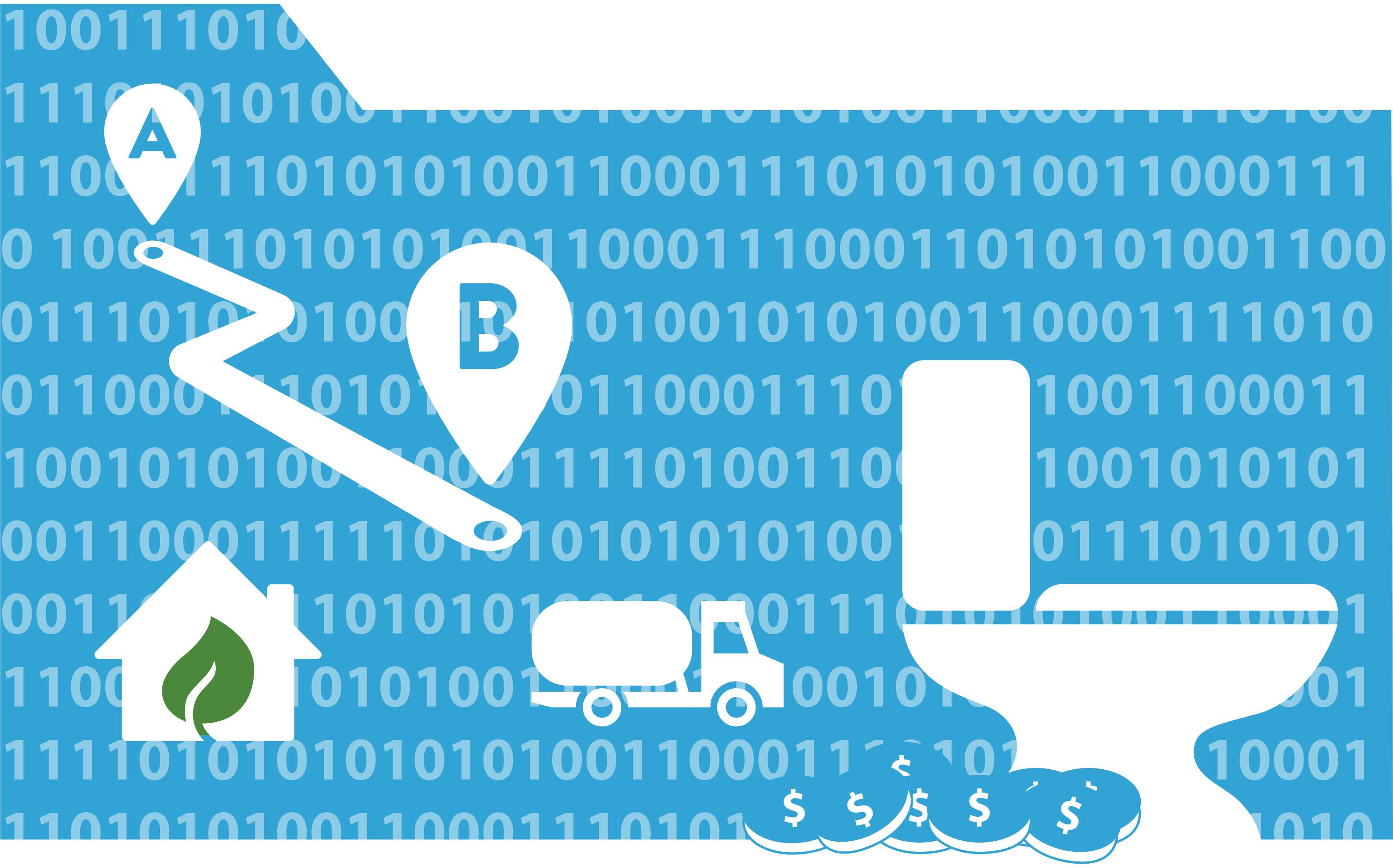
Diss.-No. ETH 23915

The optimal degree of centralisation for wastewater infrastructures

A model-based geospatial economic analysis

Eggimann Sven



ETH zürich

eawag aquatic research

Contact

Eawag Institute
Swiss Federal Institute of Aquatic Science and Technology
Überlandstrasse 133
8600 Dübendorf
Switzerland

ETH Institute
Institute of Civil, Environmental and Geomatic Engineering
ETH Zürich
Stefano-Franscini-Platz 5
8093 Zurich
Switzerland

Suggestion for quotation:

Eggimann, S., 2016. The optimal degree of centralisation for wastewater infrastructures. A model-based geospatial economic analysis. Doctoral Thesis, ETH Zurich, Zurich, Switzerland.

DISS. ETH NO. 23915

The optimal degree of centralisation for wastewater infrastructures

A model-based geospatial economic analysis

A thesis submitted to attain the degree of
DOCTOR OF SCIENCES of ETH ZURICH

(Dr. sc. ETH Zurich)

presented by

Sven Jonas Eggimann

MSc UZH in Geography, University of Zurich

born on 17.03.1988
citizen of Sumiswald, Canton of Bern

accepted on the recommendation of

Prof. Dr. Max Maurer, examiner
Prof. Dr. Bernhard Truffer, co-examiner
Prof. Dr. Robert Weibel, co-examiner

2016

Summary

Efficient infrastructure planning is essential to achieve a more sustainable future. This includes aiming for optimal and economically more efficient infrastructures in the field of urban water management (UWM). Especially network-based infrastructures need special attention as they are generally very cost intensive and have distinctly demanding planning characteristics. Today's historically evolved UWM practices tend to be network-based as in the last century enormous investments have been made to build-up vast sewer networks and large central wastewater treatment plants which centrally connect very high population percentages in many countries. The construction of large network infrastructures has resulted in a dominant centralised planning paradigm and heavily centralised wastewater management systems (WMS). However, this strong central planning paradigm is increasingly being questioned with respect to sustainability challenges and the call is becoming louder for innovative and radical approaches to address future challenges in the field of UWM. One such innovative approach is the decentralised treatment of wastewater by decentralised WMS, although in countries with high central connection rates this has so far been seen mainly as a stopgap solution. Today, connection rates differ significantly across countries: Great differences exist not only between high-income and low-income countries but also within OECD countries. It is unclear to what degree centralised or decentralised WMS should be installed in a region from the point of view of economic efficiency. Whereas the UWM literature presents specific cost assessments, there is a paucity of tools and methods to assess the costs of WMS consisting of centralised and decentralised systems (hybrid WMS). There are also few cost assessments of decentralised WMS with respect to their operation and maintenance costs. This dissertation uses an economic geospatial model-based approach to address the challenging research question of the optimal degree of centralisation. Conceptually, especially the spatially explicit full-cost modelling and the socio-technical conceptualisation of WMS are of special relevance. The basic trade-off between wastewater transportation and treatment is modelled in a geographically explicit way, economies of density for decentralised systems are assessed and a framework is presented to assess different optimal connection rates for hybrid systems. This framework is then used to assess economically sustainable connection rates in order to promote the economic argumentation for a possible sustainability transition in UWM.

This dissertation demonstrates that the use of models is a promising way to address the question of the optimal number, geographical placement and dimensioning of infrastructures, and that space has a significant influence

on the costs of centralised, decentralised and hybrid WMS. Another major finding is the importance of the co-evolution of institutional, organisational and technological innovation in the field of UWM. This thesis also indicates 'threshold-effects', i.e. when transitioning towards lower connection rates, cost differences between the central and decentral systems within a region may become negligible from certain connection rates onwards, which could potentially lead to an increase in the transition dynamics towards new system configurations. On the basis of the findings of this thesis, the introduction of more explicitly spatial price differentiation to facilitate the installation of decentralised WMS is recommended: central networks should be regulated to ensure that they only connect households as long as the average central costs of all households connected to a network decrease. The results suggest that regulations such as the mandatory connection rule should be reconsidered in countries such as Switzerland. Finally, even though the focus of this thesis is on wastewater infrastructures, the argument is made that the research challenge of determining the optimal degree of centralisation is generic and applies to many different infrastructure sectors.

Zusammenfassung

Eine effiziente Infrastrukturplanung ist für eine nachhaltige Zukunft entscheidend. Eine zentrale Rolle kommt dabei der Optimierung der ökonomischen Effizienz von Infrastrukturen in der Siedlungsentwässerung zu. Insbesondere netzbasierte Infrastrukturen bedürfen aufgrund ihrer hohen Kostenintensität und planerisch anspruchsvollen Eigenschaften spezieller Aufmerksamkeit. Im vergangenen Jahrhundert wurden enorme Investitionen in den Bau von immensen Kanalisationsnetzen und grossen, zentralisierten Kläranlagen getätigt. Dies führte in vielen Ländern dazu, dass grosse Teile der Bevölkerung mit zentralisierten Abwasserinfrastrukturen versorgt werden. Die so historisch gewachsenen Netzwerkinfrastrukturen in der Siedlungswasserwirtschaft resultieren in einem dominierenden zentralistischen Planungsparadigma mit stark zentralisierten Abwassersystemen. Dieses Planungsparadigma wird jedoch in Bezug auf seine Nachhaltigkeit zunehmend hinterfragt. Um zukünftigen Herausforderungen in der Siedlungswasserwirtschaft gerecht zu werden, wird der Ruf immer lauter nach innovativen und radikal alternativen Lösungen. Ein solcher innovativer Ansatz stellt diesbezüglich die dezentrale Abwasserreinigung mithilfe von dezentralen Abwassersystemen dar. Bis anhin wurde dieser in Ländern mit einem hohen zentralen Anschlussgrad häufig lediglich als Notlösung angesehen. Der Anschlussgrad an zentrale Abwasserinfrastrukturen unterscheidet sich in verschiedenen Ländern: Grosse Unterschiede existieren nicht nur zwischen einkommensstarken und einkommensschwachen Ländern, sondern auch innerhalb der Länder der OECD. Zu welchem Grad zentrale oder dezentrale Abwassersysteme in einer Region am besten installiert werden sollen ist auf Grundlage einer ökonomischen Argumentation nicht klar. In der Literatur werden bereits spezifische Kostenberechnungen beschrieben, in der Praxis jedoch mangelt es an Werkzeugen und Methoden, um die Kosten für hybride Abwassersysteme, welche aus zentralen und dezentralen Systemen bestehen, zu bestimmen. Überdies gibt es kaum Kostenanalysen zu dezentralen Abwassersystemen hinsichtlich deren Betriebs- und Wartungskosten. In dieser Dissertation wird ein räumlicher, modellbasierter Ansatz gewählt, um die Forschungsfrage nach dem optimalen Zentralisierungsgrad in Bezug auf dessen Wirtschaftlichkeit zu beantworten. Von besonderer konzeptioneller Relevanz ist die räumlich explizite Vollkostenanalyse und die sozio-technische Konzeptualisierung von Abwassersystemen. Die grundlegend gegenläufige Abhängigkeit zwischen den Kosten für den Transport von Abwasser und dessen Reinigung wird geografisch explizit modelliert, Dichteeffekte von dezentralen Abwassersystemen werden eruiert, und im Weiteren wird ein Ansatz vorgestellt, mit dessen Hilfe verschiedene optimale Zentralisierungsgrade für hybride Systeme bestimmt werden können. Letzterer erlaubt die

Bestimmung von ökonomisch nachhaltigeren Anschlussgraden und hilft dabei, die ökonomische Argumentation einer Nachhaltigkeit-Transition in der Siedlungswasserwirtschaft weiterzuentwickeln.

Diese Dissertation zeigt das Potential eines modellbasierten Ansatzes auf, um der Frage nach der optimalen Anzahl, der geografischen Lage und der Dimensionierung von Infrastrukturen nachzugehen zu können. Zudem verdeutlicht sie die Wichtigkeit der räumlichen Komponente als wichtiger Einflussfaktor in Bezug auf die Kosten von zentralen, dezentralen und hybriden Abwassersystemen. Diese Arbeit verdeutlicht zudem die Wichtigkeit der Koevolution institutioneller, organisationaler und technologischer Innovationen in der Siedlungswasserwirtschaft. Es werden ebenfalls mögliche ‚Schwelleneffekte' angedeutet: Bei einer allfälligen Transition zu tieferen Anschlussgraden könnte beim Erreichen von bestimmen Anschlussgraden die Kostendifferenz zwischen zentralen und dezentralen Systemen innerhalb einer Region vernachlässigbar klein werden. Dies würde die Dynamik einer Transition hin zu neuen Systemkonfigurationen forcieren. Aufgrund der Resultate dieser Arbeit ist eine explizite Preisdifferenzierung zu empfehlen, um die Einführung von dezentralen Abwassersystemen zu erleichtern. Zentrale Netzwerke sollten so reguliert werden, dass Haushalte nur angeschlossen werden, solange dadurch die durchschnittlichen Kosten aller angeschlossenen Haushalte sinken. Diese Arbeit zeigt weiter, dass Regulierungen wie die Anschlusspflicht besonders in Ländern wie der Schweiz hinterfragt werden sollten. Obwohl der Fokus dieser Arbeit auf Abwasserinfrastrukturen liegt, kommt der Bestimmung des optimalen Zentralisierungsgrades nicht zuletzt auch dadurch breit angelegte Bedeutung zu, da sich dieselbe Fragestellung in verschiedenen Infrastruktursektoren stellt.

Acknowledgements

These last few years were truly a time of transition. Fortunately, I seized the window of opportunity offered to me in 2013 by my supervisor Prof. Dr. Max Maurer to pursue a doctoral thesis at ETH Zurich and the Swiss Federal Institute of Aquatic Science and Technology.

My sincere and deep gratitude goes first to my doctoral advisors Prof. Dr. Max Maurer and Prof. Dr. Bernhard Truffer who not only made this thesis possible but shaped, sharpened and broadened my thinking inside and outside the academic world. They not only trained me in the art of science and introduced me to fascinating and challenging academic worlds but encouraged and supported me at all times. I am grateful for their excellent supervision, good humour and steadfast optimism.

I would also like to thank Prof. Dr. Robert Weibel, who kindly accepted to act as a co-examiner.

