioritisation to the estimated impacts. The utility of this integrated approach is illustrated with the case study of the Eindhoven

UWS. The boundary of the analysis is extended to include the adjacent river section of the Eindhoven UWS, i.e. the river Dommel, taking into account its dilution and purification capacity. With this new application of LCA and weighting through shadow prices, we present a novel integration of methods to account for sustainable development, compared to traditional approaches. The application presented in this chapter aims to complement the other studies performed by the managing authority in Eindhoven (WdD) to decide on upgrades for the UWS while adhering to the principles of sustainable development.

6.2. CASE STUDY: EINDHOVEN UWS

The studied system is the Eindhoven WWTP and its collection system, located in the southeast of the Netherlands. The Eindhoven WWTP treats the wastewater of 750,000 PE with a design load of 136 g COD/day/PE. The plant has a modified UCT (University Cape Town) configuration for biological COD, N and P removal. Approximately 660,000 m^3 of sludge is produced every year, roughly corresponding to 15,000 tons of dry matter content. To avoid odour nuisances the sludge is transported via a 7 km pipe system to a sludge processing installation near Mierlo. There, the sludge is dewatered and then transported (about 60,000 m^3/year of dewatered sludge) to an incineration facility (SNB) at Moerdijk (approximately 100 km away). Additional details on the Eindhoven UWS can be found in Section 3.1.2. The proposed measures to be evaluated and their principal targets (enhance nutrient removal, reduce CSOs and reduce river DO depletion) are summarised in Table 6-1.

Symbol	Measure	Target
A	Improvement to the WWTP by deepening the secondary clarifiers	Enhanced NH_4^+ removal
B	River quality improvement by installing in-stream aeration stations in the Dommel River	Reduce DO depletion in the river
C	Construction of a tertiary sand filter as an add-on to the WWTP	Enhanced NO_3^- and P removal
D	Construction of additional storage capacity as an add-on to the combined sewer system	Reduce CSOs

Table 6-1 - Measures to be evaluated for application to the UWS

Measure A involves the deepening of the 12 secondary clarifiers of the Eindhoven WWTP by 1.5 m. This has significant improvements on smoothing ammonia peaks and the average nitrate (NO_3^-) removal, by increasing the biological capacity of the plant by allowing a higher sludge mass in the activated sludge system. For measure B, the installation of newly developed in-stream aeration systems is considered to aerate flowing surface water, increasing oxygen levels and thus improving river water quality (Alp and Melching, 2011). By tackling DO depletion the protection of critical river fauna species is better ensured. This measure requires the installation of 5 aeration stations in the Dommel River with a total capacity of 1,460 kgO_2/day. For enhanced TN and P removal, Measure C is proposed – the installation of a sand filter for effluent polishing. The filter is made up of 21 units of a total volume of 1,196 m^3. Finally, with the aim of reducing the release of CSOs and prevent DO depletion, Measure D involves the construction of additional storage capacity for a total of 200,000 m^3, divided over 10 separate locations in the Eindhoven city area.

6.3. Methodology For Assessing And Weighting Life Cycle Environmental Impacts

There are four main phases in an LCA analysis: *Goal & Scope Definition*, *Inventory Analysis*, *Impact Assessment* and *Interpretation*. Description of what the application of these phases generally entails is provided in Section 3.5. Their application is described below, with the Interpretation in the Results section (6.5). Weighting is applied during the *Impact Assessment* phase. In this study we employ shadow prices to attach weights on the estimated impacts. Uncertainty analyses were also performed, first on four of the critical parameters of the LCA to assess the robustness of the impacts estimation, and secondly on the assumed shadow prices used as weights.

6.3.1. Goal & scope definition

The goal of the LCA is to assess the environmental impacts caused by the current operation of the Eindhoven UWS and to compare them against the estimated environmental impacts by each of the proposed upgrading measures. The system boundaries include the catchment under study, the sewer system with its CSOs, the WWTP, chemicals and energy used during the treatment, the river section within the catchment boundaries and pollutant loads (P, $\mathrm{NH_4^+}$, $\mathrm{NO_3^-}$) leaving the studied river section (Figure 6-1). Construction of the WWTP and sewer systems have not been taken into account as they would be the same for all measures as well as the base case. Constructions of the proposed measures have been included in the analysis.

For the sludge treatment, the dewatering installation at Mierlo and the sludge incineration facility (SNB) were taken into account along with the chemicals and energy used at said facilities. The transportation of chemicals and dewatered sludge to the incineration facility were also included. The lifespan and maintenance of mechanical equipment and constructions of the proposed measures have also been taken into account. The functional unit is ten years of system operation treating approximately 546,550,190 m^3 of wastewater in total. This allowed the inclusion of climatic variability over that period as well as seasonal dynamics of influent wastewater.

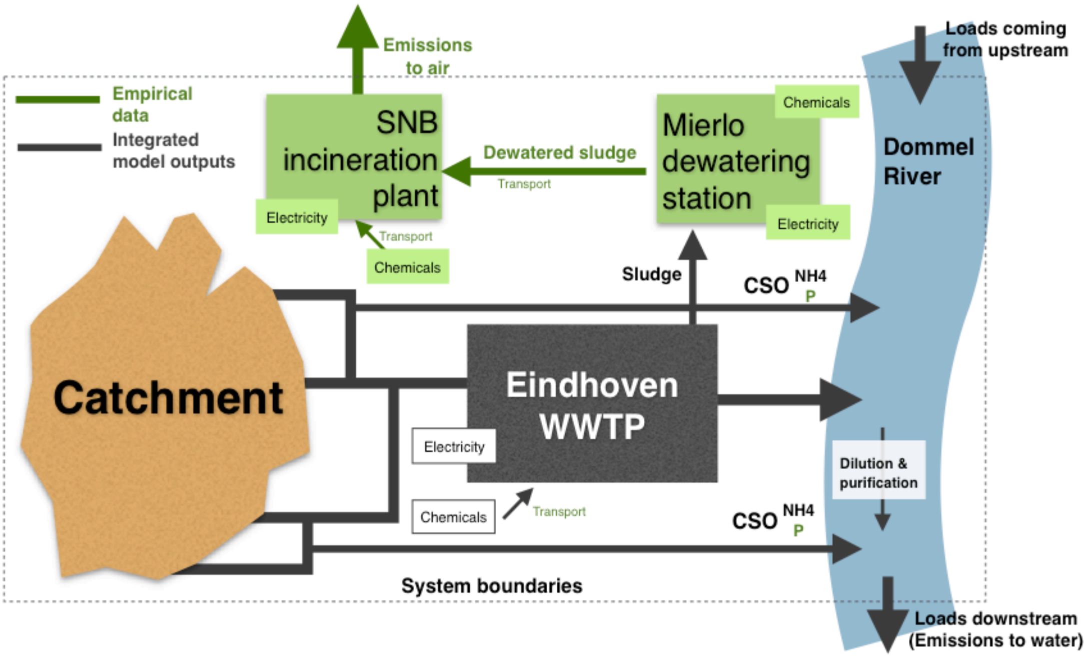

Figure 6-1 - System boundaries. For the Inventory Analysis: items in grey indicate model outputs; items in green indicate empirical data.

6.3.2. INVENTORY ANALYSIS

The inventory data consist of: (i) inputs to the system (energy and chemical consumption and transport); (ii) outputs from the system (emissions to air and water); (iii) inputs and outputs of sludge treatment; (iv) inputs and outputs of construction of each measure (materials used and generated waste); and (v) infrastructure and equipment maintenance of each measure, presented in Table 6-2 and Annex Section 10.2 (Tables 10-3 (for sludge treatment), 10-4 (for construction of each measure) and 10-5 (for the construction of the pumps needed in Measure A)). The sludge treatment data were provided by the WdD (Blom, 2014) and the SNB incineration facility (SNB, 2014) for the year 2013. Information regarding the maintenance of infrastructure and equipment was provided by the WdD. Regarding maintenance, the lifespan of constructed infrastructure and equipment was taken into account. The lifespan of all mechanical equipment was assumed to be 15 years and 30 years for all civil constructions.

The data regarding outputs to water, electricity use and sludge production were obtained using the integrated UWS model. The model has been implemented in the WEST® simulation software (www.mikepoweredbydhi.com) and has been calibrated, validated and extensively used for measure evaluations in previous studies within the KALLISTO project (Benedetti *et* *al.*, 2013b; Langeveld *et al.*, 2013a, 2013b; Weijers *et al.*, 2012). The model is described in more detail in Section 3.2.2 of the Materials and Methods chapter. For each of the measures and the base case the integrated model is simulating the climatic and influent variability occurring in the system for the duration of ten years (Langeveld *et al.*, 2013a). The ten-year time series was developed using monitoring data of precipitation, sewer water levels and flow, and water quality. A description of the development and calibration of the integrated UWS model and the monitoring data used is provided in extensive detail in Langeveld *et al.* (2013a). Concerning the measure modelling, for measure A the height of the implemented clarifiers was adjusted, for measure B an aeration input was implemented in the stream adding oxygen given by a DO set-point, for measure C a tank was added at the end of the WWTP with nutrient removal determined by mathematical equations, and for measure D storage tanks available in the WEST software were added to the sewer system.

		Base case	**A**	**B**	**C**	**D**
Inputs to the system						
Aluminium sulphate, powder	ton	61,590	37,708	61,716	68,555	63,380
Methanol (carbon source)	ton	-	-	-	17,266	-
Construction	-	-	12 deeper clarifiers	5 aeration stations	21 sand filter units	10 storage tanks
Construction maintenance cycle	years	-	30 for clarifiers 15 for pumps	15	30	30
Electricity, medium voltage	MWh	236,376	232,756	237,935	236,408	236,555
Outputs to water at the end of the river reach						
Phosphorus (P)	ton	518	554	518	461	500
Ammonium (NH_4^+)	ton	1,802	1,307	1,794	1,802	1,770
Nitrate (NO_3^-)	ton	3,127	2,834	3,128	2,787	3,156
Transport						
Transport of aluminium sulphate	Mtkm	3.39	2.07	3.39	3.77	3.49
Transport of methanol	Mtkm	-	-	-	0.95	-
Sludge for treatment	Mm^3	5.80	5.78	5.79	5.84	5.93

Table 6-2 - Inventory table for system inputs and outputs. The 10-year operation of base case and the four evaluated measures are presented along with their construction. Inventory table for sludge treatment provided in Table 10-3 of Annex. Inventory tables for construction provided in Tables 10-4 and 10-5 of Annex.

The outputs to water are the net loads released to the environment at the end of the studied reach from the WWTP and CSOs. In this manner, the purification capacity of the receiving water is taken into account in the estimated emissions. The performance of each of the measures with regards to their principal targets (enhance nutrient removal, reduce CSOs, reduce river DO depletion) can then be seen looking at the "Outputs to water at the end the river reach" (Table 6-2) of each measure. The specific contribution of CSO events to the total outputs to water is provided in detail in Table 6-3 for the base case and measure D (storage tanks). The total contribution of CSO events to the emitted NH_4^+ loads appeared to be a small part of the overall emissions (less than 1.2%), as is the contribution of the P loads (7.7% and 4.8% for the base case and measure D equivalently), attributed to the fact that the total CSO volume is less than 4% the WWTP effluent volume over 10 years. Less clear are the benefits provided by the river aeration measure (B), and are discussed in more detail in the Results and Discussion sections (6.5 and 6.6).

	Base case	**Measure D**
CSO Volume (m^3/10 years)	19,928,727	15,667,602
CSO Volume relative to the WWTP effluent	0.036	0.029
NH_4^+ load from CSOs (ton/10y)	20.16	15.17
Total NH_4^+ load emitted to river water (ton/10y)	1802.16	1770.12
Percentage in total NH_4^+ emission to river water (%)	1.12	0.86
P load from CSOs (ton/10y)	39.86	23.91
Total P load emitted to river water (ton/10y)	518.18	500.32
Percentage in total P emission to river water (%)	7.69	4.78

Table 6-3 - CSO contribution to emissions to river water during 10 years in base case and with the implementation of measure D (storage tanks)

6.3.3. Impact assessment

The data from the inventories were introduced into the Simapro® 8.0.3 software (http://www.pre-sustainability.com/) which allows for the modelling and analysis of complete LCAs in a systematic and transparent manner. To calculate the environmental impacts the ReCiPe Midpoint (H) (1.09) (Goedkoop *et al.*, 2009) method was used as it is based on the latest recommendations by the LCA community (Dahlbo et al., 2012; EC-JRC, 2011, 2010). Midpoint indicators were chosen over endpoint indicators as they assume less uncertainty (Ll. Corominas *et al.*, 2013) and they were considered sufficiently relevant by the decision-maker of this case study (WdD). The evaluated categories were: Climate Change (CC), Terrestrial Acidification (TA), Freshwater Eutrophication (FE), Marine Eutrophication (ME), Human Toxicity (HT), and Freshwater Ecotoxicity (FET). Short descriptions and units of equivalence for each of the categories are supplied in Table 3-6 of Section 3.5. The characterisation factors for the major inputs and outputs are provided in Table 10-6 of Section 10.2. The electricity mix of the Netherlands for the year 2008 was used in the impact assessment.

6.3.4. *Uncertainty of inventory values*

Four factors - use of aluminium sulphate, use of electricity, sludge production and effluent loads (including NH_4^+, NO_3^- and P) – were selected to be studied in an uncertainty analysis of the estimated impacts. A Monte Carlo analysis was performed in Simapro® 8.0.3. The studied ranges for the four factors were from -25% to +25% of the applied inventory value and assumed to be described by uniform distributions.

6.4. Shadow Prices For Monetary Weighting Of LCA Impact Categories

The estimation of shadow prices is most often based on an estimation of the damages caused by the release of a pollutant (damage costs) or by calculating the costs associated to its avoidance and removal (abatement costs). The sets of prices used in this study were presented by de Bruyn *et al.* (2010) for the ReCiPe impact categories and about 400 other pollutants for the Netherlands. A wide range of literature sources was employed reporting on abatement and damage costs to produce three sets of prices:

Set 1: Based on abatement costs, characterised at midpoint level

Set 2: Based on damage costs, estimated at ReCiPe midpoint level

Set 3: Based on implicit damage costs, estimated using ReCiPe endpoint factors

The exact shadow prices for 2008 for all three sets and a short commentary on how they were obtained are provided in the Annex (Table 10-7). To convert the prices to their 2014 equivalents, use was made of the Harmonised Index of Consumer Prices (HICP) for the Eurozone. The HICP is an indicator of inflation and price stability compiled by the European Central Bank for all countries of the European Union. The average annual rate of change of the HICP for the Eurozone between 2008 and 2014 is provided in Table 10-8 of the Annex, while the resulting estimates of the three sets of shadow prices for 2014 are given in Table 6-4. Due to the disparity between the values as well as possible theoretical preferences (e.g. damage versus abatement) all three sets were applied for weighting, as a means of addressing the uncertainty behind their estimations and their relative importance.

Impact category	Unit	Set 1 (€/kg)	Set 2 (€/kg)	Set 3 (€/kg)
CC	kg CO2 eq.	0.027	0.027	0.431
TA	kg SO2 eq.	4.510	0.697	0.254
FE	kg P eq.	11.904	1.944	1.944
ME	kg N eq.	7.645	13.651	13.651
HT	kg 1,4-DB eq.	2.512	0.022	0.042
FET	kg 1,4-DB eq.	1.496	0.087	0.087

Table 6-4 - The three sets of shadow prices based on abatement and damage costs per unit of impact category for 2008 by de Bruyn *et al.* (2010) and adapted for the year 2014 according to HICP rate of change (Table 10-8). (CC - Climate Change, TA – Terrestrial Acidification, FE – Freshwater Eutrophication, ME – Marine Eutrophication, FET – Freshwater Ecotoxicity)

6.5. Results

6.5.1. Interpretation Of LCA Results

It is clear that the measure of deeper clarifiers (A) outperforms the base case and the other three measures in its environmental impact in all categories besides Freshwater Eutrophication (FE) (Figure 6-2). This can be attributed to the higher emission of P to water from this measure. The FE category is also the only category where measure C (installation of a sand filter) performs the best due to its reduction of effluent P. The generally bad performance of measure C in comparison with the other measures in all other categories can be attributed to its increased use of aluminium sulphate and methanol. Similarly, measure D (construction of storage tanks) only performs better than the base case in the Marine Eutrophication (ME) and Freshwater Eutrophication (FE) impact categories by reducing the amount of CSOs released in the Dommel River and thus the NH_4^+ and P loads. Measure B (in-stream aeration) does not appear to have any significant impact compared to the base-case scenario. Figure 6-3 presents in more detail the most significant contributing factors to each impact category.

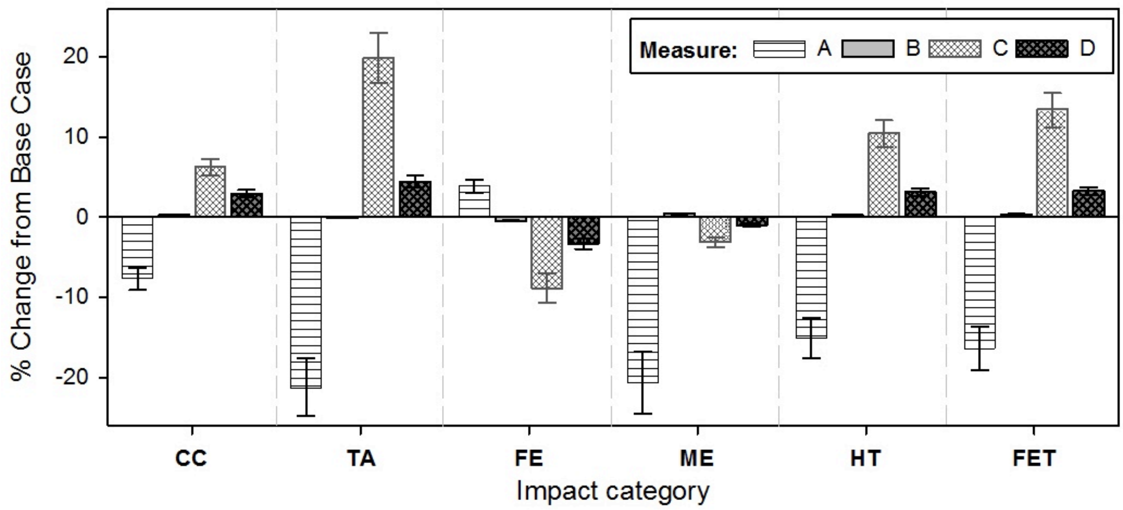

Figure 6-2 - Life Cycle Impact Assessment for 10 years of operation of the four assessed measures (A – Deeper clarifiers, B – In-stream aeration, C – Sand filter, D – Storage tanks) compared to the base case scenario and across the six evaluated impact categories (CC - Climate Change, TA – Terrestrial Acidification, FE – Freshwater Eutrophication, ME – Marine Eutrophication, HT – Human Toxicity, FET – Freshwater Ecotoxicity). The base case is supposed to be at a 0% of impacts. The error bars represent the 2.5th and 97.5th percentiles estimated during the uncertainty analysis, whereas the shaded bars represent the median.

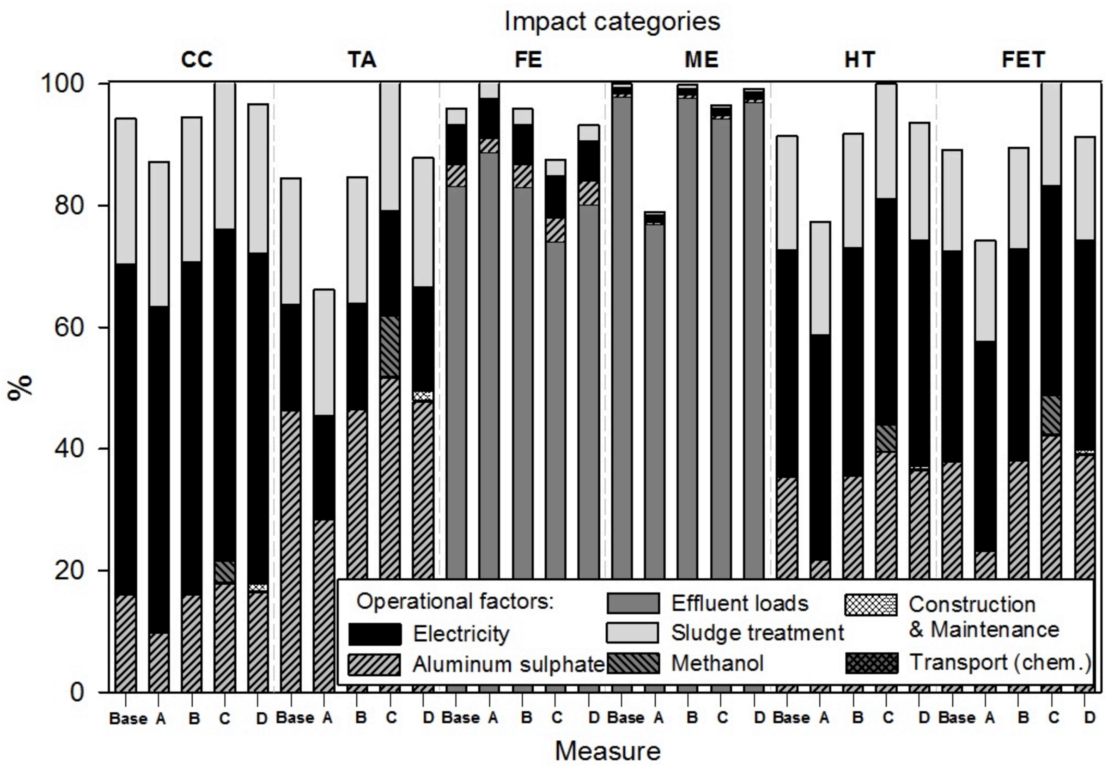

Figure 6-3 - Break-down of impacts for each of the investigated impact categories (CC - Climate Change, TA – Terrestrial Acidification, FE – Freshwater Eutrophication, ME – Marine Eutrophication, HT – Human Toxicity, FET – Freshwater Ecotoxicity) for base case and all measures (A – Deeper clarifiers, B – In-stream aeration, C – Sand filter, D – Storage tanks) after ten years of system operation.