Furthermore, I especially thank the numerous inspiring people with whom I worked together and who made their contributions inside or outside academia: Rachel Barret, Philipp Beutler, Dr. Christian Binz, Dr. Frank Blumensaat, Prof. Dr. Roy Brouwer, Annina Brunner, Dr. Damian Dominguez, Dr. Christoph Egger, Jonas Eppler, Dr. Lea Fuenfschilling, Dr. Dario Del Guidice, Dr. Heiko Gebauer, Sabine Lautenschläger, Dr. Ivana Logar, Dr. João & Joana Leitão, Matteo Mattmann, Prof. Dr. Eberhard Morgenroth, Lena Mutzner, Dr. Roni Pen, Dr. Jörg Rieckermann, Andreas Scheidegger, Prof. Dr. David Sedlak, Rosi Sieber, Dorothee Spuhler, Mariane Schneider, Matthew Moy de Vitry and Omar Wani.

A special thanks for motivational support go to Dr. Christian Hofreiter, Prof. Dr. Peter Imming and Prof. Dr. John Lennox.

I am grateful to my grandparents and parents - without them I would not be where I am. Finally and most importantly, I wish to express my gratitude to my wife Hannah for her love and patience.

Soli deo gloria

Eggimann Sven
November 2016, Zürich

“

...in general, no great breakthrough in sewage treatment technology can be expected, at least not cost-wise. One of the hopes of lowering the costs is the joining of industries and municipalities in common treatment facilities to take advantage of the economies of scale in waste water treatment.

”

Deininger and Su (1973)
Water Research

“

We assume that decentralised alternatives can already, or will soon be able to, deliver utility services of comparable quality, which means that the superiority of the centralised paradigm can no longer be taken for granted, and questions about the optimal degree of centralisation need to be addressed.

”

Eggimann et al. (2015)
Water Research

Table of Contents

Abbreviations

CR	-	connection rate
CW	-	Clarke and Wright
DC	-	degree of centralisation
DTM	-	digital terrain model
eFAST	-	extended fourier amplitude sensitivity test
EM	-	expansion module
GIS	-	geographical information system
HAC	-	hierarchical clustering
MBR	-	membrane bio-reactor
MM	-	merging module
MST	-	minimum spanning tree
NPV	-	net present value
NN	-	nearest neighbour
OECD	-	organisation for economic co-operation and development
OCR	-	optimal connection rate
ODC	-	optimal degree of centralisation
O&M	-	operation and maintenance
OST	-	on-site
PE	-	population equivalent
RGI	-	ruggedness terrain index
SBR	-	sequencing batch reactor
SNIP	-	sustainable network infrastructure planning
SOM	-	system option module
USU	-	urban structural unit
UWM	-	urban water management
VRM	-	vector ruggedness measure
WMS	-	wastewater management system
WWTP	-	wastewater treatment plant

Chapter 1

Introduction

1.1 Motivation and overall ambition

1.1.1 The significance of UWM infrastructure

The infrastructure of urban water management (UWM) is indisputably one of the vital elements of human well-being, enabling human and environmental health and laying the foundations for socio-economic development (inter alia Corcoran et al. 2010, WHO and UN-Water 2014). The primary function of UWM is to provide urban hygiene, e.g. to prevent outbreaks of life-threatening diseases such as typhoid or cholera (O'Flaherty 2005, WHO 2011, Sedlak 2014, Urich and Rauch 2014a). In most parts of the world, today's UWM systems rely heavily on network-based infrastructures, namely drinking water distribution and sewer networks. Historically, centralised network-based wastewater management systems (WMS) with progressively higher treatment performance developed in the late second half of the 19th century (Wiesmann et al. 2006, Gikas and Tchobanoglous 2009, Lofrano and Brown 2010). This centralised approach is most commonly based on transporting sewage from households and industry with the aid of gravity (or pumping) and flushing water in combined sewer systems to centralised treatment facilities. Together with other infrastructure networks such as the electricity grid, roads, district heating, or natural gas networks, they constitute the backbone of current industrialised societies and their construction in the last two centuries has been a major engineering and societal achievement (Hansman et al. 2006, Lofrano and Brown 2010, Hall et al. 2016).

On a global scale, however, the goal of sustainable UWM for all has not been reached with the centralised network-based approach, and even in places where the practice of centralised UWM has so far been successful, various challenges need to be addressed (inter alia Sadoff et al. 2015, Larsen et al. 2016). This conclusion has been confirmed by the recent sustainable development goals which increasingly focus on sustainable UWM, infrastructure development and the promotion of innovative new

Box 1.1

Sustainable development goals

Numbers 6 and 9 of the sustainable development goals of the United Nations are particularly relevant to sustainable UWM, innovation and infrastructure (UN 2015a).

Goal 6: Ensure availability and sustainable management of water and sanitation for all.

Goal 9: Build resilient infrastructure, promote inclusive and sustainable industrialization and foster innovation.

approaches to service provision (see Box 1.1). These manifold and diverse UWM-related challenges are intensively discussed in the literature and are easy to identify: they range from combined sewer overflows and leaking pipes to climate change and governance and financial challenges (inter alia Daigger 2007, Kluge and Scheele 2008, Bahri 2012, Sadoff et al. 2015, Gulbenkian 2014, WWAP 2015, Gawel 2015, Larsen et al. 2016, Hall et al. 2016). UWM infrastructures are both essential and face strong challenges, and alternative innovative approaches are necessary for more sustainable modes of consumption and production in UWM (Markard et al 2012, Hering et al. 2013, Kiparsky et al. 2013, Truffer et al. 2013).

1.1.2 Innovation and socio-technical evolution of centralised UWM

Centralised wastewater treatment includes much more than just technology or physical artefacts, and WMS have to be considered as '*socio-technical*' systems matching technologies and infrastructures with appropriate organisational and institutional models (inter alia Geels and Schot 2007, Markard and Truffer 2008, Arora et al. 2015). Technology co-evolves with society and therefore needs to be put in a broader context (Geels 2005, Van der Brugge and Rotmans 2007). In the last century, technology as well as institutional and organisational arrangements have evolved around a centralised wastewater engineering paradigm, leading to shared values and a professional culture based on civil engineering competences (inter alia Truffer et al. 2010, Kiparsky et al. 2013, Fuenfschilling and Truffer 2014). This technological, institutional and organisational centralisation is accompanied by experts and policy circles which see centralisation as the supreme solution (Graham and Marvin 2001). As a result, WMS strongly resemble the systems installed back in the Victorian age in many high-income countries (Thomas and Ford 2005). However, innovative approaches promising wastewater management with similar performance to the centralised WMS are being developed with similar levels of technological readiness. So far, however, the fundamental criticisms about the lack of innovation in handling wastewater are being invoked, and radical alternative ways of managing wastewater are rarely discussed or implemented with the same vigour as centralised approaches. Decentralised systems, such as membrane bioreactor systems, have so far mainly been introduced in niches, where the traditional approach was undisputedly unfeasible or too costly (inter alia Dahlgren et al. 2013). Examples include alpine huts, cruise ships, holiday resorts and office buildings (cf. Goymann et al. 2008, Verrecht et al. 2012, Li et al. 2013). Decentralised systems thus tend mostly to be considered as a stopgap for niche applications (Eggimann et al. 2016c). We are consequently facing an innovation deficit or crisis in the wastewater sector which calls for a paradigm shift or a sustainability transition in UWM (inter alia Thomas and Ford 2005, Pahl-Worstl et al. 2011, Kiparsky et al 2013). Sustainability-related innovations with respect to socio-technical aspects can be conceptualised as *sustainability transitions* (cf. Markard et al. 2012, Truffer and Coenen 2012) which, within the context of this thesis, relates to more sustainable system alternatives to the centralised UWM paradigm. After all, the transition towards more sustainable UWM practices needs to be understood as an innovation problem, and technological innovation is not enough to handle it but must be supplemented by innovation with respect to the societal, organisational or institutional context (Truffer et al. 2013). Even though the requirements on technological treatment to minimise health risks and provide high standards of urban hygiene are given top priority in UWM, to ensure the successful long-term viability of technologies we must equally consider the costs, social acceptance, legal and institutional arrangements, business models, customer expectations etc. – in other words the whole socio-technical dimension which tends to be overlooked (inter alia Fane and Fane 2005, Li et al. 2013, Arora et al. 2015, West et al. 2016).

1.1.3 Decentralisation as a radical alternative to UWM

In the last few years, the predominant centralised UWM approach based on vast infrastructure networks has increasingly been questioned. New concepts such as '*low impact urban design and development*', '*sustainable urban drainage systems*', '*water sensitive urban design*' or '*integrated urban water management*' (cf. Fletcher et al. 2015) have been suggested, incorporating new approaches to infrastructure design and practices such as source separation, water reclamation and reuse or nutrient recycling (inter alia Daigger 2009, Bach et al. 2014, Fletcher et al. 2015, Sharma et al. 2016, SNF 2016). However, decentralised treatment with *decentralised* WMS is certainly the most radical alternative to centralised network-based WMS (inter alia Tchobanoglous et al. 2004, Massoud et al. 2009, Libralato et al. 2012, Larsen et al. 2013, OECD 2015) (see Box 1.2).

Box 1.2

Decentralised WMS

Historically, wastewater has commonly been disposed of in a decentralised fashion with low-tech sanitation systems such as pit latrines. Decentralised systems are still often understood to be septic tanks. In this thesis, the focus is laid on small-scale mechanical-biological treatment plants (Orth 2007), i.e. treatment technologies offering the same or similar performance and service to those of centralised WMS. The manifold alternatives of decentralised systems go beyond the scope of this thesis and reference is made to the literature for more information about possible technology options (inter alia Crites and Tchobanoglous 1998, Singh et al. 2015). See also Section 1.3.1 and Section 2.3.1 for further explanations about centralised and decentralised systems.

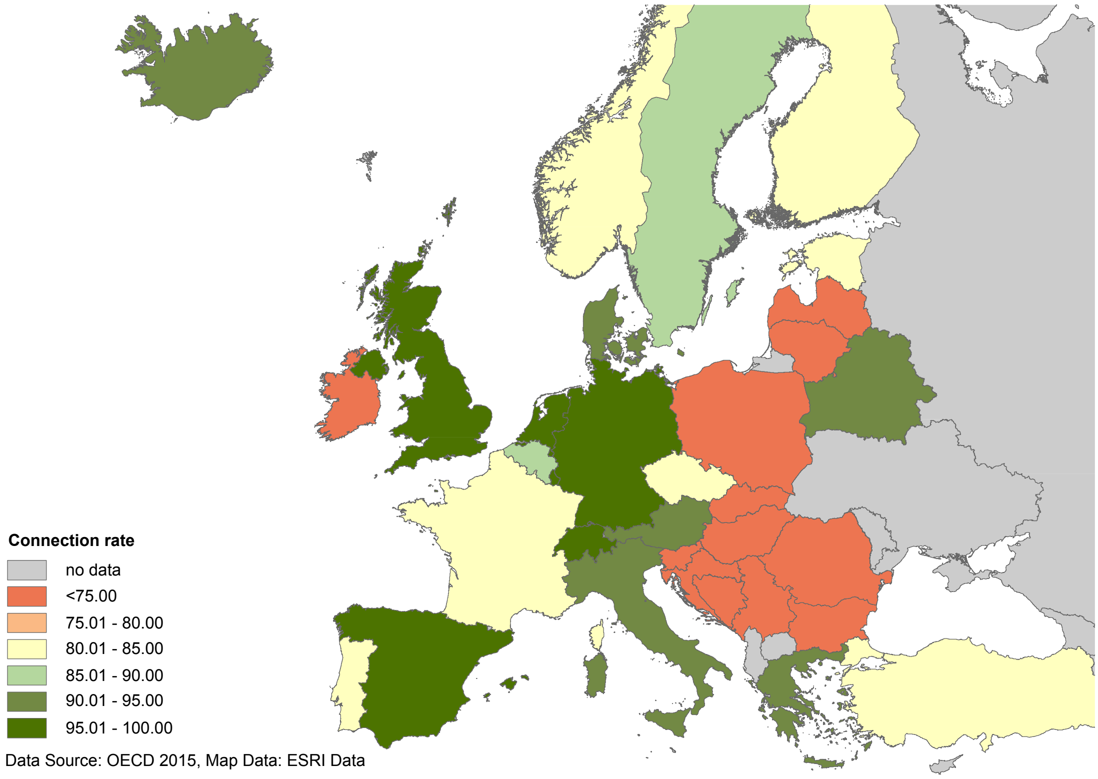


Figure 1.1: *Comparison of connection rates across European countries (own representation, data taken from OECD 2015). For a global overview see, Larsen et al. (2016).*

Great global differences exist with respect to the dominance of the centralised (or decentralised) wastewater planning paradigm. The central (wastewater) connection rate, i.e. the proportion of a population connected to a centralised WMS, is very high in many high-income countries such as Switzerland, and reaches connection rates close to 100% (UN 2015b). In low-income countries, connection rates are often low because large proportions of the population still lack basic sanitation (WHO and UN-Water 2014). However, considerable differences can be found even within OECD countries, with some countries having relatively lower connection rates of between 60 and 70% (see Fig. 1.1). However, it is important to note that lower connection rates do not necessarily represent a lower standard of sanitation, but can also express a higher share of decentralised WMS. Decentral approaches to sanitation are often perceived as rudiments dating from former times, when central sewer connections were non-existent. However, this is a problematic view, because decentralised WMS are also successfully being applied on a larger scale today (e.g. *johkasou* systems in Japan) including on new construction sites especially in high-income and high-density European Union countries or the USA (cf. Yang et al. 2011, OECD 2010/2015). Also, the reasons for implementing decentralised WMS may differ depending on the context. Nevertheless, decentralised WMS constitute a radical alternative to centralised WMS, providing an opportunity to transform existing systems in anticipation of achieving overall lower costs to society.