Evidently and as expected, the effluent loads of all measures are the main contributing factor in the Freshwater and Marine Eutrophication categories. Across all other categories, significant effects appear to be caused by the use of aluminium sulphate, electricity and the production and treatment of sludge. This illustrates a trade-off between the increased use of resources and production of sludge and the reduction of P outputs to water, observed in all measures when compared to the base case. Regarding NH_4^+ and $\mathrm{NO_3^-}$, the trade-off is less clear. For measures C and D the increased use of materials results in a $\mathrm{NO_3^-}$ and NH_4^+ reduction respectively. For measure A, these emissions are reduced even with a decreased material use. The trade-offs are less clear for measure B, as there is very little change in impacts when compared to the base case. The use of methanol as a carbon source by measure C also appears to have some effect, albeit lesser in comparison. In terms of construction, measure D causes the highest impacts. Nevertheless, in general the construction and maintenance of measures appear to have little relative effect on the total impacts in 10 years of operation (less than 2%). The effects of chemical transport seem to be of little significance (less than 0.5%).

After the uncertainty analysis of the four factors (use of aluminium sulphate, use of electricity, sludge production and effluent loads), the 2.5^{th} and 97.5^{th} percentiles of all the calculated impacts are presented for each of the measures across all categories in Figure 6-2. It can be seen by comparison that the performance of Measure A remains consistently preferable in all impact categories, barring Freshwater Eutrophication (FE).

6.5.2. Shadow Prices For Weighting Of LCA Impact Categories

In order to attach weights of abatement and damage importance three sets of shadow prices were applied to the environmental impacts estimated in the impact assessment phase. For Set 1, the estimated environmental costs can be an indicator of the environmental losses for the Dutch society stemming from its present and future emission goals. For Sets 2 and 3, the estimated environmental costs can be an indicator of the environmental losses for the Dutch society based on the damage caused by each pollutant. It is important to clarify at this point that the estimated environmental costs are originating and induced by the domestic, municipal and commercial activities producing the wastewater in this catchment rather than the treatment itself. In addition, the prices were estimated based on current abatement standards and therefore the results are meaningful if and only if applied for marginal changes in operation, such as applications of new measures.

This allows for the monetary quantification of the total life cycle impacts induced by each measure. Figure 6-4 presents the total change to the total life cycle environmental costs as compared to the base case. The error bars indicate the 2.5th and 97.5th percentiles of the induced environmental cost of each measure including the uncertainty of the inventory values. The measure with the best relative performance (Measure A) provides a reduction in environmental costs between 10 and 17% relative to the base case. Human Toxicity (HT), Marine Eutrophication (ME) and Climate Change (CC) appear to be driving the environmental costs for sets 1, 2 and 3 respectively.

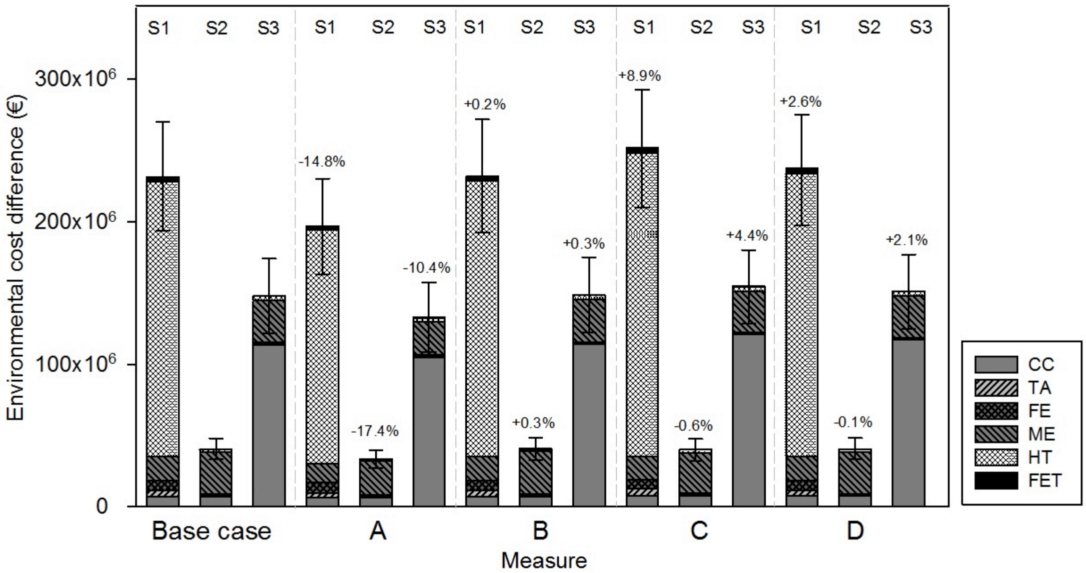

Figure 6-4 - Total Life Cycle environmental costs with respect to the emission goals of the Dutch society and their increase or reduction compared to the current base case (0€) after ten years of system operation. The error bars indicate the uncertainty on the total induced environmental cost of each measure (A – Deeper clarifiers, B – In-stream aeration, C – Sand filter, D – Storage tanks). The total environmental cost is the sum of all the environmental impacts per category (CC - Climate Change, TA – Terrestrial Acidification, FE – Freshwater Eutrophication, ME – Marine Eutrophication, HT – Human Toxicity, FET – Freshwater Ecotoxicity). S1, S2 and S3 represent the shadow price sets Set 1, Set 2 and Set 3 respectively.

As already mentioned, there seems to be an apparent trade-off between the environmental improvement of the reduced P load released and the damages brought about by the use of aluminium sulphate, electricity and sludge production. By use of shadow prices we present the average environmental damage induced per factor (input, output and sludge treatment) per m^3 of treated wastewater (Table 6-5). Accordingly, the evaluation of this trade-off is facilitated by the use of a single indicator. Table 6-5 lists the average environmental cost induced by each factor per m^3 of treated wastewater for the main inputs and outputs of this system. The numbers indicate a significantly higher environmental cost induced by the use of electricity and aluminium sulphate and the production of sludge per m^3 of wastewater, compared to the loads of P, NH_4^+ and NO_3^- released as emissions per m^3 of wastewater.

Factor	Unit	€/(unit of factor)	(Total € per factor)/(m^3 of effluent)
Output - Effluent P	ton	5,364	0.0049
Output - Effluent NH_4^+	ton	3,580	0.0295
Output – Effluent NO_3^-	ton	1,021	0.0147
Input - Aluminium sulphate	ton	562.2	0.0603
Input - Transport (chemicals)	tkm	0.066	0.0003
Input - Methanol	ton	296.5	0.0094
Input - Electricity	kwh	0.223	0.0964
Sludge treatment	m3	4.371	0.0466

Table 6-5 - Estimated average (across the three sets) of environmental monetary cost: per unit of input/output; and total for input/output per m^3 of treated wastewater

6.6. Discussion

The analysis investigates the environmental impacts of the measures by means of an LCA and then assigns relative weights of abatement and damage importance (shadow prices) to each of the impacts. This facilitates: the ranking of impacts according to the present and future emission goals of the Dutch society; the relative ranking of the performance of each of the measures - through a single monetary indicator; and the prioritisation of efforts with regards to reducing environmental impacts for the decision-maker (WdD in this case).

The life cycle evaluation of various impact categories allows for an investigation of the environmental impacts of each measure at the global scale. The environmental performance of the measures, compared against each other, seems to be fairly consistent during the uncertainty analysis. The applied ranges of [-25% to +25%] on the use of aluminium sulphate, electricity, sludge production and effluent do not appear to alter the relative performance of the four measures (Figure 6-2). More specifically, while measures C (sand filter) and D (storage tanks) have reduced outputs to water, their high use of materials - mainly aluminium sulphate and methanol - cause them to have the poorest overall environmental performance. The overall impacts from construction have appeared to be relatively low (less than 2%). Had salvage values of equipment and infrastructure not been accounted for, construction impacts would increase (at most by a factor of 3 for the case of civil constructions). Given the relatively low impacts however, this is not expected to affect the results of the analysis.

This issue brings about an obvious comparison between the use of such materials like aluminium sulphate and electricity and nutrient outputs to water. With the employment of shadow prices this task is facilitated, as the single monetary indicator allows for a direct comparison per m^3 of wastewater. Nonetheless, evaluating the trade-offs after the impacts have been weighted to the same unit simplifies the comparison between the two. The use of a monetary unit allows for their comparison with other aspects of operation and other economic activities (Harmelen *et al.*, 2007): for example, one can compare the environmental damage induced by the electricity required per m^3 of wastewater (now estimated in €) with the environmental damage induced by emitted pollutants per m^3 of wastewater (also estimated in €) and the actual market price paid to purchase the electricity. The much higher environmental cost of aluminium sulphate and electricity used per m^3 of wastewater in comparison to the environmental cost induced by P and NH_4^+ per m^3 of wastewater can be very valuable information to prioritise operational decisions for the system. In the case of using alternative chemicals instead of aluminium sulphate or a different electricity mix with more renewable sources of energy, this trade-off and prioritisation would expectedly change.

With regards to the shadow prices used, it should be noted that they stem from both abatement efforts put into each environmental impact (Set 1) and estimated environmental damage costs brought about by each impact (Sets 2 and 3). That is to say that the costs in Set 1 are a result of how society and its policy perceive them to be; Sets 2 and 3 represent costs with regards to the value of environmental damage. Arguably, if sustainability is to be achieved, environmental policy should aim to equate the two and therefore reach an 'optimal' level of pollution where abatement equals damage. By example of one of the most significant impact categories in this study (HT) its abatement price value used is 2.51 €/kg1,4-DBeq. emitted to air for year 2014. The reported values based on damage costs are 0.022 and 0.042 €/kg1,4-DBeq. emitted to air for year 2014. Differences between the two damage-costs sets are also observed in the categories of CC, TA and HT. These discrepancies are attributed either to abatement costs being significantly higher than the damage costs generated with the release of some pollutants, due to the fact that future costs of Set 3 are not discounted (Ahlroth *et al.*, 2011) or due to the generally great uncertainty surrounding the valuation of some categories, particularly the HT category (Renou *et al.*, 2008).

Measures B and D have already been investigated in previous studies of the system (Benedetti *et al.*, 2013b; Langeveld *et al.*, 2013a, 2013b; Weijers *et al.*, 2012) with two principal aims: target the depletion of DO in the river for critical fauna species and reduce the release of CSOs. The studies have found significant improvements in reducing DO depletion in the river with the application of the river aeration measure (B), which showed clear advantages over the other measures (Benedetti *et al.*, 2013b). However, the investigation of the environmental benefits provided by this measure is still limited when applying LCA, even when the most affected sections of the stream are included within the evaluation boundaries. This is mainly due to the fact that the impacts of DO depletion, particularly for river biodiversity, are not accounted for in LCAs. Even though attempts to develop and include biodiversity aspects in LCA have been on-going for more than a decade now, standardised methodologies are still in their primal stages (Loubet *et al.*, 2014; Pfister *et al.*, 2009). Studies taking into account local specific conditions and characteristics of the receiving medium have not been very apparent in the literature – arguably due to limitations of this methodological framework (Curran *et al.*, 2011; Renou *et al.*, 2008): impacts are regarded as generic in space, aggregated over long time horizons, strongly dependent on the chosen functional unit, and with distinct impact pathways so as to avoid double counting. Even though new approaches are being developed to improve on the resolution of geographical and temporal scales of characterisation factors, no general consensus has been reached yet (Loubet *et al.*, 2014). In these efforts for improved regionalisation of life cycle impacts, deterministic modelling of the systems can be a valuable tool. As previously mentioned, to the best of our knowledge, we consider this the first LCA study for an UWS that employs a deterministic integrated model taking into account hydraulics as well as biochemical processes of the system.

Inherent limitations to the modelling process of UWSs also exist, mainly regarding model uncertainty or the lack of certain processes in the WWTP and the sewer (e.g. N_2O production) or pollutants (e.g. micro-pollutants) in current tools. This is mainly due to the fact that there is not yet a general consensus on a deterministic model for these processes. Modelling uncertainty for this integrated model and system has already been studied during other studies (Benedetti *et al.*, 2013a, 2013b) on the same system. The inclusion of additional processes and pollutants would expectedly increase the estimated impacts, particularly in the Climate Change (CC) category with added GHG emissions and in the toxicity categories with the inclusion of micro-pollutants and heavy metals. Furthermore, the emission of suspended solids has demonstrated a reduction with the application of measures A and D during the analysis. However as no characterisation factor was available these loads were not taken into account during the impact assessment.

On these grounds, this investigation is meant to complement the other studies performed for this particular UWS, principally looking into the most cost-effective upgrades to reach qualities set by the EU WFD at the Dommel River (Benedetti *et al.*, 2013b). Measure B (river aeration) has a clear advantage in this respect (Benedetti *et al.*, 2013b) (most cost-effective reduction of DO depletion and NH_4^+ peaks). On the other hand, measure D (storage tanks) with its significantly higher costs and poor performance with regards to the WFD objectives (Benedetti *et al.*, 2013b) is clearly inferior to the other measures – a result also supported by the outcomes of the LCA presented in this study. Exact cost figures cannot be provided at this moment, but preliminary cost estimates indicate that the installation of the sand filter (measure C) and the deepening of secondary clarifiers (measure A) have costs of the same magnitude, yet significantly lower investment costs than measure D and slightly higher than the installation of river aerators (measure B).

The results of this analysis are meant to complement the investigations for the identification of the most appropriate measure with weighted life cycle impacts of each of the options. The use of shadow prices to weight the impact categories simplifies the process of prioritising for the water management agency as costly and time-consuming methods of gauging social perceptions are avoided.

7. DISCUSSION

7. *DISCUSSION*

This chapter discusses the accomplishment of the stated objectives of this work, its relation to the changing state of water governance in the industrialised world, application of the work to academia and water management, and finally some limitations and future work.

7.1. *ACCOMPLISHMENT OF PHD OBJECTIVES*

7.1.1. DESCRIBE A DECISION-SUPPORT FRAMEWORK FOR DECISIONS IN UWSS

The framework is presented in Chapter 4 of this thesis. It is based on the principles of adaptive management and it incorporates socio-economic, environmental and technical aspects of operation. It includes analyses to address both context and valuation uncertainty explicitly and separately: robustness, reliability and resilience analyses and a valuation uncertainty analysis for model and other valuation assumptions.

7.1.2. APPLY FRAMEWORK TO A REAL UWS CASE STUDY

In Chapter 5 the presented framework is applied on the Congost UWS to demonstrate its applicability and utility for real-world decision-making problems. An integrated model of the system was developed for this purpose, as well as future scenarios for climate, population and energy price. The application demonstrated a way to include environmental externalities in assessments through shadow price monetisation. The sustainability assessment included capital and operational costs for each assessed measure which were balanced against the monetised environmental benefits. Metrics for robustness, reliability and resilience were provided in order to assess the impacts of context uncertainty on the system. A valuation uncertainty analysis assessed the effects of uncertain model parameters on the estimated performance.

7.1.3. DEMONSTRATE AN ALTERNATIVE APPROACH TO SUSTAINABILITY ASSESSMENT

Finally, in Chapter 6 an alternative sustainability assessment (part (iii) of the framework) is presented and applied on the Eindhoven UWS. This novel integration of methods estimated the life-cycle impacts of each of the proposed measures and applied shadow prices to weigh the estimated impacts to a monetised indicator. The application made use of an integrated model of the Eindhoven UWS in order to take into account the functions of the receiving medium in the assessment.

7.2. THIS WORK AS PART OF A CHANGING WATER GOVERNANCE

Talk about 'shifting the paradigm' of water governance has been appearing in literature since the early nineties, with multiple examples throughout the years (Cortner and Moote, 1994; Gleick, 2000; MacGarvin and Johnston, 1993; Niemczynowicz, 1991; Pahl-Wostl, 2007). A catchphrase most of the scientific community can get behind, it appears as though it is now more prevalent than ever.

It is arguably in part attributable to the fact that most important issues related to sanitation and public health are already mostly under control, at least in the industrialised world. Daigger (2007) glorifies environmental engineers as "lifesavers" for the implementation of modern water and wastewater systems in the 20th century. A drive to shift the paradigm from a public health perspective to an environmental one is therefore a logical step forward for both policy and academia. The environmental perspective – the "green revolution" – is not merely rooted in the need to tackle wider issues than public health; it is supported by our ever-increasing understanding of the dependency of the human civilisation on a healthy environment.

This leads to a second major argument for a paradigm shift: humans and their structures are parts of a global network of interdependent natural and constructed systems with relationships more intricate and delicate than we have ever before understood. Water and wastewater systems can no longer be assessed and governed independently from other environmental cycles of nutrients, climate, energy and other aspects.

Recognising this fact has opened a Pandora's box of new challenges to tackle – giving rise to the third driver for a shifting paradigm in water management: growing needs, wider frontiers and higher standards. These include environmental impacts not necessarily related to water quality, threats to the system's stability from future changes in climate and society, more stringent legislation and others discussed in more detail in the introduction and previous chapters.

So what is "the shift" that is being advocated? It has taken many forms throughout the years and the ideas proposed differ in where they put most emphasis. Reading through the literature one can observe some recurring themes making up the necessary elements of governance if a shift in paradigm were to occur:

- Integrated management, both regarding its sectors but also in how issues are tackled
- Participatory governance, with a broader range of involved stakeholders
- Sustainable development as the underlying principle and metric on which systems are assessed and operate
- Proactive instead of retroactive management tackling issues at the source rather than at their effects
- Open and shared data and information to facilitate communication between academia, policy and industry
- Iterative learning cycles of management with adaptive systems and planning procedures

It appears as though little emphasis is put on the importance of new technology to be developed and the role it will play in the new paradigm – whatever that may be. This is not to suggest that innovation and technical developments are not always needed, but rather that the currently available technology is not the limiting factor. There is technology today to help achieve many of the goals set out: energy and nutrient recovery, source separation, water conservation and reclamation (further examples and more discussion can be found in Daigger (2009) and van Loosdrecht and Brdjanovic (2014)). More technology will of course allow for more creativity when set to tackle any challenges, but there is still a lot to be gained by using the already available technology more effectively. Why the available technology has not yet revolutionised water governance can be attributed to various factors, including – but not limited to: institutional red tape, public acceptance, professional segregation, and political stagnation. So what is actually lacking is not the technology but an ability to envision water governance performing at higher levels and instruments that facilitate the full exploitation of available technology towards those higher goals (Daigger, 2007; Pahl-Wostl et al., 2008a). The author believes that the framework presented in this thesis can act as one of those instruments, and the two case studies provided represent examples of that.

The framework offers a novel approach in integrating tools for decision-support in UWS planning. It exemplifies a way that state-of-the-art tools and methods can be brought together in a framework for the holistic assessment of the UWS.

7.3. Application of this work

7.3.1. Chapter 4: The Framework

Chapter 4 mainly dealt with the introduction of the decision-support framework. The presented framework can essentially act as a bridge between currently available tools and technologies and a more contemporary UWS governance that our improved understanding and emerging challenges demand. It embraces adaptive management as its main structure and guides the user through the necessary analyses for a sustainable UWS, robust, reliable and resilient under future changes. Its adaptive component is naturally very applicable to most management situations in the industrialised world. Adaptation is also what the current UWS demands, as has already been argued in the introduction and later chapters, given that the infrastructure is ageing and many new challenges lay ahead.

Effort was put during its development to provide a guiding structure yet maintain its flexibility to include a variety of analyses and metrics. Flexibility has been identified as an important factor in the utilisation of DSTs in real-world problems. Tools should be flexible enough to meet the users' requirements and to allow them to be used however the users see fit, both in terms of their objectives but also in terms of authority and expertise (McIntosh *et al.*, 2011). The framework allows for the application of any tool deemed suitable by the decision-maker for the purposes of sustainability and uncertainty analyses, as long as they conform to the adaptive management principles of the framework. The few notable examples of frameworks already in literature focus on adaptation to climate change impacts but do not allow for other types of uncertainty or sustainability analyses. The two case studies presented used different methods for the "Performance Assessment" part of the framework – a CBA and a LCA. The CBA contrasted the long-term costs of each measure with its long-term environmental benefits to society. The LCA accounted for the environmental pollution brought about by each measure and then monetised it into a social cost. The two represent two different attempts to account for decision sustainability; the former justifies an investment given that environmental benefits enjoyed by society are produced, the latter justifies an investment given that it minimises costs to the environment (and by extension society). Arguments supporting (or dismissing) both approaches can be made. Authors have also argued against monetisation in environmental assessments, something that both approaches employ. The objection is often on the grounds that it is "reductionist" and that it neglects fundamental issues for true sustainability, such as equity and the need for multiple perspectives (Gasparatos *et al.*, 2009; Spangenberg and Settele, 2010). On this rationale the choice of this part of the framework application is left to stakeholder preference. The stakeholder deliberation process, where the criteria for evaluation are set, is an important part of this framework. Naturally, if the stakeholder group prefers a different set of criteria, the outcomes might turn out different. It is the opinion of the author that this “bias” is not really a disadvantage of the framework, as it is one of the ways making the tool more widely applicable. Besides, attempting to somehow benchmark or standardise metrics to account for the sustainable development of the system would be a grand endeavour, way beyond the potential of a doctoral project. In addition, the adaptive component of the framework allows for the selected criteria to be changed in future applications if they are no longer fit for purpose.