1.1.4 The challenge of determining the lowest cost connection rates

Costs play an important role in WMS infrastructure planning. Lowest-cost UWM configurations can essentially be found by solving the basic trade-off between the cost of transportation of sewage and the expenses incurred in treating wastewater in suitable wastewater treatment plants (WWTP). As shown later in this thesis, decentralised WMS are characterised by very different cost characteristics than centralised ones, and the question remains whether the introduction of decentralised WMS makes sense from an economic point of view, and more importantly, to what extent. This economic question is truly important, as the future strength of our economies depends on the financing of their infrastructures (inter alia Hansman et al. 2006). Despite the increasing attention focused on decentralised approaches in UWM, however, it remains unclear to what degree centralised or decentralised WMS should best be installed in a particular region to provide UWM services at lowest cost to society. This is surprising, as this question and trade-off mentioned above have already been discussed in academia for many decades (inter alia Downing 1969, Converse 1972, Hovey et al. 1977, Abd el Gawad and Butter 1995, Ambros 1996, Van Afferden et al. 2015).

The main objective of this thesis is therefore to push the understanding of the optimal degree of centralisation (ODC) for WMS from an economic point of view. Although interests focused here on economic challenges, a major role is certainly also played by social, environmental or technical challenges (cf. Brears 2014, Section 5.1). Nevertheless, this focus on economic aspects follows the generally observed trend of an increased public focus on cost efficiency, which is especially relevant in view of the highly capital intensive UWM infrastructures (Knops 2008, Fuenfschilling and Truffer 2014, Hall et al. 2016). Optimising and improving our knowledge about UWM planning and preventing unwise investments in costly infrastructures is especially urgent given today's under-investment, shrinking subsidies and ageing infrastructures (inter alia Maurer and Herlyn 2006, WEF 2010, ASCE 2011) (see Section 1.4.1).

1.2 Goals and research questions

The main goals of this doctoral thesis are the promotion of geospatial modelling tools and the development of an economic argument for a possible sustainability transition towards more sustainable WMS. The overall aim is to provide insights about economically sustainable UWM and to develop tools and methods which help to optimise the planning of sanitation infrastructures. The chosen research focus is driven by the following overarching and generic question:

What is the optimal degree of centralisation for wastewater infrastructures?

A single dissertation clearly falls short of addressing this extensive research question in all its facets, which certainly is not the aim here. The particular focus is on the economic geospatial modelling and planning aspects of network-based wastewater infrastructures, especially within a Swiss context.[1] Within this chosen focus, decentralised and hybrid WMS (as compared to centralised WMS) are of primary interest because fewer economic assessments and examples of geospatial modelling are so far available for hybrid WMS (cf. Section 1.5).

The overarching research question is addressed by several more specific but closely related questions (see also Section 1.6 for research questions for each chapter). They can be summarised as follows:

- *How can the degree of centralisation be defined?*
- *How does space influence the cost of centralised, decentralised and hybrid WMS?*
- *How can spatial cost influences be modelled?*
- *How do economies of scale, density effects and costs of transportation interact at settlement level?*
- *How can heuristic geospatial modelling tools help to determine the optimal number, location and sizing of wastewater management systems in a region from an economic point of view?*
- *What role do institutional and organisational settings play in reaching optimal connection rates?*

[1] All selected case studies are from various Swiss regions. The practical relevance of the question about the optimal degree of centralisation may naturally differ depending on the context. Although a Swiss perspective is chosen in this thesis, the Swiss context is not unique and lessons can be learnt for similar contexts in other countries.

1.3 UWM planning and the ODC

1.3.1 Degree of centralisation

The terminology of *centralisation* and *decentralisation* is most commonly used in relation to institutional theory and political sciences. In these contexts, decentralisation is broadly understood as a '*transfer of authority from central to local government*' (Oxford Dictionaries 2015) and centralisation is the opposite, namely a transfer from local to central government. This thesis refers to a different kind of decentralisation relating to technology and infrastructure, i.e. *technological (de)centralisation*. Technological centralisation is loosely defined here as a shift from distributed to concentrated modes of production and consumption of goods and services. This shift is usually accompanied by technological transformations, i.e. various technologies are used for centralised or decentralised systems. Whereas political decentralisation was anticipated as a megatrend, especially in the 1980s (cf. Naisbitt 1982), decentralisation of production and consumption has only recently been identified as a trend, fuelled by various technological advances such as the internet of things and modularisation (cf. Dahlgren et al. 2013, Ignaczak 2014, Greengard 2015).

Eggimann et al. (2016c) distinguish between three main approaches in UWM which constitute a fundamentally different alternative to conventional centralised UWM, namely *on-site water treatment* (Fewtrell et al. 2005, Peter-Varbanets et al. 2009), *on-site water reuse* (Guo and Englehardt 2015) and *on-site wastewater treatment* (Brands 2014, Singh et al. 2015). They all share the characteristic of being decentralised modes of service provision in UWM. Sanitation services in a region may thus in principle be provided by centralised WMS or decentralised on-site WMS (Orth 2007, Libralato et al. 2012).[2] However, a clear distinction between centralised and decentralised WMS is problematic because of its scale-dependence. Reference is made to the literature for terminological issues (e.g. Crites and Tchobanoglous 1998, Tchobanoglous and Leverenz 2013, Singh et al. 2015) (see also Section 2.3.1 for an elaborate discussion on this topic). This thesis generally assumes the use of commercially available package treatment plants for decentralised on-site WMS. Such plants typically range from just a few to several hundred population equivalents. Finally, a *hybrid* WMS consists of both centralised and decentralised WMS (inter alia Tchobanoglous and Leverenz 2013, Poustie et al. 2014, Sapkota et al. 2015).[3]

---

[2] In the literature, the terms *decentralised*, *on-site* are often used interchangeably.

[3] In the literature, the terms *dispersed* or *distributed* are often used interchangeably.

Finally, the optimal dimensioning, (geographical) placement and deciding on the number of facilities (known as the facility location problem) is an old issue and represents a basic research challenge for many different applications (inter alia Current et al. 2002, Daskin 2013). Other sectors also depend on physical network infrastructures, such as district heating or electricity. However, the challenge of determining the optimal service provision, also exists for services where no physically built infrastructure network is involved, such as the placement and dimensioning of schools, hospitals or as biomass composting plants. Table 1.1 gives an exemplary overview including literature from very diverse fields to argue that estimating the ODC is not restricted to the field of UWM but is a truly generic and broadly relevant question.[4]

1.3.2 Setting the planning context of UWM systems

Over time, different aspects have been emphasised in the literature on UWM infrastructure planning, favouring either centralised or decentralised approaches. From early times onwards, the primary goal of UWM engineering was to provide urban hygiene by evacuating storm and wastewater through drainage and sewer networks to prevent cholera or typhoid outbreaks (O'Flaherty 2005, Sedlak 2014, Urich and Rauch 2014a). While urban hygiene was the initial motivation for investing in UWM infrastructures, economic considerations have always been present in planning because of the limitation of financial resources. There has been considerable scepticism about the development of treatment costs as well as a strong trust in economies of scale for wastewater treatment, and these were and still are used today to justify an '*upgrade and expansion*' practice (Moss 2001). Thus Deininger and Su (1973) wrote 40 years ago that '*in general, no great breakthrough in sewage treatment technology can be expected, at least not cost-wise*' and that regional wastewater treatment is highly favoured because of economies of scale. The scientific UWM community thus focused heavily on centralisation for many decades (inter alia Moss 2001, Graham and Marvin 2001, Fane and Fane 2005, Dominguez 2008, Kiparsky et al. 2013, Fuenfschilling and Truffer 2014, Lieberherr and Truffer 2015, Lieberherr and Fuenfschilling 2016, cf. Section 1.3.2). However, this centralisation and a focus on the safe removal of waste and storm water as the primary criteria was not a linear process, and a strong centralisation trend can be observed especially in the second half of the 20th century. In Fig. 1.2 the development of the connection rate is shown over time in a Swiss context. Because of this strong focus on economies

[4] I personally feel that the topic of wastewater, infrastructure planning and specifically UWM has been neglected by my own academic field of geography (cf. Jewitt 2011). This is surprising, as UWM has a strong spatial component and geospatial modelling techniques as well as GIS are increasingly applied to it.

Table 1.1: *Overview of possible application fields of central versus decentral service provision. The table is partly inspired by Markard (2011). All examples given were selected because they explicitly raised the question of optimal DC.*

Sector	Service	Exemplary Literature
Energy[A]	District heating and cooling	Möller and Lund 2010, Gils et al. 2013, Nielsen and Möller 2013, Stennikov and Iakimetc 2016
	Electricity generation (e.g. hydro power, solar power, wind power…)	Zvoleff et al. 2009, Johnson and Ogden 2012, Johnson et al. 2008, Kocaman et al. 2012, Levin and Thomas 2012, Sanoh et al. 2012/2014, Parshall et al. 2009, Deichmann et al. 2011, Kaundinya et al. 2013, Hins et al. 2015
	Fuel distribution (e.g. gas, hydrogen, petrol/oil, pellets)	Yang and Ogden 2007, Baufumé et al. 2013, Johnson and Ogden 2012, Stiller et al. 2010
	Air conditioning	Franzske et al. 2003, Bhatia 2016
Water[A]	Drinking water supply	Wenban-Smith 2009, Marques and Witte 2010, Newman et al. 2014, Sapkota et al. 2015
	Wastewater treatment	Downing 1969, Converse 1972, Deininger and Su 1973, Abd El Gawad and Butter 1995, Starkl et al. 2012, Zeferino et al. 2012, Libralato et al. 2012, Lee et al. 2013, Poustie et al. 2014, Morera et al. 2015, Arora et al. 2015, Van Afferden et al. 2015, Baron et al. 2015, Eggimann 2013, Eggimann et al. 2015/2016a/b, Cornejo et al. 2016
	Urine or faecal sludge collection	Flotats et al. 2009, Kennedy-Walker et al. 2014, Marufuzzaman et al. 2015, Tilley 2016
	Water disinfection	Mintz et al. 2001
	Desalination	Shahabi et al. 2015
	Rainwater reclamation or harvesting, wastewater reuse	Friedler and Hadari 2006, Woods et al. 2013, Guo and Englehardt 2015, Alnouri et al. 2015, Guo et al. 2016
Waste[A],[B]	Solid waste disposal	Nakou et al. 2014
Education[B]	Schooling (e.g. universities, libraries)	Current et al. 2002
Health[B]	Elderly care, child care	Floyd et al. 2013
Security[B]	Fire emergency, police	Daskin 2013
Other[B]	Biomass composting	Zurbrügg et al. 2004, Perpiñá et al. 2009
	Cleaning	Christoffersen et al. 2007
	...	...

[A] Services which are provided by physical network infrastructures for centralised systems. Depending on the service, however, alternative decentralised technologies may not depend on physically built network infrastructures.

[B] Services which are provided without a physical network infrastructure for both central and decentralised systems. However, the service provision may depend heavily on the street network.

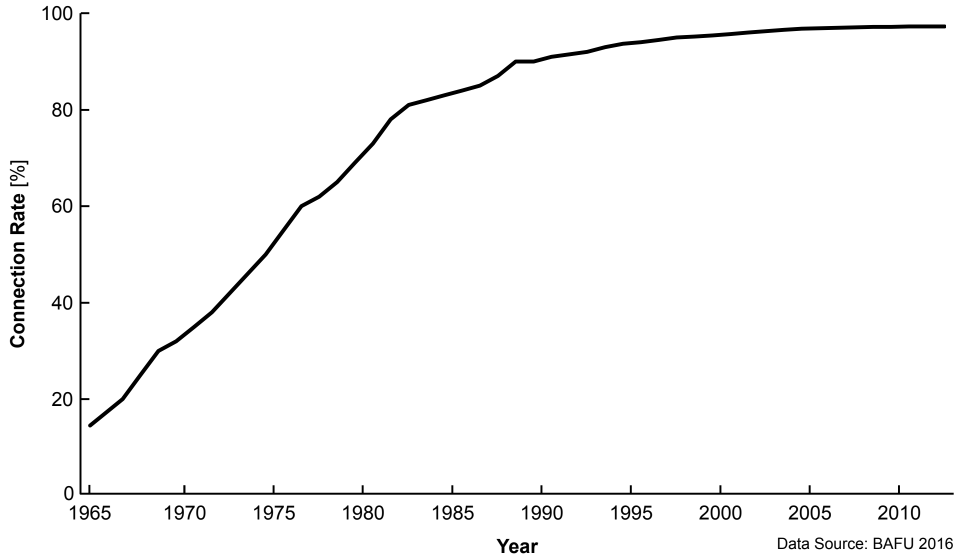


Figure 1.2: *Development of the central connection rate for Switzerland in the last 50 years.*

of scale, centralisation generally went along with publicly owned infrastructures and the introduction of regulations such as the mandatory connection rule to strengthen and protect the centralised planning paradigm. Switzerland has issued mandatory connection rules, like other countries such as Austria or Germany, and the individual cantons are responsible for wastewater treatment according to Article 11 of the Federal Act on the Protection of Waters (1991) (see Box 1.3). A central sewer connection is thus compulsory if it is '*expedient*' and '*reasonable*' (VSA 2005) (see Eggimann 2013 for more information).