An additional novelty of this framework is the combination of analyses for the impacts of context and valuation uncertainty. Uncertainty plays an important role in the framework and is classified in two types based on its origin. Their intrinsic difference is also clearly defined and separated which facilitates the process of tackling and adapting to each type, as the methods and procedures for each one differ. When reviewing similar available frameworks very few take valuation uncertainty into account and no one goes as far as explicitly differentiating between the two.

Explicit definitions of the concepts of robustness, reliability and resilience have also been provided, something that very few studies have done in the past. As the literature on the concepts is not always in agreement and given the complex nature of the system, properly defining the concepts is a complicated endeavour. Resilience in particular is especially problematic, as its meaning changes between the fields of ecology, engineering and decision-making (Zhang *et al.*, 2012) – all pertinent to the decision problems this framework aims to address. It can be argued that the definition of resilience used here is simplistic, as it only takes into account the speed of recovery. Other authors chose to include the extent of failure (Butler *et al.*, 2014), or the “gracefulness” of failure (Scott *et al.*, 2012) within the definition. Literature also offers the concept of vulnerability to account for these effects (Fowler *et al.*, 2003) and in order to avoid further complications, it has been left out of the current version of the framework. It can be included however if deemed necessary, as the framework is laid out with great flexibility as already discussed in Chapter 4.

Finally, the very nature of the framework as well as its outcomes are such that they can be easily communicated and interpreted by a wide range of stakeholders. Even though the framework allows for increased degrees of complexity in the analyses used, its structure and fundamental principles are presented in a simple fashion. Jargon was intentionally avoided as well as recommendations for advanced tools. This is so as to not limit its utility to specific experts proficient in a field, but to allow for a broad spectrum of investigators and practitioners to comprehend and if desired apply the framework.

7.3.2. Chapter 5: Framework application to the Congost catchment

Chapter 5 presented an application of the introduced framework to a real-life decision-making situation. The application was based mainly on a model of the Congost catchment, including the two WWTPs and river. The model, developed in WEST®, describes the Congost river in more detail than was previously available by other models of the same catchment. It includes temperature, radiation and seasonal flow profiles and allows for a more accurate investigation of the river processes than was previously possible. This is a widely studied catchment by both academia and the local decision-makers (Corominas and Neumann, 2014; Devesa *et al.*, 2009; Prat *et al.*, 2012) and the developed model can be a useful tool for further applications by both.

In order to study the impacts of future changes on the system, scenarios were developed based on projections. The projections included population growth, rainfall intensification and electricity price increase. The projections used were developed by respective expert agencies and were accordingly scaled and related to the studied area to create the scenarios. The scenarios themselves can be applied for various other analyses of the same system, model-based or otherwise, for anyone requiring using them. They cover a 30-year time span and represent both moderate and extreme outcomes. In addition, the procedures used for the development of scenarios are transferrable to most other model-based evaluations of UWSs. The scenarios represent commonly studied future changes (population, climate, energy market), relevant to most UWSs and of particular interest to most UWS planners.

The concepts of robustness, reliability and resilience are often not explicitly defined and applied in water and wastewater management, mainly due to the general ambiguity of the terms. Many authors neglect to explicitly define them and often use them interchangeably. A search through the literature of water and wastewater management has not revealed any studies that explicitly define and investigate all these concepts for a system. In this application metrics for all three have been provided, as well as an illustrative application through the case study. The presented metrics are simple and widely applicable to a variety of systems, not least to WWTPs.

Finally, the illustrative application itself has been a relatively low-cost, desktop application, with no specialised equipment necessary, except from the WEST® software. The data used were mostly publicly available as well as the projections. The application itself is thus fairly transferable to other case studies and research efforts.

7.3.3. CHAPTER 6: ALTERNATIVE SUSTAINABILITY ASSESSMENT AT THE EINDHOVEN CATCHMENT

Chapter 6 presented a sustainability assessment for measures proposed for application in the UWS of Eindhoven, as an alternative application of step (iii) of the framework. A novel integration of methods was presented with LCA at its core, complemented with dynamic integrated modelling of the UWS and social weighting of the estimated impacts. The methods used are not particularly groundbreaking themselves; the novelty of the application is brought about by their integration taking conventional LCA a step further.

By use of shadow prices, the estimated life cycle impacts acquired monetary values which were then aggregated to a single value for each measure, to facilitate comparison. With regards to evaluating trade-offs, the task is also simplified after the impacts have been weighted to the same unit. In addition, the use of a monetary unit allows for their comparison with other aspects of operation and other economic activities (Harmelen *et al.*, 2007): for example, one can compare the environmental damage induced by the electricity required per m^3 of wastewater (now estimated in €) with the environmental damage induced by emitted pollutants per m^3 of wastewater (also estimated in €) and the actual market price paid to purchase the electricity. This can serve as a demonstration of a possible integration of estimated life cycle impacts and social benefits with other economic instruments and analyses.

Regarding the use of the deterministic integrated model, it was mainly an attempt to include the receiving water body's characteristics in the analysis. Studies taking into account local specific conditions and characteristics of the receiving water body have not been very apparent in the literature (Curran *et al.*, 2011; Renou *et al.*, 2008). Even though new approaches are being developed to improve on the resolution of geographical and temporal scales of characterisation factors, no general consensus has been reached yet (Loubet *et al.*, 2014). To the best of the authors' knowledge this has been the first LCA study for an UWS that employs a deterministic integrated model taking into account hydraulics as well as biochemical processes of the system, and as such it can be viewed as an interesting new contribution to the LCA literature.

Finally, the results of the LCA are of interest to the decision-maker (WdD) directly. They are complementing other studies performed for this particular UWS, principally looking into the most cost-effective upgrades to reach qualities set by the EU WFD at the Dommel River. The application provides an additional insight concerning the long term and environmental impacts of the measures under evaluation.

7.4. PRESENT LIMITATIONS AND FUTURE STEPS

The development of this framework has been ambitious; it is an attempt to move away from traditional approaches and rather present a way to combine established tools and analyses for a decision-support framework that is greater than the sum of its parts. In a critical retrospect, the framework would significantly benefit from a stronger emphasis on the adaptive capacity of UWSs. It seems short-sighted to evaluate a system for its robustness, resilience and reliability in an adaptive management setting, but not take into consideration its ability to be flexible to new settings and capacity to adapt. Especially when pondering upon the possibility of entirely unexpected changes to the system, the "unknown unknowns", the "black swans". It would be naïve to assume a system is prepared for all that might befall it during its decades-long lifetime. Incorporating a flexibility component, along with possibly vulnerability and risk, would therefore be an important addition to an upgraded version of the framework.

Regarding the sustainability assessment component of the framework and the inclusion of social benefits, the discussion put most focus on one method: the monetisation of externalities by use of shadow prices. It has been argued by some authors (Guest *et al.*, 2009) that such monetisation of nonmarket impacts eliminates the independence of the environmental and social dimensions of sustainable development, and is arguably contrary to the principle of balancing the considerations across all these bases. It would therefore be beneficial to extend the discussion on possible alternative methods to be applied during the sustainability assessment.

This would also facilitate a better approach to adaptive management, as proper stakeholder participation is integral to any adaptive management endeavour: a broad range of perspectives facilitates the recognition of new challenges and needs for institutional change (Pahl-Wostl *et al.*, 2008a). Perspectives on the system and its state present another great uncertainty that can be of fundamental importance and has not been adequately addressed within this work. Valuation uncertainty within this work has been used as a broad term to cover all assumptions made regarding the conceptualisation (or model) of the system and can include model structure, model parameters and inputs among others. In the application, the valuation uncertainty analysis has been kept fairly simple, focussing on model parameters (and WWTP parameters at that). Nonetheless, valuation uncertainty should in general cover a wider scope of valuation assumptions, pertinent to the very understanding of the system and its context by an individual. In this case the term “model” is used broadly to refer to both the conceptual formulation and any mathematical model or algorithm used to represent it. A model, any model, is essentially an abstraction, a simplification used to interpret reality. As such, one can only induce that it must differ from individual to individual – and indeed from reality itself. This discrepancy represents valuation uncertainty. This uncertainty is present at the very core of the problem formulation. Environmental management problems are often value-laden and subjective to the individual perceiving them, and the individuals involved in environmental management (the stakeholders) are many. The stakeholders invariably have different levels of expertise and knowledge, in addition to simply different perspectives. This can inevitably lead to vastly different understandings of the problem itself that are often equally valid (Ascough *et al.*, 2008; Brugnach *et al.*, 2008). If reaching any solution is based on the very understanding of the problem, how is a commonly acceptable solution to be found if a commonly acceptable problem definition does not exist? Measures to identify and minimise valuation uncertainty should therefore be running through the whole decision-making process, from problem conceptualisation to the identification of possible solutions. Model uncertainty (used in this thesis as a simplified valuation uncertainty) is in practice assessed as an “end-of-pipe” analysis, accompanying the results (Refsgaard *et al.*, 2007) – as has also been presented in this framework for the sake of simplicity. In my opinion this presents the largest limitation of the framework offered in this thesis and should act as an additional foundational principle for future versions, along with adaptive management and sustainable development.

Any future approaches based on this framework should therefore include the plurality of perspectives with respect to the issue at hand, as a mean to address this valuation uncertainty (Brugnach *et al.*, 2008). Social learning occurs through the mechanism of deliberation, reflecting on values and perspectives and enhancing knowledge (Schusler *et al.*, 2003). Additional guidance on how to include relevant stakeholders in a deliberation process to explicitly integrate the multiple perspectives would also significantly benefit the framework and encourage future applications.

8. CONCLUSIONS

8. CONCLUSIONS

The main conclusions of this thesis are summarised below.

Sustainable development of the UWS demands that socio-political, environmental, technical and financial aspects are taken into account. Uncertainty has different origins and can compromise the decision-making process and its outcomes in different ways – as such, it is vital for it to be part of any assessment. Adaptive management presents itself as the best available approach to address these concerns and uncertainties. Despite their importance these issues are not often addressed and no decision-support framework that captures all has yet appeared in literature. **The principal outcome of this thesis has been to provide such a framework.**

Chapter 4 presents the **structured framework to support in an UWS decision-making** that is:

- Following the principles of adaptive management for planning and intervention decisions in UWSs;
- Includes relevant stakeholders in the decision-making process;
- Addresses the system's sustainability, by including financial, social and environmental criteria;
- Robust, resilient and reliable against context variability; and,
- Accompanied by an uncertainty estimate of valuation assumptions.

Chapter 5 presented an **illustrative application of the conceptual framework** presented in Chapter 4 on a real case study.

- The application consisted of a set of various analyses for system performance which suggests that a bigger volume than the one implemented might have been more suitable
- The NPV including avoided pollutant externalities was positive for all tested volumes, suggesting an overall benefit to the environment and society

- The results of the application of scenarios for changes in population, climate and energy price highlighted the significance of taking into account a wide range of future changes that might affect both the technical but also the economic performance of a system
- Population growth had the most significant impacts on the metrics of system performance used (NPV and effluent TN), followed by increasing electricity prices for NPV and finally by rainfall intensification due to climate change

Chapter 6 presented an **alternative application of the sustainability assessment part of the framework** presented in Chapter 4.

- The application presented a novel integration of methods to evaluate measures proposed for implementation on a real UWS case study making use of:
    - A integrated deterministic UWS model, including the receiving water body
    - A LCA of six midpoint impact categories
    - Weighting of the estimated impacts using shadow prices as a proxy for social preferences
    - Uncertainty analyses of system inputs and outputs and used shadow prices
- The trade-off comparison between materials (energy, chemicals and others) and effluent quality was facilitated by a monetary indicator
- The additional use of chemicals was shown to have a higher environmental cost than the avoided effluent pollution at that level of treatment

9. REFERENCES

9. References

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10. ANNEX

10. ANNEX

10.1. SUPPLEMENTARY INFORMATION FOR CONGOST CATCHMENT MODEL DEVELOPMENT AND APPLICATION FOR CASE STUDY IN CHAPTER 5

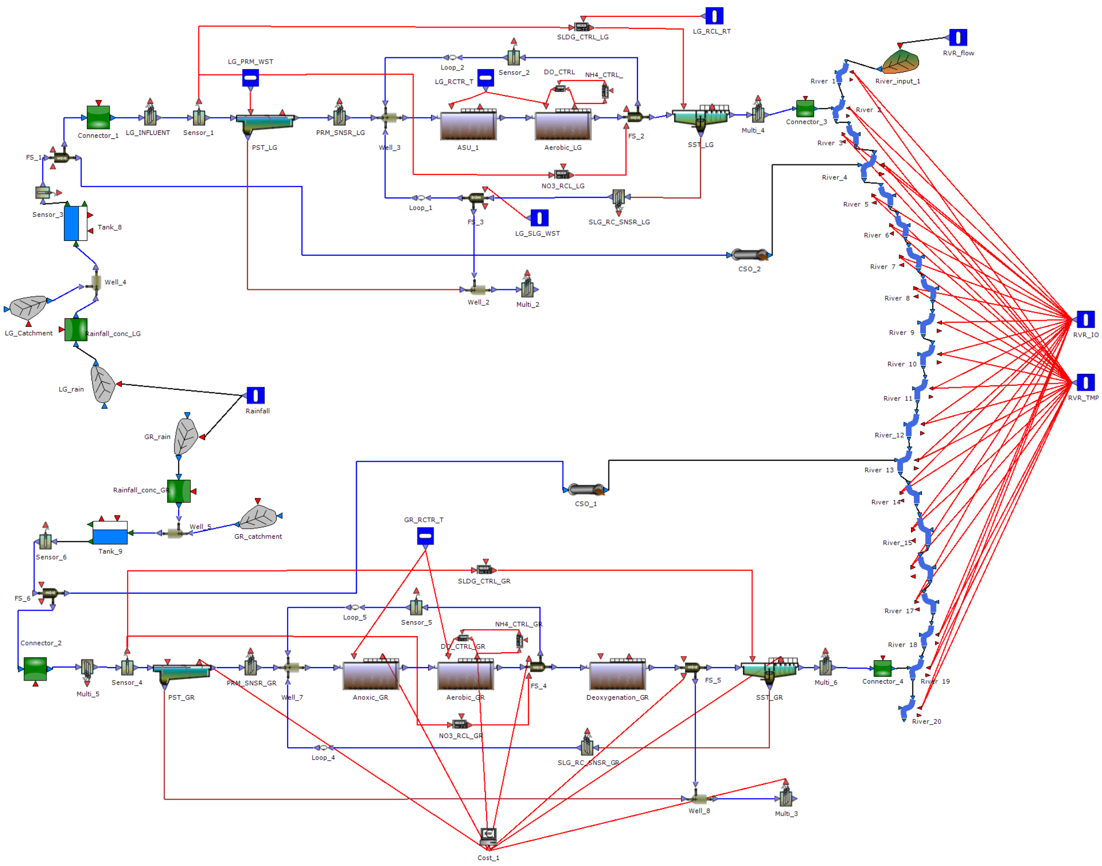

Figure 10-1 - Model layout for the Congost catchment as developed in WEST®

Primary settler			
Name	Value	Unit	Description
Manipulated Variables			
Operational			
Q_Under	3500	m3/d	Underflow rate
Parameters			
Conversion factors			
F_TSS_COD	0.75	-	Fraction TSS/COD
Dimension			
V_Clar	4623	m3	Volume of the clarifier
Energy			
F_Energy_FlowRate	0.04	dUnit/dUnit	Conversion factor Energy needed/Pump flow rate
Settling			
alfa	3.00		Otterpohl and Freund function constant
beta	9.00		Otterpohl and Freund function constant
Anoxic reactor			
Name	Value	Unit	Description
Manipulated Variables			
Operational			
Kla	0.00	1/d	Oxygen transfer coefficient
Temp	15.00	degC	Temperature of the activated sludge
Parameters			
Aeration			
OTR_Energy	1800.00	g/kWh	Oxygen transfer rate per energy input
Composition parameters			
i_N_BM	0.07	g/gCOD	Nitrogen content of biomass X_H, X_PAO, X_AUT
i_N_S_F	0.03	g/gCOD	Nitrogen content of soluble substrate S_F
i_N_S_I	0.01	g/gCOD	Nitrogen content of inert soluble COD S_I
i_N_X_I	0.02	g/gCOD	Nitrogen content of inert particulate COD X_I
i_N_X_S	0.04	g/gCOD	Nitrogen content of particulate substrate X_S

i_P_BM	0.02	g/gCOD	Phosphorus content of biomass X_H, X_PAO, X_AUT
i_P_S_F	0.01	g/gCOD	Phosphorus content of soluble substrate S_F
i_P_S_I	0.00	g/gCOD	Phosphorus content of inert soluble COD S_I
i_P_X_I	0.01	g/gCOD	Phosphorus content of inert particulate COD X_I
i_P_X_S	0.01	g/gCOD	Phosphorus content of particulate substrate X_S
i_TSS_BM	0.90	g/gCOD	TSS to biomass ratio for X_H, X_PAO, X_AUT
i_TSS_X_I	0.75	g/gCOD	TSS to X_I ratio
i_TSS_X_S	0.75	g/gCOD	TSS to X_S ratio
Conversion factors			
F_BOD_COD	0.65	-	Conversion factor BOD/COD
Dimension			
Vol	648.00	m3	Volume of reactor
Kinetic			
DOsat	11.00	g/m3	DO at saturation
K_A	4.00	-	Saturation coeff for S_A (acetate)
K_ALK	0.10	-	Saturation coeff for alkalinity (HCO3-)
K_ALK_AUT	0.50	-	Saturation coeff of autotrophs for alkalinity
K_F	4.00	-	Saturation/inhibition coeff for growth on S_F
K_IPP	0.02	-	Inhibition coeff for X_PP storage
K_MAX	0.34	-	Maximum ratio of X_PP/X_PAO
K_NH	0.05	-	Saturation coeff for NH_4^+ (nutrient)
K_NH_AUT	1.00	-	Saturation coeff of autotrophs for NH_4^+
K_NO	0.50	-	Saturation/inhibition coeff for NO_3^-
K_O	0.20	-	Saturation/inhibition coeff for oxygen
K_O_AUT	0.50	-	Saturation/inhibition coeff of autotrophs for oxygen
K_P	0.01	-	Saturation coeff for phosphorus (nutrient)
K_PHA	0.01	-	Saturation coeff for PHA
K_PP	0.01	-	Saturation coeff for poly-phosphate
K_PS	0.20	-	Saturation coeff for phosphorus in PP storage
K_X	0.10	-	Saturation coeff for particulate COD

K_fe	4.00	-	Saturation coeff for fermentation on S_F
Q_PHA	3.00	1/d	Rate constant for storage of PHA (base: X_PP)
Q_PP	1.50	1/d	Rate constant for storage of PP
Q_fe	3.00	1/d	Maximum rate for fermentation
Temp_Ref	20.00	degC	Reference temperature of the activated sludge
b_AUT	0.15	1/d	Decay rate
b_H	0.40	1/d	Rate constant for lysis and decay
b_PAO	0.20	1/d	Rate constant for lysis of X_PAO
b_PHA	0.20	1/d	Rate constant for lysis of X_PHA
b_PP	0.20	1/d	Rate constant for lysis of X_PP
k_PRE	1.00	1/d	Rate constant for P precipitation
k_RED	0.60	1/d	Rate constant for P redissolution
k_h	3.00	gCOD/(gCOD*d)	Hydrolysis rate constant
mu_AUT	1.00	1/d	Maximum growth rate
mu_H	6.00	1/d	Maximum growth rate on substrate
mu_PAO	1.00	1/d	Maximum growth rate
n_NO_AUT_d	0.33	-	Anoxic reduction factor for decay of autotrophs
n_NO_Het	0.80	-	Reduction factor for denitrification
n_NO_Het_d	0.50	-	Anoxic reduction factor for decay of heterotrophs
n_NO_Hyd	0.60	-	Anoxic hydrolysis reduction factor
n_NO_PAO	0.60	-	Amount of PAO organisms active under anoxic conditions
n_NO_P_d	0.33	-	Anoxic reduction factor for decay of PAO, PP and PHA
n_fe	0.40	-	Anaerobic hydrolysis reduction factor
theta_K_X	0.90		Temperature correction factor for K_X
theta_Q_PHA	1.04		Temperature correction factor for Q_PHA
theta_Q_PP	1.04		Temperature correction factor for Q_PP
theta_Q_fe	1.07		Temperature correction factor for Q_fe
theta_b_AUT	1.12		Temperature correction factor for b_AUT
theta_b_H	1.07		Temperature correction factor for b_H

theta_b_PAO	1.07		Temperature correction factor for b_PAO
theta_b_PHA	1.07		Temperature correction factor for b_PHA
theta_b_PP	1.07		Temperature correction factor for b_PP
theta_k_h	1.04		Temperature correction factor for k_h
theta_mu_AUT	1.11		Temperature correction factor for mu_AUT
theta_mu_H	1.07		Temperature correction factor for mu_H
theta_mu_PAO	1.04		Temperature correction factor for mu_PAO
Mixing energy			
Kla_Min	20.00	1/d	Lowest kLa value that ensures adequate mixing
ME_unit	0.01		Energy requirement per unit of volume for mixing in a AS tank
Mixing_When_Aerated	0.00		Mixing activity during aeration
Stoichiometry			
Y_AUT	0.24	gCOD/gN	Yield For Autotrophic Biomass
Y_H	0.63	gCOD/gCOD	Yield For Heterotrophic Biomass
Y_PAO	0.63	-	Yield coeff (biomass/PHA)
Y_PHA	0.20	-	PHA requirement for PP storage
Y_PO	0.40	-	PP requirement (S_PO4 release) per PHA stored
f_S_I	0.00	-	Fraction of inert COD in particulate substrate
f_X_I	0.10	-	Fraction of inert COD generated in biomass lysis
Aerobic reactor			
Name	Value	Unit	Description
Manipulated Variables			
Operational			
Kla	0.00	1/d	Oxygen transfer coefficient
Temp	15.00	degC	Temperature of the activated sludge
Parameters			
Aeration			
OTR_Energy	1800.00	g/kWh	Oxygen transfer rate per energy input
Composition parameters			