Since the 1980s, the central values of network infrastructure planning shifted increasingly towards commercialisation, privatisation and economic efficiency (inter alia Knops 2008, Fuenfschilling and Truffer 2014). The view of networks as natural monopolies has been weakened and the introduction of competition as a consequence of liberalisation and deregulation is increasingly considered as feasible (inter alia Seidenstat 2000, Knops 2008, Lieberherr and Fuenfschilling 2016). Within the scope of this transformation, decentral approaches and determining the ODC are gaining relevance because different forms of WMS are conceivable as well as the potential of providing lower UWM costs to society. Determining the ODC is arousing interest because decentralised WMS can today be increasingly considered as fully-fledged technological substitutes for centralised WMS. The previous scepticisms of the scientific community with respect to decentralisation in UWM has increasingly given way to more optimistic views in recent publications (inter alia Tjandraatmadja et al. 2005, Libralato et al 2012, Larsen et al. 2009/2013, Poustie et al. 2014, Guo et al. 2014, Sapkota et al. 2015, Guo and Englehardt 2015, Reymond et al. 2016). Accordingly, decentralisation is no longer seen in contradiction

Box 1.3

Mandatory connection regulation

Art. 11 Duty to connect to sewers and to accept polluted waste water

1. Polluted waste water which originates in an area served by public sewers shall be discharged into such sewers.
2. The areas served by public sewers shall include:
   a. building zones;
   b. other zones as soon as they are connected to the public sewers (Art. 10 para. 1, let. b);
   c. other zones where connection to the public sewers is expedient and reasonable.
3. The person responsible for the sewers is obliged to accept waste water and convey it to the appropriate central waste water treatment plant.

to centralisation but is rather thought of as its complement on the basis of a way of thinking more akin to hybrid systems (inter alia Gleick 2003, Libralato et al 2012, Marlow et al. 2013). In Switzerland, centralisation is very strong indeed, now reaching connection rates close to 100% (BAFU 2016). However, some particularly rural areas are starting to explore more hybrid WMS. Besides this increasing interest in hybrid systems thinking, more extreme concepts can be identified in today's literature, such as *autarky* or *footloose living* (Hamilton et al. 2004), especially in energy-related literature (inter alia Funcke and Bauknecht 2016) but increasingly also in the UWM literature (inter alia Larsen et al. 2013). These views tend to value independence from centralised infrastructures, for reasons such as vulnerability to terrorism or personal lifestyle preferences such as self-sufficiency (inter alia Vannini and Taggart 2013). However, contrasting concepts such as multi-utility infrastructures are also found: instead of focusing on single infrastructure networks, their authors plead for multi-utility infrastructures in which different infrastructures are combined and interlinked (inter alia Karaca et al. 2013a/2013b/2015). Undisputedly, such concepts generally strengthen centralised approaches as reliance on networked infrastructures increases.

Over the last 50 years not only have decentralised WMS been increasingly taken seriously, but the focus on economic efficiency is also moving towards broader approaches. More and more multiple criteria, such as greenhouse gas emissions or resource recovery, are included in an integrated fashion in the cost analysis and the call for integrated planning and environmental sustainability is getting louder (inter alia Brown et al. 2009, Lee et al. 2013, Lienhoop et al. 2014, Morera et al. 2015, Hendrickson et al. 2015, Baron et al. 2015, Grant 2016, Cornejo et al. 2016). Certainly, the context of today's UWM planning can only partly be described in such a simplified linear way as was done in this section. Multiple planning trends with different emphases may prevail at the same time in the same place for different actors and may even contradict each other, adding to the complexity of planning for an optimal mix of central and decentral WMS.

1.3.3 Planning challenges relating to networks and cost

The economies of scale already mentioned in Section 1.3.2 for wastewater treatment in the centralised WMS approach have been (and still are) probably the most important (economic) argument in favour of centralisation (cf. Townend 1959, Downing 1969, Adams et al.1972, Maurer et al. 2006/2010). However, there are also strong economic arguments in favour of decentralised approaches. The reasons for the attractiveness of decentralised approaches are closely linked to the specific challenges of network-based service provision, which are briefly outlined below (see Fig. 1.3).

First of all, network infrastructures show considerable *lock-in* tendencies (Arthur 1989), i.e. infrastructure investment decisions can lock-in the development paths of whole societies (Hall et al. 2016). Typically, networks in UWM have grown over time and decisions made in the past affect today's decision space, resulting in path dependencies (Herder and Wijnia 2012). In other words, in the case of switching from centralised to decentralised systems (or the other way round), already built infrastructures may become obsolete and constitute sunk investments. So these sunk investments become part of the *transaction costs* in the case of a system transformation, as the existing infrastructure must be included in the cost calculation (Marlow et al. 2013). Sunk costs are so prominent for WMS because of the high durability of the infrastructures: typical infrastructure

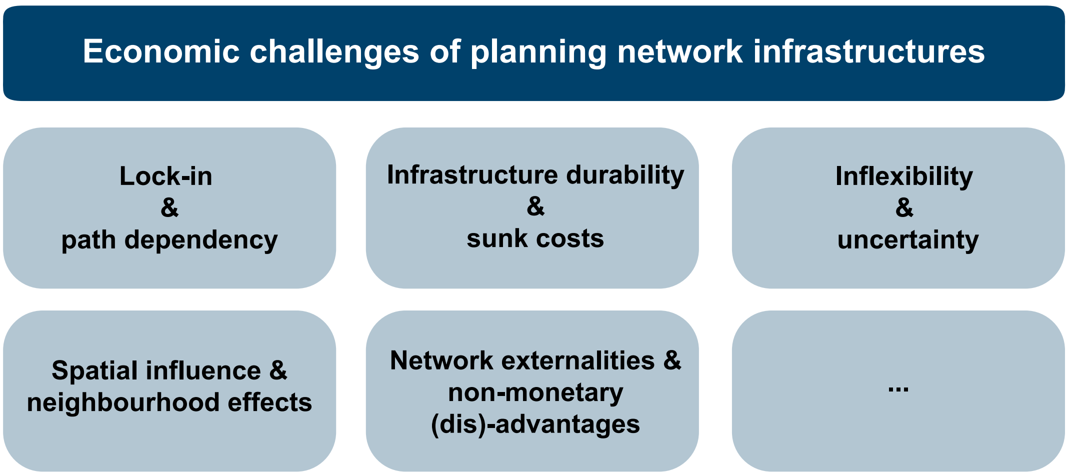


Figure 1.3: *Key challenges of planning network infrastructures relating particularly to costs and network characteristics.*

lifetimes used for depreciation are around 25 years for WWTP and around 80 years for sewers (inter alia Maurer et al. 2005). This long life expectancy of the core technological components allows depreciating initial investments over long periods (Markard 2011). However, such high asset durability in combination with high capital intensity also results in very high upfront investments and the potential risk of sunk costs, high idle capacities and the need for planning for very different context conditions (Dominguez and Gujer 2006, Dominguez 2008, Truffer et al. 2013). Urich and Rauch (2014) for example write:

> *'Traditionally, infrastructure design is based on the assumption that key drivers for the urban water infrastructure, such as population growth, water demand and climate change impacts, can be predicted 30 to 50 years into the future. Experience with infrastructure built on these assumptions has revealed it can lead to problematic designs and decisions.'*

To avoid problematic designs means aiming at *adequate* infrastructures, i.e. ensuring that no money is wasted by making unnecessarily high investments in infrastructures which are not used to their full capacity (Knops 2008). Population dynamics heavily affect sanitation infrastructure planning and complicate matters, especially because of the geographical rootedness of sewer networks and because infrastructure networks are inflexible once they have been built. A specific strand of literature has evolved around flexibility planning and engineering under conditions of uncertainty (e.g. with respect to population growth or population shrinkage, new technologies…) (inter alia Siedentop and Fina 2010, De Neufville and Scholtes 2011, Urich and Rauch 2014b). In many parts of the world, the growth paradigm is traditional, but the population of many European cities and regions is declining. This decline typically results in oversized infrastructures and effects cost efficiency because networks are under-utilized (inter alia Schiller and Siedentop 2005, Schiller 2010). Dominguez and Gujer (2006) illustrate for a Swiss context how challenging it is to plan optimal infrastructures because of the difficulty of predicting future developments.

Moreover, it is challenging to assess the costs of UWM infrastructure networks because of *network externalities* and *non-monetary (dis-)advantages*. According to different generic rules formulated e.g. by Melcafe (Doyle 2011) and Reeds (Cushman 2010), a typical characteristic of networks is that their value does not increase in a linear fashion with the number of people connected. Following this logic, investing in larger networks consequently yields greater value. Hansman et al. (2006) write that '*connecting more people to a network increases its utility. However, significant costs – those of managing, protecting, and making the system reliable*

– may increase more than linearly with network size and scope. Thus, networks may have an optimal scale […].' It is a non-trivial task to monetarise such different networks effects or the different non-monetary advantages or disadvantages of centralised or decentralised WMS. Examples of advantages of networks over decentralised options which are difficult to monetarise include easier control and monitoring, exposure to vandalism and theft or vulnerability and resilience (inter alia Libralato et al. 2012, Starkl et al. 2012). The various different (dis)advantages of centralised and decentralised WMS have been discussed in the literature, although often with a vagueness as regards specific economic effects (inter alia Gikas and Tchobanoglous 2009, Libralato et al. 2012).

Finally and most importantly, UWM planning is challenging because of spatial influences and neighbourhood cost effects. The geographic dimension is inherent to infrastructure planning in UWM, and overall regional sanitation costs depend on geographical factors such as topography, geological characteristics, climate and spatial population distribution (inter alia Zvoleff et al. 2009, Eggimann et al. 2015, Wenban-Smith 2009). As shown in this thesis, it is challenging to assess the costs to single households for both centralised and decentralised WMS: in both cases, the costs depend strongly on the choices and spatial distribution of neighbouring households and their position with respect to other households (cf. Eggimann et al. 2015/2016a/b).

All these different challenges facing network infrastructures clearly show the complexity of determining the optimal degree of centralisation which would need to be considered to fully address the research questions presented here. The research approach selected in the following chapter provides more background on how these issues are addressed within this thesis.

1.4 Research approach

1.4.1 The potential of a model-based approach

Within this doctoral thesis, a model-based approach is chosen to approach the multi-faceted challenge of assessing the optimal mixing rates of centralised and decentralised WMS. More specifically, an explicit geospatial optimisation framework with respect to overall regional costs is developed. The philosophy behind this approach is not to determine *theoretically* optimal solutions and focus on detailed engineering aspects. Rather, the approach chosen in this thesis is *pragmatic*, meaning that the goal is to provide approximate solutions which are however grounded in concrete case studies. Nevertheless, the primary motivation for choosing a model-based approach (rather than opting for detailed engineering studies) is much more fundamental, namely to achieve *understanding* and *the ability to make predictions* (inter alia Sitte 2009). Models help to structure and formalise problems, allowing the understanding of the problem to be improved and serving as a learning environment. Models help to conceptualise complex systems into simpler systems by abstracting the key elements. Moreover, they also allow *hypotheses to be tested*, i.e. specifically for this thesis the hypothesis relating to costs and the spatial differentiation of the ODC. By assessing and visualizing the ODC for different cost assumptions, the modelling framework presented here helps to identify potential paths of technology diffusion. Another important aim and motivation for modelling is to provide input for planning practices and to support the policy making processes (cf. Pelzer 2015). In the context of this thesis, successful models become useful planning support systems for planners and policymakers and promote a dialogue about more sustainable infrastructure planning. This is doubtlessly particularly true if they provide spatially explicit visual results and are easily applicable to specific case studies. Economic geospatial modelling is additionally powerful because despite all the problematic aspects of focusing on economics, outputs given in cost terms are presumably most intuitively obvious for decision makers.

Moreover, choosing a model-based approach obviously has shortcomings as well. The strength of conceptualising complex systems into simpler systems also means that the multi-dimensionality and complexity of the research question always obliges model-builders to decide actively about broad simplifications of their model (i.e. to make decisions about leaving or ignoring certain aspects) (see also Section 5.1). Models therefore always reflect the model-builder's perspective (inter alia Van den Ven 2007). In this thesis, the most obvious model simplification is the reduction of economic geospatial properties. Furthermore, very strong assumptions are made in the modelling process such as that individuals will always make rational choices on the basis of being perfectly well informed about costs and pick the lowest-cost WMS to maximise their overall profits (cf. Calhoun 2002). Finally, the use of computers and models for planning and decision making remains challenging in practice, despite all the listed potentials (inter alia Brail and Klosterman 2001, Brail 2008, Pelzer 2015) and '*better computer support does not automatically imply a better decision*' (Cortes 2000).

1.4.2 Modelling the ODC in UWM

Extensive computational model-building has been going on in UWM since the rise of computers, which now play an important role in the design and management of WMS (inter alia Urich and Rauch 2014b, Bach et al. 2014). The objective functions of many modelling approaches are similar, differing only in their emphasis depending on the aim and model context (see Box 1.4). The economic focus of the approach chosen in this thesis is therefore not completely new. Also, some model-based approaches include economic and spatial criteria: Geographical Information Systems (GIS) are typically used for multi-criteria spatial decision analysis, especially for locating optimal placement sites for treatment plants (inter alia Makropoulos et al. 2007, Zhao et al. 2009, Deepa and Krishnaveni 2012). Additionally, virtual (not in 'true' geographical space) case studies have been suggested to explore UWM-related questions (Sitzenfrei 2010). Furthermore, numerous approaches exist to model pipes in a geospatially explicit way for transporting water and wastewater either by designing actual network layouts or estimating material stock by virtual network layouts (inter alia Greene et al. 1999, Urich et al. 2010, Brand and Ostfeld 2011, Blumensaat et al. 2012, Bieupoude et al. 2012, Sitzenfrei et al. 2013b, Maurer et al. 2013, Bach et al. 2014, Sitzenfrei 2016). Many modelling approaches are also applied to different types of fluids such as gas, hydrogen or oil (Marcoulaki et al. 2012, Baufumé et al. 2013). Commonly, network layouts are generated with aid of shortest-path algorithms, topographic details and the existing road network because of the correlation between

Box 1.4

Objective functions

De Melo and Câmara (1994) exemplary list in addition to achieving minimum economic costs the following criteria:

- Minimize the environmental impact
- Maximize the system reliability
- Maximize the system flexibility under uncertain conditions
- Assure equity between system users
- Maximize the benefits of reusing treated effluent

sewer-network placement and the existing road network (Haile 2009). In other words, numerous modelling approaches already deal with aspects relating to optimal UWM infrastructure design.

However, with some notable exceptions, the models applicable to real-world case studies designed to find ODC from a cost-optimisation point of view considering transportation and treatment of wastewater are limited (cf. Section 1.4.2). Among these exceptions, two categories can be differentiated: either only a small number of alternative case studies are generally ranked as a means of deciding on the best solution (inter alia Abd el Gawad and Butter 1995, Lee et al. 2013, Van Afferden et al. 2015), or different optimisation techniques are applied for generating and choosing the best system alternative(s) (inter alia Leitão et al. 2005, Zeferino et al. 2012). Infrastructure system layouts in these examples are generally optimised with respect to both the cost of construction and operation of sewer networks[5] and the costs of treatment in WWTPs.[6] These studies, which are also based on an optimisation focus with respect to centralised or decentralised WMS, make it obvious that it is a major challenge to assess the single most optimal hybrid WMS configuration within a region. Heuristic optimisation approaches such as simulated annealing, evolutionary algorithms and hierarchical clustering are consequently often used to determine approximate solutions. Heuristic approaches using detailed cost or geographic information as their input have long been suggested as the way forward in sanitation planning, because the *theoretical* optimum is usually not of primary interest and finding approximate solutions is already a big step forward (inter alia de Melo and Câmara 1994, Eggimann et al. 2015).

[5] Wastewater is bulky and heavy at source and depends on geography. High costs may be accrued for transportation, especially if pumping is necessary. Pumping costs are commonly included with terrain complexity, and sewers are built with respect to trench depths (or a constant trench depth is assumed) and/or the constraints of flow velocities.

[6] The detailed (hydraulic) design is generally not the focus of these modelling approaches and they do not have the same purpose as a detailed engineering analysis.

1.5 Relevance

The management and planning of network-based infrastructures is relevant from both scientific and practical points of view. The scientific relevance for academia is discussed in (Section 1.5.1), the practical relevance for society as a whole in (Section 1.5.2).

1.5.1 Scientific relevance

Elucidating OCR is of scientific relevance for very diverse reasons. The most interesting and relevant conceptual innovations of this thesis for the scientific community are threefold, namely i.) spatially explicit full-cost modelling, including several returns to scale, ii.) the socio-technical conceptualisation of WMS, and iii.) the dynamic approach to UWM.

i.) *Spatially explicit full-cost modelling including several returns to scale*: The identification and consideration of the most important cost factors of central, decentral and hybrid WMS in space is of key scientific interest because the complexity of considering different cost elements has so far often been neglected. However, by including the complexity of a full-cost analysis, we can move forward in our understanding of the 'true' cost efficiency of UWM infrastructure layouts. Even though cost arguments generally play an important role in decision-making, so that more sophisticated cost analyses are essential, the economic theory of UWM with respect to deciding between central and decentral WMS has so far not been fully understood. Within this dissertation, several gaps are identified and addressed in the underlying economic theoretical framework used to assess the potential of central or decentral WMS: *First*, the explicit geospatial trade-off between the cost of sewage transportation and wastewater treatment considering (dis)-economies of scale has been poorly understood. On this point, much of the literature has focused particularly on economies of scale in treatment while neglecting diseconomies of scale in sewer construction (inter alia Townend 1959, Downing 1969, Adams et al. 1972, Deininger and Su 1973). This thesis shows the scientific relevance of including several non-linear cost effects in the cost analysis, such as (dis)-economies of scale or density effects. So far, modelling approaches considering the trade-off between the costs of the sewer networks and the treatment costs have mostly been abstract, making a contribution principally from a theoretical point of view based on simple case study designs. Models which allow real-world applications to determine (near) optimal solutions and go beyond comparing a limited number of system alternatives are scarce. This thesis therefore shows the relevance of considering full network costs (e.g. modelling costs with respect to the position in the network), topography and hybrid system constellations. *Second*, considering decentralised WMS truly adds to the complexity of UWM planning because the costs of decentralised sanitation systems, particularly those relating to operation and maintenance, have so far not been well understood. Thus Truffer et al. (2013) write that the costs of OST systems depend on '*economies of repetition*', i.e. on the number of installations within a specific neighbourhood. Prior to this thesis, however a coherent framework that could be used to assess such spatially dependent costs has however been lacking. *Third*, the interplay between centralised and decentralised WMS in particular is not well understood with respect to costs, i.e. research on the dynamic interplay of the costs of centralised and decentralised sanitation systems (hybrid systems) is scarce.

ii.) *Socio-technical conceptualisation of WMS*: So far, the scientific community has acquired only a limited understanding of how technical, political and economic factors interact (Hansman et al. 2006), and little is known about institutional dimensions and appropriate governance transformation strategies (inter alia Bolton and Foxon 2015). The research focus of this thesis uses a socio-technical understanding of WMS to address these research questions and highlights the relevance of conceptualising sanitation systems as socio-technical entities, i.e. not arguing from a technical perspective alone. However, it is of similar importance to address the role of institutions (cf. Geels 2006) and organisations with respect to the ODC. This is particularly true as both technological innovations and those relating to the institutional UWM framework are important in the field of sanitation (Kiparsky et al. 2013). Later in this thesis we will once again prove the relevance of analysing the co-evolution of institutions and technologies.

iii.) *Dynamic considerations of UWM:* WMS need to be understood and modelled as dynamic systems, i.e. a static consideration falls short of providing optimal solutions in the long term. This thesis highlights the fact that the scientific community should carefully consider innovations in technological systems (e.g. with respect to cost developments and technologies), thus enabling new modes of service production. Such considerations give way to a perspective taking into account the transitions of UWM infrastructure systems towards new and hopefully more sustainable configurations. One important aspect of such a non-static approach, which has become possible on the basis of this thesis, is the mapping of possible transitions. Models and geospatial mapping techniques are of great help in decision-making, as they can be used to test and visualise hypotheses. Within sustainability transition research, the '*geography of transition*' approach focuses explicitly on the geographic or spatial aspects of transitions (inter alia Coenen and Truffer 2012). However, even though very different models have been developed to simulate the phenomenon of technological transitions and the diffusion of technology (cf. Zeppini et al. 2014, Haan et al. 2016), they are often not space-specific. So there is in particular a research deficit in the spatial aspects and modelling approaches of mapping transitions (inter alia Smith et al. 2010). Despite the abundant literature on technological diffusion (inter alia Rogers 1983, Grübler 1990), little knowledge is available about the spatially explicit diffusion of WMS. Too often, UWM and similar systems are conceptualised in purely static terms and lack on explorative focus on when and where these systems are bound to change. Finally, it would be highly relevant to the scientific approach to seriously start considering the implications of possible significant cost decreases (or scenarios where costs would even fall below those of centralised systems) of decentralised systems and not to exclude this possibility in advance.

This thesis thus pleads for a conceptual approach that considers the three dimensions of analysing (UWM) infrastructures outlined above. Such an approach to *socio-technical infrastructure transformation* would certainly be a fruitful way to address further research needs (see Section 5.1) and could be applied to other infrastructure sectors. The overarching significance of such generic research questions, i.e. their transferability to other service provision sectors and the wider field of UWM, appears promising. In UWM, the question of the ODC or optimal scaling is asked not only with respect to wastewater treatment but also to the drinking water supply, water disinfection, desalination or rainwater harvesting and reuse. Section 1.3.1 provides many further examples for alternative sectors which face similar challenging questions about the basic choice and scaling between central and decentral technologies.[7]

[7] So far, little effort has made to contrast these different socio-technical systems in order to draw lessons for UWM. This comparison carried out at different levels: examples would be to contrast different modelling techniques (e.g. with respect to optimisation or network modelling) or the regulatory frameworks of other sectors where mandatory connection rules also exist, to see how the decision-making takes place about who is allowed or denied the option of going off-grid (e.g. for district heating). A further example would be to compare the role of costs in decision-making and how other sectors go about this with respect to an approach.

1.5.2 Practical relevance

Investments in UWM infrastructure need to be carefully planned, and there are various main practical reasons for elucidating the ODC in the context of this thesis. Economic sustainability is of particular practical relevance due to the capital intensity of the UWM infrastructure, so that even minor optimisations can potentially save tremendous financial resources. Furthermore, decentralisation serves as a potential alternative for tackling some of the ecological limitations of the traditional centralised approach (e.g. leaking pipes, combined sewer overflows, etc.) (inter alia Larsen et al. 2013/2016). In practice, detailed engineering studies are needed to determine ODCs. However, there is also a practical need for tools to screen potential ODCs in order to improve our understanding of the possible diffusion paths of spatial technology. The knowledge of where central or decentral systems are most suitable would be of help in practice, especially if the cost savings for alternative system layouts can be quantified. These cost estimates help to set the potential cost savings in context, with all the potential disadvantages resulting from a system transformation. However, the practical relevance for determining optimal WMS differs strongly depending on the context: For example in communities with already built centralised infrastructures (such as in east Germany or parts of the Alps) decision-makers may need to consider whether to start decentralising such highly centralised infrastructures (devolution) because of financial pressures or depopulation (inter alia Schiller 2010, Schiller and Siedentop 2005, Siedentop and Fina 2010). In other contexts, however, decision-makers may consider extending the existing infrastructure (or building simplified sewerage systems), or constructing new sewer networks and consequently increase the degree of centralisation (inter alia Bakalian et al. 1993, Morera et al. 2015, Dendup and Tshering 2015). Finally, new system configurations are increasingly becoming possible thanks to mass production, dynamic economies of scale and the development of new sensors and these promise to be cost-competitive in different contexts (Dahlgren et al. 2013, Eggimann et al. 2016b/c).

This thesis focuses on Swiss case studies with existing highly centralised network infrastructures where such provision has proved successful in the recent decades. However, decentralised WMS provide an opportunity to reconsider today's approaches to UWM planning. In Switzerland, the replacement value of the total publicly and privately owned UWM infrastructure is estimated to be around US$ 61 billion or US$ 8000 per person.[8] Switzerland has relatively high per capita replacement values in an international comparison. However, these replacement values are usually also several thousand US$ per person in other countries (Maurer et al. 2005) and therefore constitute one of the highest public infrastructures costs (cf. Schalcher et al. 2011). Much of the Swiss wastewater infrastructure was built in the economic boom of the 1960s, 70s, and 80s and was heavily subsidised (Müller and Kramer 2000, cf. Fig. 1.2). As a consequence, in the next few years Switzerland (and other countries with a similar infrastructure history) will enter a new investment cycle because of the need to replace ageing infrastructure, thus putting adding additional financial pressure on communities (Maurer et al. 2012, Hering et al. 2013, GDI 2013, American Water Works Association 2012). This follows a global trend, as major investments are needed globally to sustain existing infrastructures. International examples show that Switzerland also needs to take the issue of investment backlogs seriously, as the postponement of infrastructural investment can lead to dramatic investment gaps such as are seen in the United States, where the anticipated capital funding gap is estimated to reach $84 billion dollars by 2020 on current trends (ASCE 2011). Underinvestment in infrastructure is therefore increasingly seen as a global risk (OECD 2006/7, Urban Land Institute and Ernst & Young 2007, WEF 2010). The advantages of decentralisation play out particularly in the case of shrinking budgets, highly uncertain future demand and the difficulties of long-term financing (e.g. because of urgent up-front investments) of centralised infrastructures. If communities enter a phase of infrastructural renewal and have depreciated infrastructures, such as in Switzerland, these periods constitute windows of opportunity to re-think the degree of centralisation and make any necessary adaptations (cf. Koziol 2006, Maurer and Herlyn 2006, GDI 2013).

[8] The estimated CHF 100 billion replacement costs according to Maurer and Herlyn (2006) are converted to Purchasing Power Parities (PPP) US$ where 1 US$ in 2006 corresponds to CHF 1.65 (World Bank 2015). This total cost is then normalised at the total Swiss population in 2006 (BFS 2016).

1.6 Thesis overview

In the remaining chapters of this thesis, three individual publications are presented which address different research questions relating to the overarching question of the ODC. The scientific publications are presented in chronological order of journal submission, reflecting the development of the argumentation in this thesis. As this doctoral thesis is a cumulative dissertation and all main chapters can be read individually, some lines of thought and arguments may be repeated in the different chapters. All publications have been published or submitted to the academic journal Water Research with the aim of contributing to the ongoing debate about centralised and decentralised infrastructures especially within the UWM (engineering) community.

Chapter 2 (Eggimann et al. 2015) presents a heuristic tool for the ODC from an economic optimisation point of view. The introduced ODC is a measure which considers fully hybrid approaches to sanitation infrastructure, i.e. it allows an optimisation over the whole spectrum of possible WMS dimensions. Within this chapter, the foundations are laid for a full cost assessment of hybrid wastewater infrastructure as presented in Chapter 4. A Sustainable Network Infrastructure Planning (SNIP) approach is introduced and illustrated on the basis of a Swiss case study. Within this chapter, the following research questions (see Section 1.2) are particularly addressed:

- *How can the degree of centralisation be defined?*
- *How does space influences the costs of centralised WMS and how can these spatial cost influences be modelled?*
- *How can heuristic geospatial modelling tools help to determine the optimal number, location and sizing of wastewater management systems in a region from an economic point of view?*
- *How can spatial cost influences be modelled?*

Chapter 3 (Eggimann et al. 2016a) introduces a generic methodology aiming to improve our understanding of the costs of decentralised on-site WMS, which have so far been less well understood. Because the facility location problem is closely related to transportation (of sludge and scum in the case of decentralised treatment) it is closely linked to optimal vehicle routing (Nagy and Salhi 2007). Space-dependent costs, i.e. economies of densities, are therefore modelled for on-site WMS with the aid of heuristic routing algorithms. Within this chapter, the following research question is addressed:

- *How does space influence the cost of decentralised systems and how can these spatial cost influences be modelled?*

Chapter 4 (Eggimann et al. 2016b) combines the findings from Chapters 2 and 3 to provide a coherent framework for a total cost assessment of hybrid sanitation infrastructures. The focus of this chapter is to combine the full-cost framework with organisational and institutional aspects. Within this chapter, the following research questions are addressed:

- *How do economies of scale, density effects and costs of transportation interact at settlement level?*
- *What role do institutional and organisational settings play in achieving optimal connection rates?*

Chapter 5 presents an outlook onto potentially fruitful future research areas building on the research presented in this thesis, including policy and practice recommendations.

Chapter 6 concludes with general lessons learnt and the main contributions of this doctoral thesis.

1.7 Declaration of personal contribution

This doctoral thesis is a cumulative thesis and therefore based on publications which were written together with different co-authors. Sven Eggimann is the first author of all included publications and the main contributor with the greatest intellectual and analytical contribution. The personal contribution of Sven Eggimann towards this dissertation principally includes the following:

- the development of all geospatial models (including programming, testing, working with the cluster, data collection etc.)
- the drafting and framing of publications
- the revision of all manuscripts, organising collaboration with the co-authors
- the identification and pursuit of the research focus
- the close supervision of the work of several interns relating to this work
- reviewing related literature and incorporating the reviews in this thesis
- large parts of the creative thinking.