i_N_BM	0.07	g/gCOD	Nitrogen content of biomass X_H, X_PAO, X_AUT
i_N_S_F	0.03	g/gCOD	Nitrogen content of soluble substrate S_F
i_N_S_I	0.01	g/gCOD	Nitrogen content of inert soluble COD S_I
i_N_X_I	0.02	g/gCOD	Nitrogen content of inert particulate COD X_I
i_N_X_S	0.04	g/gCOD	Nitrogen content of particulate substrate X_S
i_P_BM	0.02	g/gCOD	Phosphorus content of biomass X_H, X_PAO, X_AUT
i_P_S_F	0.01	g/gCOD	Phosphorus content of soluble substrate S_F
i_P_S_I	0.00	g/gCOD	Phosphorus content of inert soluble COD S_I
i_P_X_I	0.01	g/gCOD	Phosphorus content of inert particulate COD X_I
i_P_X_S	0.01	g/gCOD	Phosphorus content of particulate substrate X_S
i_TSS_BM	0.90	g/gCOD	TSS to biomass ratio for X_H, X_PAO, X_AUT
i_TSS_X_I	0.75	g/gCOD	TSS to X_I ratio
i_TSS_X_S	0.75	g/gCOD	TSS to X_S ratio
Conversion factors			
F_BOD_COD	0.65	-	Conversion factor BOD/COD
Dimension			
Vol	6720.00	m3	Volume of reactor
Kinetic			
DOsat	11.00	g/m3	DO at saturation
K_A	4.00	-	Saturation coeff for S_A (acetate)
K_ALK	0.10	-	Saturation coeff for alkalinity (HCO3-)
K_ALK_AUT	0.50	-	Saturation coeff of autotrophs for alkalinity
K_F	4.00	-	Saturation/inhibition coeff for growth on S_F
K_IPP	0.02	-	Inhibition coeff for X_PP storage
K_MAX	0.34	-	Maximum ratio of X_PP/X_PAO
K_NH	0.05	-	Saturation coeff for NH_4^+ (nutrient)
K_NH_AUT	1.00	-	Saturation coeff of autotrophs for NH_4^+
K_NO	0.50	-	Saturation/inhibition coeff for NO_3^-
K_O	0.20	-	Saturation/inhibition coeff for oxygen
K_O_AUT	0.50	-	Saturation/inhibition coeff of autotrophs for oxygen

K_P	0.01	-	Saturation coeff for phosphorus (nutrient)
K_PHA	0.01	-	Saturation coeff for PHA
K_PP	0.01	-	Saturation coeff for poly-phosphate
K_PS	0.20	-	Saturation coeff for phosphorus in PP storage
K_X	0.10	-	Saturation coeff for particulate COD
K_fe	4.00	-	Saturation coeff for fermentation on S_F
Q_PHA	3.00	1/d	Rate constant for storage of PHA (base: X_PP)
Q_PP	1.50	1/d	Rate constant for storage of PP
Q_fe	3.00	1/d	Maximum rate for fermentation
Temp_Ref	20.00	degC	Reference temperature of the activated sludge
b_AUT	0.15	1/d	Decay rate
b_H	0.40	1/d	Rate constant for lysis and decay
b_PAO	0.20	1/d	Rate constant for lysis of X_PAO
b_PHA	0.20	1/d	Rate constant for lysis of X_PHA
b_PP	0.20	1/d	Rate constant for lysis of X_PP
k_PRE	1.00	1/d	Rate constant for P precipitation
k_RED	0.60	1/d	Rate constant for P redissolution
k_h	3.00	gCOD/(gCOD*d)	Hydrolysis rate constant
mu_AUT	1.00	1/d	Maximum growth rate
mu_H	6.00	1/d	Maximum growth rate on substrate
mu_PAO	1.00	1/d	Maximum growth rate
n_NO_AUT_d	0.33	-	Anoxic reduction factor for decay of autotrophs
n_NO_Het	0.80	-	Reduction factor for denitrification
n_NO_Het_d	0.50	-	Anoxic reduction factor for decay of heterotrophs
n_NO_Hyd	0.60	-	Anoxic hydrolysis reduction factor
n_NO_PAO	0.60	-	Amount of PAO organisms active under anoxic conditions
n_NO_P_d	0.33	-	Anoxic reduction factor for decay of PAO, PP and PHA
n_fe	0.40	-	Anaerobic hydrolysis reduction factor
theta_K_X	0.90		Temperature correction factor for K_X

theta_Q_PHA	1.04		Temperature correction factor for Q_PHA
theta_Q_PP	1.04		Temperature correction factor for Q_PP
theta_Q_fe	1.07		Temperature correction factor for Q_fe
theta_b_AUT	1.12		Temperature correction factor for b_AUT
theta_b_H	1.07		Temperature correction factor for b_H
theta_b_PAO	1.07		Temperature correction factor for b_PAO
theta_b_PHA	1.07		Temperature correction factor for b_PHA
theta_b_PP	1.07		Temperature correction factor for b_PP
theta_k_h	1.04		Temperature correction factor for k_h
theta_mu_AUT	1.11		Temperature correction factor for mu_AUT
theta_mu_H	1.07		Temperature correction factor for mu_H
theta_mu_PAO	1.04		Temperature correction factor for mu_PAO
Mixing energy			
Kla_Min	20.00	1/d	Lowest kLa value that ensures adequate mixing
ME_unit	0.01		Energy requirement per unit of volume for mixing in a AS tank
Mixing_When_Aerated	0.00		Mixing activity during aeration
Stoichiometry			
Y_AUT	0.24	gCOD/gN	Yield For Autotrophic Biomass
Y_H	0.63	gCOD/gCOD	Yield For Heterotrophic Biomass
Y_PAO	0.63	-	Yield coeff (biomass/PHA)
Y_PHA	0.20	-	PHA requirement for PP storage
Y_PO	0.40	-	PP requirement (S_PO4 release) per PHA stored
f_S_I	0.00	-	Fraction of inert COD in particulate substrate
f_X_I	0.10	-	Fraction of inert COD generated in biomass lysis
Secondary settler			
Name	Value	Unit	Description
Manipulated Variables			
Operational			
Q_Under	200.00	m3/d	Underflow rate

Parameters			
Conversion factors			
F_TSS_COD	0.75	-	Fraction TSS/COD
Energy			
F_Energy_FlowRate	0.04	dUnit/dUnit	Conversion factor Energy needed/Pump flow rate
Settling			
f_ns	0.01	-	Non-settleable fraction of suspended solids

Table 10-1 - Model parameters for primary settler, anoxic reactor, aerobic reactor, controllers and secondary settler

Scenario	A.1.a	A.2.a	B.1.a	B.2.a	C.1.a	C.2.a
Probability	*0.09*	*0.06*	*0.18*	*0.12*	*0.09*	*0.06*
Effluent TN (mg/L)						
S.MIN	11.63	11.63	11.79	11.79	11.93	11.93
S.1	11.25	11.25	11.41	11.41	11.55	11.55
S.2	10.65	10.65	10.79	10.80	10.92	10.93
S.3	9.98	9.98	10.11	10.11	10.23	10.23
S.4	9.60	9.60	9.73	9.73	9.84	9.84
S.5	9.16	9.17	9.28	9.28	9.38	9.38
S.MAX	8.90	8.90	9.00	9.00	9.10	9.10
Volume of CSOs (m^3)						
S.MIN	62,120	86,202	219,001	246,493	601,210	625,541
S.1	1,438	8,199	5,511	17,886	15,633	32,768
S.2	-	-	-	-	-	-
S.3	-	-	-	-	-	-
S.4	-	-	-	-	-	-
S.5	-	-	-	-	-	-
S.MAX	-	-	-	-	-	-
NPV (€)						
S.MIN	110,082,990	110,080,210	115,032,919	115,036,998	119,032,946	119,043,400
S.1	111,066,605	111,102,696	116,350,210	116,381,831	121,070,194	121,103,295
S.2	112,464,226	112,511,184	117,841,284	117,892,931	122,669,157	122,717,961
S.3	114,016,066	114,067,145	119,497,619	119,550,204	124,426,473	124,470,698
S.4	114,868,679	114,912,821	120,400,070	120,451,735	125,373,003	125,419,219
S.5	115,903,999	115,953,878	121,513,874	121,560,566	126,549,256	126,602,170
S.MAX	116,529,121	116,580,163	122,182,491	122,229,995	127,259,375	127,312,105
Scenario	**A.1.b**	**A.2.b**	**B.1.b**	**B.2.b**	**C.1.b**	**C.2.b**
Probability	*0.06*	*0.04*	*0.12*	*0.08*	*0.06*	*0.04*
Effluent TN (mg/L)						

S.MIN	11.63	11.63	11.79	11.79	11.93	11.93
S.1	11.25	11.25	11.41	11.41	11.55	11.55
S.2	10.65	10.65	10.79	10.80	10.92	10.93
S.3	9.98	9.98	10.11	10.11	10.23	10.23
S.4	9.60	9.60	9.73	9.73	9.84	9.84
S.5	9.16	9.17	9.28	9.28	9.38	9.38
S.MAX	8.90	8.90	9.00	9.00	9.10	9.10
Volume of CSOs (m^3)						
S.MIN	62,120	86,202	219,001	246,493	601,210	625,541
S.1	1,438	8,199	5,511	17,886	15,633	32,768
S.2	-	-	-	-	-	-
S.3	-	-	-	-	-	-
S.4	-	-	-	-	-	-
S.5	-	-	-	-	-	-
S.MAX	-	-	-	-	-	-
NPV (€)						
S.MIN	109,033,233	109,030,580	113,933,891	113,938,292	117,892,119	117,902,885
S.1	110,015,065	110,051,137	115,248,016	115,279,829	119,922,340	119,955,745
S.2	111,411,347	111,458,257	116,737,534	116,789,204	121,519,502	121,568,322
S.3	112,961,948	113,013,018	118,392,480	118,445,062	123,275,355	123,319,457
S.4	113,815,496	113,859,542	119,295,876	119,347,527	124,222,752	124,268,935
S.5	114,848,747	114,898,627	120,407,424	120,454,060	125,396,517	125,449,474
S.MAX	115,473,458	115,524,505	121,075,604	121,123,055	126,106,157	126,158,904

Table 10-2 - Net Present Value (NPV), CSO volume and effluent TN concentrations of all proposed solutions, estimated for all scenarios. NPV after 20 years of operation including environmental costs and benefits.

10.1.1. MODERATE AND EXTREME SCENARIO GENERATION

R code used to generate moderate and extreme scenarios of rainfall for the Congost catchment based on baseline rainfall series from 2007. Presented in Figure 3-6.