However, Prof. Dr. Max Maurer and Prof. Dr. Bernhard Truffer took on the role of principal investigators, supervising and contributing accordingly in terms of ideas and intellectual guidance, steering the overall writing and formation process of this thesis and the included publications. All publications have been written together, including framing the storyline, polishing the manuscripts with multiple revision rounds (also additionally based on feedback of anonymous reviewers). Max Maurer was particularly involved in crafting Chapter 2 and Bernhard Truffer particularly in Chapter 4. The framing Chapters 1, 5 and 6 of this thesis were written by Sven Eggimann with feedback from both supervisors. Everything was proof-read by Margarete and Richard Michell. ■

1.8 References

Abd El Gawad, H., Butter, J., **1995**. Clustering of towns and villages for centralized wastewater treatment. *Water Science and Technology*, 32(11), 85–95.

Adams, B.J., Dajani, J.S., Gemmell, R.S., **1972**. On the centralization of wastewater treatment facilities. *Water Resources Bulletin*, 8 (4), 669–678

Alnouri, S.Y., Linke, P., El-Halwagi, M., **2015**. A synthesis approach for industrial city water reuse networks considering central and distributed treatment systems. *Journal of Cleaner Production*, 89, 231–250.

American Water Works Association, **2012**. Buried No Longer: Confronting America's Water Infrastructure Challenge. American Water Works Association, Denver, CO, USA.

Arora, M., Malano, H., Davidson, B., Nelson, R., George, B., **2015**. Interactions between centralized and decentralized water systems in urban context: A review. *WIREs Water* 2015(2), 623–634.

Arthur, W.B., **1989**. Competing technologies, increasing returns, and lock-in by historical events. *The Economic Journal*, 99(394) 116–131.

ASCE, **2011**. Failure to Act: The economic impact of current investment trends in water and wastewater treatment infrastructure. American Society of Civil Engineers, Virginia, USA.

Bach, P.M., Rauch, W., Mikkelsen, P.S., McCarthy, D.T., Deletic, A., **2014**. A critical review of integrated urban water modelling – Urban drainage and beyond. *Environmental Modelling & Software*, 54, 88–107.

BAFU, **2016**. Indicator Connection rate and performance level of wastewater treatment plants. http://www.bafu.admin.ch/umwelt/indikatoren/08605/08610/index.html?lang=de (Accessed: 09.08.2016).

Bahri, A., **2012**. Integrated Urban Water Management. TEC Background Papers No. 16, Global Water Partnership, Stockholm, Sweden.

Bakalian, A., Wrigth, A., Otis, R., Netto, J.D.A., **1993**. Simplified Sewerage: Design Guidelines. UNDP-World Bank Water & Sanitation Program. Washington, USA.

Baron, S., Kaufmann Alves, I., Schmitt, T.G., Schöffel, S., Schwank, J., **2015**. Cross-sectoral optimization and visualization of transformation processes in urban water infrastructures in rural areas. *Water Science and Technology*, 72(10), 1730–1738.

Baufumé, S., Grüger, F., Grube, T., Krieg, D., Linssen, J., Weber, M., Hake, J-F., Stolten, D., **2013**. GIS-based scenario calculations for a nationwide German hydrogen pipeline infrastructure. *International Journal of Hydrogen Energy*, 38(10), 3813–3829.

BFS, **2016**. Population size and population composition - Data indicators. Available at: http://www.bfs.admin.ch/bfs/portal/en/index/themen/01/02/blank/key/bevoelkerungsstand.html (Accessed: 17.03.2016)

Bhatia, A., **2016**. Centralized vs decentralized air conditioning systems. Available at: http://www.webcitation.org/6hCRkaTzT

Bieupoude, P., Azoumah, Y., Neveu, P., **2012**. Optimization of drinking water distribution networks: Computer-based methods and constructal design. *Computers, Environment and Urban Systems*, 36(5), 434–444.

Binz, C., Lu, Y., Truffer, B., Shi, Y., **2013**. Institutional trajectory for diffusing on-site wastewater treatment in urban China. *Water Science and Technology*, 68(5), 1180–1187.

Blumensaat, F., Wolfram, M., Krebs, P., **2012**. Sewer model development under minimum data requirements. *Environmental Earth Sciences*, 65(5), 1427–1437.

Bolton, R., Foxon, T.J., **2015**. Infrastructure transformation as a socio-technical process - Implications for the governance of energy distribution networks in the UK. *Technological Forecasting and Social Change*, 90(PB), 538–550.

Brail, R.K., (eds) **2008**. Planning Support Systems. Lincoln Institute of Land Policy. Cambridge, MA, USA.

Brail, R.K., Klosterman, R.E., (eds), **2001**. Planning Support Systems. Integrating geographic information systems, models, and visualization tools. ESRI Press, Redlands, CA, USA.

Brand, N., Ostfeld, A., **2011**. Optimal Design of Regional Wastewater Pipelines and Treatment Plant Systems. *Water Environment Research*, 83(1), 53–64.

Brands, E., **2014**. Prospects and challenges for sustainable sanitation in developed nations: a critical review. *Environmental Reviews*, 22(4), 346–363.

Brears, R., **2014**. Transitioning towards urban water security in asia-pacific. Working Paper, 12. NFG Research Group 'Asia Perceptions of the EU'. Freie Universität Berlin, Berlin, Germany.

Brown, R.R., Keath, N., Wong, T.H.F., **2009**. Urban water management in cities: historical, current and future regimes. Water Science and Technology : *A Journal of the International Association on Water Pollution Research*, 59(5), 847–855.

Calhoun, C., (eds) **2002**. *Dictionary of the Social Sciences*. Oxford University Press, New York, USA.

Christoffersen, H., Paldam, M., Würtz, A. H., **2007**. Public versus Private Production and Economies of Scale. *Public Choice*, 130(3/4), 311–328.

Coenen, L., Truffer, B., **2012**. Places and Spaces of Sustainability Transitions: Geographical Contributions to an Emerging Research and Policy Field. *European Planning Studies*, 20(3), 367–374.

Converse, A.O., **1972**. Optimum number and location of treatment plants. Water Pollution Control Federation, 44(8), 1629–1636.

Corcoran, E., Nellemann, C., Baker, E., Bos, R., Osborn, D., Savelli, H. (eds), **2010**. Sick Water? The central role of wastewater management in sustainable development. A Rapid Response Assessment. United Nations Environment Programme, UN-HABITAT, GRID-Arendal.

Cornejo, P.K., Zhang, Q., Mihelcic, J.R., **2016**. How does scale of implementation impact the environmental sustainability of wastewater treatment integrated with resource recovery? *Environmental Science & Technology*, 50(13), 6680–6689.

Cortes, U., Sanchez-Marre, M., Ceccaroni, L., R-Roda, I., Poch, M. **2000**. Artificial Intelligence and Environmental Decision Support Systems. *Applied Intelligence*, 13(13), 77–91.

Crites, R., Tchobanoglous, G., **1998**. Small and Decentralized Wastewater Management Systems. WCB McGraw-Hill, Boston, USA.

Current, J., Daskin, M., Schilling, D., **2002**. Discrete network location models. Chapter 3. In: Drezner, Z., Hamacher, H. (eds), *Facility Location Theory: Applications and Methods*, Springer-Verlag, Berlin, 81–118.

Cushman, D., **2010**. Reed's law and how multiple identities make the long tail just that little bit longer. In: Breslin, J.G., Burg, T.N., Kim, H.G., Raftery, T., Schmidt, J.H. (eds), *Recent trends and developments in social software*. Springer, Heidelberg, Germany.

Dahlgren, E., Göçmen, C., Lackner, K., Van Ryzin, G., **2013**. Small modular infrastructure. *The Engineering Economist*, 58(4), 231–264.

Daigger, G.T., **2007**. Wastewater Management in the 21st Century. *Journal of Environmental Engineering*, 133(7), 671–680.

Daigger, G.T., **2009**. Evolving urban water and residuals management paradigms: water reclamation and reuse, decentralization, and resource recovery. *Water Environment Research: A Research Publication of the Water Environment Federation*, 81(8), 809–823.

Daskin, M.S., **2013**. Network and Discrete Location: Models, Algorithms, and Applications. Wiley, New York, USA.

Deepa, K., Krishnaveni, M., **2012**. Suitable site selection of decentralised treatment plants using multicriteria approach in GIS. *Journal of Geographic Information System*, 4, 254–260.

Deichmann, U., Meisner, C., Murray, S., Wheeler, D., **2011**. The economics of renewable energy expansion in rural Sub-Saharan Africa. *Energy Policy*, 39(1), 215–227.

Deininger, R.A., Su, S.Y., **1973**. Modelling regional waste water treatment systems. *Water Research*, 7, 633–646.

De Melo, J.J., Câmara, A.S., **1994**. Models for the optimization of regional wastewater treatment systems. European *Journal of Operational Research*, 73, 1–16.

Dendup, N., Tshering, K., **2015**. Demand for Piped Drinking Water and a formal Sewer System in Bhutan. SANDEE Working Papers, No. 97–15.

De Neufville, R., Scholtes, S., **2011**. *Flexibility in Engineering Design*. The MIT Press, Cambridge, USA.

Dominguez, D., Gujer, W., **2006**. Evolution of a wastewater treatment plant challenges traditional design concepts. *Water Research*, 40, 1389–1396.

Dominguez, D., **2008**. Handling future uncertainty - strategic planning for the infrastructure sector. Dissertation, ETH Zürich, Switzerland.

Downing, P.B., **1969**. The Economics of Urban Sewage Disposal. Praeger, New York, USA.

Doyle, D., **2011**. A dictionary of marketing. 3ed, Oxford University Press.

Fane, A.G., Fane, S.A., **2005**. The role of membrane technology in sustainable decentralized wastewater systems. *Water Science and Technology*, 51(10), 317–325.

Eggimann, S., **2013**. Potential for a sustainability transition of decentralised wastewater systems for Switzerland on the level of urban structural units (Potenzial für eine Nachhaltigkeits-Transition dezentraler Abwassersysteme auf Strukturtypebene in der Schweiz). Master Thesis, University of Zürich, Switzerland (in German).

Eggimann, S., Truffer, B., Maurer, M., **2015**. To connect or not to connect? Modelling the optimal degree of centralisation for wastewater infrastructures. *Water Research*, 84, 218–231.

Eggimann, S., Truffer, B., Maurer, M., **2016**a. Economies of density for on-site waste water treatment. *Water Research*, 101, 476–489.

Eggimann, S., Truffer, B., Maurer, M., **2016**b. The cost of hybrid waste water systems: A systematic framework for specifying minimum cost-connection rates. *Water Research*, 103, 472–484.

Eggimann, S., Mutzner, L., Wani, O., Schneider, M., Moy de Vitry, M., Beutler, P., Maurer, M., **2016**c. The potential of knowing more – a review of data-driven urban water management. *Submitted to Environmental Science & Technology*.

Fane, A.G., Fane, S.A., **2005**. The role of membrane technology in sustainable decentralized wastewater systems. Water Science and Technology, 51(10), 317–325.

Federal Act on the Protection of Waters, **1991**. 814.20. https://www.admin.ch/opc/en/classified-compilation/19910022/index.html (Accessed 10.08.2016).

Fewtrell, L., Kaufmann, R.B., Kay, D., Enanoria, W., Haller, L., Colford, J.M., **2005**. Water, sanitation, and hygiene interventions to reduce diarrhoea in less developed countries: a systematic review and meta-analysis. *Lancet Infect Dis*, 5(1), 42–52.

Fletcher, T.D., Shuster, W., Hunt, W.F., Ashley, R., Butler, D., Arthur, S., Trowsdale, S., Barraud, S., Semadeni-Davies, A., Bertrand-Krajewski, J.-L., Steen, P., Mikkelsen, S., Rivard, G., Uhl, M., Dagenais, D., Viklander, M., **2015**. SUDS, LID, BMPs, WSUD and more – The evolution and application of terminology surrounding urban drainage. *Urban Water Journal*, 12(7), 525–542.

Flotats, X., Bonmatí, A., Fernández, B., Magrí, A., **2009**. Manure treatment technologies: on farm versus centralized strategies. NE Spain as case study. *Bioresource Technology*, 100, 5519–5526.

Floyd, K., Skeva, J., Nyirenda, T., Gausi, F., Salaniponi, F., **2003**. Cost and cost-effectiveness of increased community and primary care facility involvement in tuberculosis care in Lilongwe District, Malawi. *International Journal of Tuberculosis and Lung Disease*, 7(9), 29–37.

Franzske, U., Heidenreich, R., Ehle, A., Ziller, F., **2003**. Comparison between decentralised and centralised air conditioning systems. ILK Dresden, Germany.

Friedler, E., Hadari, M., **2006**. Economic feasibility of on-site greywater reuse in multi-storey buildings. *Desalination*, 190(1-3), 221–234.

Fuenfschilling, L., Truffer, B., **2014**. The structuration of socio-technical regimes—Conceptual foundations from institutional theory. *Research Policy*, 43(4), 772–791.

Funcke, S., Bauknecht, D., **2016**. Typology of centralised and decentralised visions for electricity infrastructure. *Utilities Policy*, 2–9.

Gawel, E. (eds), **2015**. Die Governance der Wasserinfrastruktur (Band I & II). Rahmenbedingungen, Herausforderungen und Optionen. Duncker & Humblot, Berlin, Deutschland.

GDI, **2013**. Aufbruch im Untergrund. Szenarien für eine erfolgreiche Zusammenarbeit im Tiefbau. Gottlieb Duttweiler Institute, Zürich, Schweiz.

Geels, F., **2005**. Co-evolution of technology and society: The transition in water supply and personal hygiene in the Netherlands (1850–1930)—a case study in multi-level perspective. *Technology in Society*, 27(3), 363–397.

Geels, F.W., **2006**. The hygienic transition from cesspools to sewer systems (1840–1930): The dynamics of regime transformation. *Research Policy*, 35(7), 1069–1082.