Read baseline precipitation series

```
> precip<-read.table(file.choose())
```

Calculate series mean, variance and standard deviation

```
> med.gam<-sapply(precip,mean)

> var.gam<-sapply(precip,var)

> sd.gam<-sapply(precip,sd)
```

Calculate shape and scale parameters of series

```
> pshape<-(med.gam/sd.gam)^2

> pscale<-var.gam/med.gam
```

Convert series into quantiles

```
> precip<-sapply(precip,as.numeric)

> quantiles<-rank(precip)/(length(precip)+1)
```

Generate new rainfall series (new_precip) using quantiles and adjusted shape (shape2) and scale (scale2) parameters

```
> new_precip <- qgamma(quantiles,shape=shape2,scale=scale2)
```

10.2. Supplementary Information for Eindhoven catchment *model and application for case study in Chapter 6*

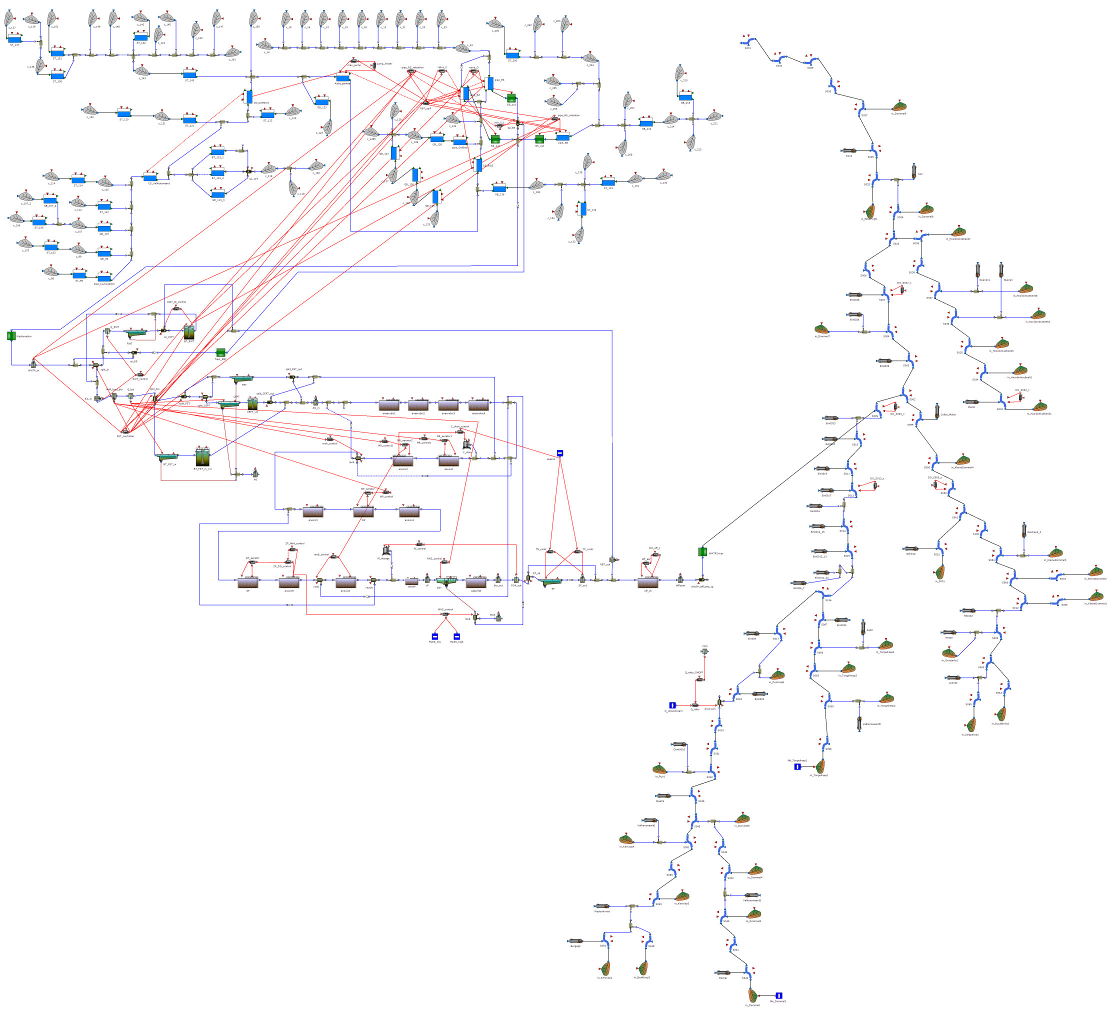

Figure 10-2 - Model layout for the Eindhoven catchment as developed in WEST®

Mierlo sludge dewatering plant	
Inputs to the system (Electricity/ fuels)	**kWh/m³ sludge**
Electricity	2.67
Natural gas	0.53
Inputs to the system (Materials)	**kg/m³ sludge**
Acrylonitrile	0.00042
Tap water	0.72
Outputs to further treatment (Sludge)	**ton/m³ sludge**
Dewatered sludge going to the SNB incineration plant	0.089
SNB incineration plant	
Inputs to the system (Electricity/ fuels)	**kWh/ton dewatered sludge**
Electricity (from the grid)	53.98
Electricity (produced)	7.87
Natural gas	10.44
Inputs to the system (Materials)	**kg/ton dewatered sludge**
Hydrochloric acid (30%)	4.27
Sodium hydroxide solution (50%)	7.40
Limestone	16.65
GAC (adsorbent)	1.82
Performax 3400	0.028
Biocides	0.021
Chlorine dioxide	0.076
Potassium hydroxide (Advantage 101M)	0.00047
Phosphoric acid	0.0021
Emissions to air	**kg/ton dewatered sludge**
Sulfur dioxide (SO2)	0.011
Nitrogen oxides (NOx)	0.092
Ammonia (NH3)	0.016
Hydrogen chloride (HCl)	0.0017

Carbon monoxide (CO)	0.0057
Methane, biogenic	0.0027
Dust particulates	0.0022
Hydrogen fluoride (HF)	0.0001
Dinitrogen monoxide (N2O)	0.00025
Carbon dioxide (CO2), biogenic	0.32
Transport	**tkm/ton dewatered sludge**
Transport of dewatered sludge	99
Transport of hycrochloric acid	0.21
Transport of sodium hydroxide	0.37
Transport of limestone	0.83
Transport of adsobent	0.091
Transport of Performax 340	1.41
Transport of biocides	1.047
Transport of chloride dioxide	3.78
Transport of Advantage 101M	0.024
Transport of Drewcor 2170	0.021
Transport of TMT 15	1.14
Transport of phosphoric acid	0.11
Avoided products	**kWh/ton dewatered sludge**
Electricity	0.99

Table 10-3 - Inventory table for sludge treatment operation (dewatering at Mierlo -7 km from the WWTP- and incineration at SNB -100 km from Mierlo-)

A. Construction deeper clarifier (1 piece)			**B. Construction aeration station (1 piece)**			**C. Construction sand filter (1 piece)**			**D. Construction storage tank (1 piece)**		
Inputs for construction (Materials/fuels)											
Steel, chromium steel	20808	kg	Steel, chromium steel	33	kg	Diesel, burned in building machine	2275	MJ	Diesel, burned in building machine	1037217	MJ
Pump (110 KW)	1	pc.	Polyurethane, rigid foam	1	kg	Reinforcing steel	1226	kg	Reinforcing steel	30813	kg
						Wire drawing, steel	17.58	kg	Wire drawing, steel	440.19	kg
						Concrete, normal	46.26	m3	Concrete, normal	1162.08	m3
						Silica sand	82828	kg			
Transport											
Transport of steel	832.32	tkm				Transport of concrete	1860	tkm	Transport of concrete	847718	tkm
Outputs to further treatment											
						Disposal, inert waste	46.5	ton	Disposal, inert waste	21193	ton

Table 10-4 - Inventory tables for the constructions of each measure. Inventory for the construction of the pump is provided in Table S3.

Construction Pump (110 KW)	
Inputs to the system (Materials)	**kg**
Cast iron	1795.63
Steel, low-alloyed	508.76
Aluminum	70.69
Copper	86.17
Steel	115.58
Synthetic rubber	31.58
Diethylene glycol	46.44
Acrylonitrile-butadiene-styrene copolymer	0.41
Bronze	1.96
Polysulfide, sealing compound	3.16
Polyester resin, unsaturated	4.39
Inputs to the system (Electricity/heat)	**kwh**
Electricity	4818
Heat (for electricity)	1094.5
Diesel	5.06
Heat (for diesel)	218.9
Transport	**tkm**
Transport of polyester	538

Table 10-5 - Inventory table for the construction of the pump used in measure A

	Category	CC	TA	FE	ME	HT	FET
	Unit	kg CO2 eq	kg SO2 eq	kg P eq	kg N eq	kg 1,4-DB eq	kg 1,4-DB eq
Aluminium sulfate	1kg	0.726	0.008	0.0004	0.0002	0.486	0.016
Transport	1tkm	0.111	0.0005	0.00001	0.00003	0.016	0.0008
Electricity	1kWh	0.642	0.0008	0.0002	0.00009	0.132	0.004
Methanol	1kg	0.599	0.006	0.0001	0.00007	0.212	0.009
Sludge treatment	1m^3	11.515	0.038	0.003	0.002	2.742	0.073
Phosphorus (P)	1kg	-	-	1.000	-	-	-
Ammonium (NH_4^+)	1kg	-	-	-	0.780	-	-
Nitrate (NO_3^-)	1kg	-	-	-	0.230	-	-

Table 10-6 - Characterisation factors used during the Impact Assessment for major inputs and outputs

Impact category	**Set 1 (€/kg)**	**Set 2 (€/kg)**	**Set 3 (€/kg)**	**Calculation of abatement**	**Calculation of damage**
CC	0.025	0.025	0.395	CO_2 trading price under the EU Emissions Trading System Cost reductions under the Clean Development Mechanism of the Kyoto Protocol Marginal costs of reducing domestic emissions Fines for exceeding usage norms under the Fertiliser Act (Netherlands) Maximum tolerable risk levels set by Dutch legislation	Various studies with estimations of damage for pollutants at the midpoint level (CO2, SO2, P, etc.), or at the endpoint level (DALY and PDF) converted to midpoint impacts using endpoint characterisation factors
TA	4.13	0.638	0.233		
FE	10.9	1.78	1.78		
ME	7.00	12.5	12.5		
HT	2.30	0.0206	0.0386		
FET	1.37	0.08	0.08		

Table 10-7 - Estimated shadow prices for 2008 and commentary on how they were obtained. From de Bruyn *et al.* (2010).

Year	**HICP annual average rate of change (%)**
2008	3.3
2009	0.3
2010	1.6
2011	2.7
2012	2.5
2013	1.4
2014	0.4

Table 10-8 - Annual average rate of change of the Harmonised Index of Consumer Prices (HICP) for the Eurozone. Data publically available by the European Central Bank.