Geels, F.W., Schot, J., **2007**. Typology of sociotechnical transition pathways. *Research Policy*, 36(3), 399–417.

Gikas, P., Tchobanoglous, G., **2009**. The role of satellite and decentralized strategies in water resources management. *Journal of Environmental Management*, 90(1), 144–152.

Gleick, P.H., **2003**. Global freshwater resources: soft-path solutions for the 21st century. *Science*, 302, 1524–1528.

Goymann, M., Wittenwiler, M., Hellweg, S., **2008**. Environmental Decision Support for the Construction of a " Green " Mountain Hut Environmental Decision Support for the Construction of a "Green" Mountain Hut. *Environmental Science & Technology*, 42(11), 4060–4067.

Graham, S, Marvin, S, **2001**. Splintering Urbanism. Routledge, London, UK.

Grant, G., **2016**. The Water Sensitive City. John Wiley & Sons, Chichester, UK.

Greene, R., Agbenowosi, N., Loganathan, G.V., **1999**. GIS-Based Approach to Sewer System Design. *Journal of Surveying Engineering*, 125(1), 36–57.

Greengard, S., **2015**. The internet of things. MIT Press, London, England, UK.

Grübler, A., **1990**. The rise and fall of infrastructures: dynamics of evolution and technological change in transport. Physica-Verlag Heidelberg, Germany.

Gulbenkian (eds), **2014**. Water and the Future of Humanity. Revisiting Water Security. Calouste Gulbekian Foundation, Springer, New York, USA.

Guo, T., Englehardt, J., Wu, T., **2014**. Review of cost versus scale: Water and wastewater treatment and reuse processes. *Water Science and Technology,* 69(2), 223–234.

Guo, T., Englehardt, J.D., **2015**. Principles for scaling of distributed direct potable water reuse systems: A modeling study. *Water Research*, 75, 146–163.

Guo, T., Englehardt, J.D., Fallon, H. J., **2016**. Modeling the Economic Feasibility of Large-Scale Net-Zero Water Management: A Case Study. *Water Environment Research*, 88(9), 811–823.

Haan, F.J. De, Rogers, B.C., Brown, R.R., Deletic, A., **2016**. Many roads to Rome : The emergence of pathways from patterns of change through exploratory modelling of sustainability transitions. *Environmental Modelling & Software*, 85, 279–292.

Haile M.G. **2009**. GIS-based estimation of sewer properties from urban surface information. Master Thesis. University of Dresden.

Hall, J.W., Tran, M., Hickford, A.J., Nicholls, R.J., **2016**. The Future of National Infrastructure. A System-of-Systems Approach. Cambridge University Press, Cambridge. Great Britain.

Hamilton, B.A., Pinkham, R.D., Hurley, E., Watkins, K., Lovins, A.B., Magliaro, J., Etnier, C., Nelson, V., **2004**. Valuing decentralized wastewater technologies: a catalog of benefits costs and economic analysis techniques. Rocky Mountain Institute, Snowmass, USA.

Hansman, R.J., Magee, C., Neufville, R. de., Robins, R., Roos, D., **2006**. Research agenda for an integrated approach to infrastructure planning, design and management. *International Journal of Critical Infrastructures*, 2(2/3), 146–159.

Hendrickson, T.P., Nguyen, M., Sukardi, M., Miot, A., Horvath, A., Nelson, K.L., **2015**. Life-cycle Energy Use and Greenhouse Gas Emissions of a Building-scale Wastewater Treatment and Non-potable

Reuse System. *Environmental Science & Technology*, 49(17), 10303–10311.

Herder, P.M., Wijnia, Y., **2012**. A Systems View on Infrastructure Asset Management. In: Van der Lei, T., Herder, P., Wijnia, Y., (eds), *Asset Management*. Chapter 3, Springer, Dordrecht Heidelberg London New York.

Hering, J.G., Waite, T.D., Luthy, R.G., Drewes, J.E., Sedlak, D.L., **2013**. A changing framework for urban water systems. *Environmental Science and Technology*, 47(19), 10721–10726.

Hines, P.D.H., Blumsack, S., Schläpfer, M., **2015**. Centralized versus Decentralized Infrastructure Networks, 1–8. Retrieved from http://arxiv.org/abs/1510.08792 (Accessed: 24.08.2016).

Hovey, W.H., Schroeder, E.D., Tchobanoglous, G., **1977**. Optimal Size of Regional Wastewater Treatment Plants. California Water Resources Center, Contribution No. 161, University of California, CA, USA.

Ignaczak, N.M., **2014**. 21 technologies that will decentralize the world. Shareable. WebCite® http://www.webcitation.org/6hapK2xXN

Jewitt, S., **2011**. Poo gurus? Researching the threats and opportunities presented by human waste. Applied Geography, 31, 761–769.

Johnson, N., Yang, C., Ogden, J., **2008**. A GIS-based assessment of coal-based hydrogen infrastructure deployment in the state of Ohio. *International journal of hydrogen energy*, 37 (20), 5287–5303.

Johnson, N., Ogden, J., **2012**. A spatially-explicit optimization model for long-term hydrogen pipeline planning. *International Journal of Hydrogen Energy*, 37(6), 5421–5433.

Karaca, F., Camci, F., Raven, P.G., **2013**a. City blood: A visionary infrastructure solution for household energy provision through water distribution networks. *Energy*, 61, 98–107.

Karaca, F., Raven, P. G., Machell, J., Varga, L., Camci, F., Chitchyan, R., Boxall, J., Ulanicki, B., Skworcow, P., Strzelecka, A., Ozawa-Meida, L., Janus, T. **2013**b. Single infrastructure utility provision to households: Technological feasibility study. *Futures*, 49, 35–48.

Karaca, F., Raven, P.G., Machell, J., Camci, F., **2015**. A comparative analysis framework for assessing the sustainability of a combined water and energy infrastructure. *Technological Forecasting and Social Change*, 90(PB), 456–468.

Kaundinya, D.P., Balachandra, P., Ravindranath, N.H., Ashok, V., **2013**. A GIS (geographical information system)-based spatial data mining approach for optimal location and capacity planning of distributed biomass power generation facilities: A case study of Tumkur district, India. *Energy*, 52, 77–88.

Kennedy-Walker, R., Holderness, T., Barr, S., Alderson, D., Evans, B., **2014**. Network modelling for road-based faecal sludge management. *Proceedings of the Institution of Civil Engineers*, 167(3), 157–165.

Kiparsky, M., Sedlak, D.L., Thompson, B.H., Truffer, B., **2013**. The innovation deficit in urban water: The need for an integrated perspective on institutions, organisations, and technology. *Environmental Engineering Science*, 30(8), 395–408.

Kluge, T., Scheele, U., **2008**. Von dezentralen zu zentralen Systemen und wieder zurück? Räumliche Dimensionen des Transformationsprozesses in der Wasserwirtschaft. In: Moss, T., Neumann, M., Wissen, M., (eds), 2008. *Infrastrukturnetze und Raumentwicklung*. Oekom Verlag, München, Germany.

Knops, H., **2008**. The impact of technical characteristics of network industries upon the governance of infrastructure adequacy. In: Arts, G., Dicke, W., Hancher, L. (eds), *New perspectives on investment in infrastructures*. Amsterdam University Press, Amsterdam, The Netherlands.

Kocaman, A.S., Huh, W.T., Modi, V., **2012**. Initial layout of power distribution systems for rural electrification: a heuristic algorithm for multilevel network design. *Applied Energy*, 96, 302–315.

Koziol, M., **2006**. Transformationsmanagement unter den besonderen Bedingungen der Schrumpfung. In: Kluge, T., Libbe, J., (eds), *Transformation netzgebundener Infrastruktur. Strategien für Kommunen am Beispiel Wasser.* Mercedes-Druck, Berlin, Germany.

Larsen, T.A., Alder, A.C., Eggen, R.I.L., Maurer, M., Lienert, J., **2009**. Source Separation: Will We See a Paradigm Shift in Wastewater Handling? *Environmental Science & Technology*, 43(16), 6121–6125.

Larsen, T.A., Udert, K.M., Lienert, J., (eds) **2013**. Source separation and decentralization for wastewater management. IWA Publishing, London, UK.

Larsen, T.A., Hoffmann, S., Lüthi, C., Truffer, B., Maurer, M., **2016**. Emerging solutions to the water challenges of an urbanizing world. *Science*, 352(6288), 928–933.

Lee, E.J., Criddle, C.S., Bobel, P., Freyberg, D. L., **2013**. Assessing the scale of resource recovery for centralized and satellite wastewater treatment. *Environmental Science and Technology*, 47, 10762–10770.

Leitão, J., Matos, J.S., Gonçalves, A.B., Matos, J.L., **2005**. Contribution of Geographic Information Systems and location models to planning of wastewater systems, *Water Science and Technology*, 52(3), 1–8.

Levin, T., Thomas, V.M., **2012**. Least-cost network evaluation of centralized and decentralized contributions to global electrification. *Energy Policy*, 41, 286–302.

Li, L., Binz, C., Lu, Y., Truffer, B., Shi, Y., **2013**. Institutional trajectory for diffusing on-site wastewater treatment in urban China. *Water Science and Technology*, 68(5), 1180–1187.

Libralato, G., Ghirardini, A.V., Avezzù, F., **2012**. To centralise or to decentralise: An overview of the most recent trends in wastewater treatment management. *Journal of Environmental Management*, 94, 61–68.

Lieberherr, E., Truffer, B., **2015**. The impact of privatization on sustainability transitions: a comparative analysis of dynamic capabilities in three water utilities. *Environmental Innovation and Societal Transitions*, 15, 101–122.

Lieberherr, E., Fuenfschilling, L., **2016**. Neoliberalism and sustainable urban water sectors: a critical reflection of sector characteristics and empirical evidence. *Environment and Planning C: Government and Policy*, 1–16.

Lienhoop, N., Al-Karablieh, E.K., Salman, A.Z., Cardona, J.A., **2014**. Environmental cost-benefit analysis of decentralised wastewater treatment and re-use: a case study of rural Jordan. *Water Policy*, 16(2), 323–339.

Lofrano, G., Brown, J., **2010**. Wastewater management through the ages: a history of mankind. *Science of the Total Environment*, 408, 5254–5264.

Makropoulos, C.K., Argyrou, E., Memon, F.A., Butler, D., **2007**. A suitability evaluation tool for siting wastewater treatment facilities in new urban developments. *Urban Water Journal*, 4(2), 61–78.

Marcoulaki, E.C., Papazoglou, I.A., Pixopoulou, N., **2012**. Integrated framework for the design of pipeline systems using stochastic optimisation and GIS tools. *Chemical Engineering Research and Design*, 90(12), 2209–2222.

Markard, J., Truffer, B., **2008**. Technological innovation systems and the multi-level perspective: Towards an integrated framework. *Research Policy*, 37(4), 596–615.

Markard, J., **2011**. Infrastructure sector characteristics and implications for innovation and sectoral change. *Journal of Infrastructure Systems*, 17(3), 107–117.

Markard, J., Raven, R., Truffer, B., **2012**. Sustainability transitions: An emerging field of research and its prospects. *Research Policy*, 41(6), 955–967.

Marlow, D.R., Moglia, M., Cook, S., Beale, D.J., **2013**. Towards sustainable urban water management: A critical reassessment. *Water Research*, 47(20), 7150–7161.

Marques, R.C., De Witte, K., **2011**. Is big better? On scale and scope economies in the Portuguese water sector. *Economic Modelling*, 28(3), 1009–1016.

Marufuzzaman, M., Eksioğlu, S. D., Hernandez, R., **2015**. Truck versus pipeline transportation cost analysis of wastewater sludge. *Transportation Research Part A: Policy and Practice*, 74,14–30.

Massoud, M.A, Tarhini, A., Nasr, J.A., **2009**. Decentralized approaches to wastewater treatment and management: applicability in developing countries. *Journal of Environmental Management*, 90(1), 652–659.

Maurer, M., Rothenberger, D., Larsen, T.A., **2005**. Decentralised wastewater treatment technologies from a national perspective: at what cost are they competitive? Water Science and Technology, 5(6), 145–154.

Maurer, M., Herlyn, A., **2006**. Status, Costs and Investment Needs of Wastewater Removal in Switzerland (Zustand, Kosten und Investitionsbedarf der Schweizerischen Abwasserentsorgung). Eawag, Zürich, Switzerland (in German).

Maurer, M., Rothenberger, D., Larsen, T.A., **2006**. Decentralised wastewater treatment technologies from a national perspective: at what cost are they competitive? *Water Science and Technology Water Supply,* 15(6), 145–154.

Maurer, M., Wolfram, M., Herlyn, A., **2010**. Factors affecting economies of scale in combined sewer systems. *Water Science and Technology*, 62(1), 36–41.

Maurer, M., Chawla, F., Horn, J. v., Staufer, P., **2012**. Abwasserentsorgung 2025 in der Schweiz. Schriftreihe der Eawag Nr. 21, Eawag, Dübendorf, Schweiz.

Maurer, M., Scheidegger, A., Herlyn, A., **2013**. Quantifying costs and lengths of urban drainage systems with a simple static sewer infrastructure model. *Urban Water Journal*, 10(4), 268–280.

Mintz, E.D., Bartram, J., Lochery, P., Wegelin, M., **2001**. Not just a drop in the bucket: Expanding access to point-of-use water treatment systems. *American Journal of Public Health*, 91(10), 1565–1570.

Möller, B., Lund, H., **2010**. Conversion of individual natural gas to district heating: Geographical studies of supply costs and consequences for the Danish energy system. *Applied Energy*, 87(6), 1846–1857.

Morera, S., Comas, J., Poch, M., Corominas, L., **2015**. Connection of neighboring wastewater treatment plants:

economic and environmental assessment. *Journal of Cleaner Production*, 90, 34–42.

Moss, T., **2001**. Flow management in urban regions: introducing a concept. In: Guy, S., Marvin, S., Moss, T., 2001. Urban infrastructure in transition: networks, buildings and plans. Earthscan, Abingdon, UK.

Müller, A., Kramer, D., **2000**. Market Economic Instruments of Sewage Disposal, Ecoplan, Bern, Schweiz.

Nagy, G., Salhi, S., **2007**. Location-routing: Issues, models and methods. *European Journal of Operational Research*, 177(2), 649–672.

Naisbitt, J., **1982**. Megatrends. Ten New Directions Transforming Our Lives. Warner Books.

Nakou, D., Benardos, A., Kaliampakos, D., **2014**. Assessing the financial and environmental performance of underground automated vacuum waste collection systems. *Tunnelling and Underground Space Technology*, 41(1), 263–271.

Newman, J., Dandy, G.C., Maier, H.R., **2014**. Multiobjective optimization of cluster-scale urban water systems investigating alternative water sources and level of decentralization. *Water Resources Research*, 50, 7915–7938

Nielsen, S., Möller, B., **2013**. GIS based analysis of future district heating potential in Denmark. *Energy*, 57, 458–468.

OECD, **2006**. Infrastructure to 2030: Telecom, Land Transport, Water and Electricity. OECD Publishing, Paris, France.

OECD, **2007**. Infrastructure to 2030 (Volume 2): Mapping Policy for Electricity, Water and Transport. OECD Publishing, Paris, France.

OECD, **2010**. OECD Environmental Performance Reviews Japan. OECD Publishing, Paris, France.

OECD, **2015**. Water and cities. Ensuring sustainable futures. OECD Studies on Water. OECD Publishing, Paris, France.

O'Flaherty, B., **2005**. City economics. Harvard University Press, USA.

Orth, H., **2007**. Centralised versus decentralised wastewater systems? *Water Science and Technology*, 56(5), 259–266.

Oxford Dictionaries, **2015**. Decentralization. Link: http://www.oxforddictionaries.com/definition/english/decentralization?q=-decentralisation (Accessed: 01.08.2016).

Pahl-Wostl, C., Jeffrey, P., Isendahl, N., Brugnach, M., **2011**. Maturing the New Water Management Paradigm: Progressing from Aspiration to Practice. *Water Resources Management*, 25(3), 837–856.

Parshall, L., Pillai, D., Mohan, S., Sanoh, A., Modi, V., **2009**. National electricity planning in settings with low pre-existing grid coverage: development of a spatial model and case study of Kenya. *Energy Policy*, 37(6), 2395–2410.

Pelzer, P., **2015**. Usefulness of planning support systems. Conceptual perspectives and practitioners' experiences. Doctoral Thesis. InPlanning, Groningen, The Netherlands.

Perpiñá, C., Alfonso, D., Pérez-Navarro, A., Peñalvo, E., Vargas, C., Cárdenas, R. **2009**. Methodology based on Geographic Information Systems for biomass logistics and transport optimisation. *Renewable Energy*, 34(3), 555–565.

Peter-Varbanets, M., Zurbrügg, C., Swartz, C., Pronk, W., **2009**. Decentralized systems for potable water and the potential of membrane technology. *Water Research*, 43(2), 245–65.

Poustie, M.S., Deletic, A., Brown, R.R., Wong, T., de Haan, F.J., Skinner, R., **2014**. Sustainable urban water futures in developing countries: the centralised, decentralised or hybrid dilemma. *Urban Water Journal*, 1–16.

Reymond, P., Renggli, S., Lüthi, C., **2016**. Towards Sustainable Sanitation in an Urbanising. In: Ergen, M, (eds), 2016. *Sustainable Urbanization,* Chapter 5, INTECH, 115–134.

Rogers, E., **1983**. Diffusion of Innovations, 3rd Edition. The Free Press, New York, USA.

Sadoff, C.W., Hall, J.W., Grey, D., Aerts, J.C.J.H., Ait-Kadi, M., Brown, C., Cox, A., Dadson, S., Garrick, D., Kelman, J., McCornick, P., Ringler, C., Rosegrant, M., Whittington, D., Wiberg, D., **2015**. Securing Water, Sustaining Growth. Report of the GWP/OECD Task Force on Water Security and Sustainable Growth, University of Oxford, Oxford, UK.

Sanoh, A., Parshall, L., Sarr, O.F., Kum, S., Modi, V., **2012**. Local and national electricity planning in Senegal: scenarios and policies. *Energy for Sustainable Development*, 16(1), 13–25.

Sanoh, A., Kocaman, A. S., Kocal, S., Sherpa, S., Modi, V., **2014**. The economics of clean energy resource development and grid interconnection in Africa. *Renewable Energy*, 62, 598–609.

Sapkota, M., Arora, M., Malano, H., Moglia, M., Sharma, A., George, B., Pamminger, F., **2015**. An integrated framework for assessment of hybrid water supply systems. *Water*, 8(1), 4, 1–19.

Schalcher, H.-R., Boesch, H.-J., Bertschy, K., Sommer, H., Matter, D., Gerum, J., Jakob, M., **2011**. Was kostet das Bauwerk Schweiz in Zukunft und wer bezahlt dafür? Nationales Forschungsprogramm 54 – Nachhaltige Siedlungs- und Infrastrukturentwicklung. vdf Hochschulverlag, Zürich.

Schiller, G., Siedentop, S., **2005**. Infrastrukturfolgekosten der Siedlungsentwicklung unter Schrumpfungsbedingungen. *disP - The Planning Review*, 41(160), 83–93.

Schiller, G., **2010**. Cost Evaluation of the Adaptation of Waste Water Treatment Systems Under Shrinkage (Kostenbewertung der Anpassung Zentraler Abwasserentsorgungssysteme bei Bevölkerungsrückgang). IÖR Schriften, Band51, Rhombos, Berlin, Germany.

Sedlak, D., **2014**. Water 4.0: The past, present, and future of the world's most vital resource. Yale University Press, New Haven & London.

Seidenstat, P., **2000**. Emerging competition in water and wastewater industries. Water Resources Update, 117, 6–12.

Shahabi, M.P., McHugh, A., Anda, M., Ho, G., **2015**. Comparative economic and environmental assessments of centralised and decentralised seawater desalination options. *Desalination*, 376, 25–34.

Sharma, A., Pezzaniti, D., Myers, B., Cook, S., Tjandraatmadja, G., Chacko, P., Chavoshi, S., Kemp, D., Leonard, R., Koth, B., Walton, A., **2016**. Water Sensitive Urban Design: An Investigation of Current Systems, Implementation Drivers, Community Perceptions and Potential to Supplement Urban Water Services. *Water*, 8(7), 272.

Siedentop, S., Fina, S., **2010**. Urban sprawl beyond growth: the effect of demographic change on infrastructure costs. Flux, n°79/80, 90–100.

Singh, N.K., Kazmi, A.A., Starkl, M., **2015**. A review on full-scale decentralized wastewater treatment systems: techno-economical approach. *Water Science and Technology*, 71, 468–478.

Sitte, R., **2009**. About the Predictability and Complexity of Complex Systems. In: Aziz-Alaoui, M.A., Bertelle, C., (eds), *From System Complexity to Emergent Properties*, Springer, Berlin Heidelberg, Germany.

Sitzenfrei, R., **2010**. Stochastic generation of urban water systems for case study analysis. Doctoral Dissertation, Leopold Franzens Universität Innsbruck, Austria.

Sitzenfrei, R., Möderl, M., Rauch, W., **2013**b. Automatic generation of water distribution systems based on GIS data. *Environmental Modelling & Software*, 47, 138–147.

Sitzenfrei, R. **2016**. A Review on Network Generator Algorithms for Water Supply Modelling and Application Studies. World Environmental and Water Resources Congress 2016, 505–515.

Smith, A., Voß, J.-P., Grin, J., **2010**. Innovation studies and sustainability transitions: The allure of the multi-level perspective and its challenges. *Research Policy*, 39(4), 435–448.

SNF (Swiss National Science Foundation), **2016**. NRP 73 Sustainable Economy National Research Programme. Call document. Bern, Switzerland. Link: http://www.snf.ch/SiteCollectionDocuments/nfp/nfp73_ausschreibung_en.pdf (Accessed: 29.08.2016)

Starkl, M., Parkinson, J., Narayana, D., Flamand, P., **2012**. Small is beautiful but is large more economical? Fresh views on decentralised versus centralised wastewater management. *Water 21(June)*, 45–47.

Stennikov, V.A., Iakimetc, E.E., **2016**. Optimal planning of heat supply systems in urban areas. *Energy*, 110, 157–165.

Stiller, C., Bünger, U., Møller-Holst, S., Svensson, A.M., Espegren, K.A., Nowak, M., **2010**. Pathways to a hydrogen fuel infrastructure in Norway. *International Journal of Hydrogen Energy*, 35(7), 2597–2601.

Tchobanoglous, G., Ruppe, L., Leverenz, H., Darby, J., **2004**. Decentralized wastewater management: challenges and opportunities for the twenty-first century. *Water Science and Technology: Water Supply,* 4(1), 95–102.

Tchobanoglous, G., Leverenz, H., **2013**. The rationale for decentralization of wastewater infrastructure. In: Larsen, T.A., Udert, K.M., Lienert, J. (eds), *Source Separation and Decentralization for Wastewater Management,* Larsen, IWA Publishing, London, UK.

Thomas, D.A., Ford, R.R., **2005**, The Crisis of Innovation in Water and Wastewater. Edward Elgar Publishing, Cheltenham, UK.

Tilley, E., **2016**. Cost-effectiveness and community impacts of two urine-collection programs in rural South Africa. Environmental Science, *Water Research and Technology*, 2, 320–335.

Tjandraatmadja, G., Burn, S., McLaughlin, M., Biswas, T., **2005**. Rethinking urban water systems - Revisiting concepts in urban wastewater collection and treatment to ensure infrastructure sustainability. Water Science and Technology: Water Supply, 5(2), 145–154.

Townend, C.B., **1959**. The economics of the disposal of sewage and trade effluents. *Bull. World Health Organ* 20, 535–562.

Truffer, B., Störmer, E., Maurer, M., Ruef, A., **2010**. Local strategic planning processes and sustainability transitions in infrastructure sectors. *Environmental Policy and Governance*, 20(4), 258–269.

Truffer, B., Coenen, L., **2012**. Environmental innovation and sustainability transitions in regional studies. *Regional Studies*, 46, 1–21.

Truffer, B., Binz, C., Gebauer, H., Störmer, E., **2013**. Market success of on-site treatment: a systematic innovation problem. In: Larsen, T.A., Udert, K.M., Linert, J. (eds), *Source separation and decentralization for wastewater management*. IWA Publishing, London, UK.

UN, **2015**a. Transforming our world: The 2030 agenda for sustainable development. UN. Link: https://sustainabledevelopment.un.org/post2015/transformingourworld (Accessed: 09.06.2016)

UN, **2015**b. Environmental Indicators. Inland Water Resources, Wastewater. Archived by WebCite® http://www.webcitation.org/6gQ6z23bk

Urban Land Institute and Ernst & Young (eds), **2007**. Infrastructure 2007: a Global Perspective. ULI e the Urban Land Institute, Washington, D.C., USA.

Urich, C., Sitzenfrei, R., Möderl, M., Rauch, W., **2010**. An agent-based approach for generating virtual sewer systems. *Water Science and Technology*, 62(5), 1090–1097.

Urich, C., Rauch, W., **2014**a. Modelling the urban water cycle as an integrated part of the city: A review. *Water Science and Technology*, 70(11), 1857–1872.

Urich, C., Rauch, W., **2014**b. Exploring critical pathways for urban water management to identify robust strategies under deep uncertainties. *Water Research*, 66, 374–389.

Van Afferden, M., Cardona, J.A., Lee, M.-Y., Subah, A., Müller, R.A., **2015**. A new approach to implementing decentralized wastewater treatment concepts. *Water Science and Technology*, 72(11), 1923–1930.

Van de Ven, A.H., **2007**. Engaged Scholarship. A guide for organizational and social research. Oxford University Press, Oxford, UK.

Van der Brugge, R., Rotmans, J., **2007**. Towards transition management of European water resources. *Water Resources Management*, 21(1), 249–267.

Vannini, P., Taggart, J., **2013**. Voluntary simplicity, involuntary complexities, and the pull of remove: The radical ruralities of off-grid lifestyles. *Environment and Planning A*, 45(2), 295–311.

Verrecht, B., James, C., Germain, E., Birks, R., Barugh, A., Pearce, P., Judd, S., **2012**. Economical Evaluation and Operating Experiences of a Small-Scale MBR for Nonpotable Reuse. *Journal of Environmental Engineering*, 138(5), 594–600.

VSA, **2005**. Abwasser im ländlichen Raum. Leitfaden für Planung, Evaluation, Betrieb und Unterhalt von Abwassersystemen bei Einzelliegenschaften und Kleinsiedlungen.

West, C., Kenway, S., Hassall, M., Yuan, Z., **2016**. Why do residential recycled water schemes fail? A comprehensive review of risk factors and impact on objectives. *Water Research*, 102, 271–281.

WEF, **2010**. Global Risks 2010: A Global Risk Network Report. A Global Risk Network Report. Geneva, Switzerland.

Wenban-Smith, H.B., **2009**. Economies of scale, distribution costs and density effects in urban water supply: a spatial analysis of the role of infrastructure in urban agglomeration. London School of Economics and Political Science, London, UK.

WHO, **2011**. Valuing Water, Valuing Livelihoods. Guidance on Social Cost-benefit Analysis of Drinking-water Interventions, with special reference to Small Community Water Supplies. IWA Publishing, London, UK.

WHO, UN-Water, **2014**. Investing in water and sanitation: increasing access, reduction inequalities. UN-Water global Analysis and Assessment of Sanitation and Drinking-Water, GLAAS, WHO, Geneva, Switzerland.

Wiesmann, U., Choi, I.S., Dombrowski, E.-M., **2006**. Historical Development of Wastewater Collection and Treatment. In: *Fundamentals of Biological Wastewater Treatment*, Wiley, Weinheim, Germany.

Woods, G.J., Kang, D., Quintanar, D.R., Curley, F.E., Davis, S.E., Lansey, K.E., Arnold, R.G., **2013**. Centralized versus decentralized wastewater reclamation in the houghton area of Tucson, Arizona. *Journal of Water Resources Planning and Management*, 313–324.

World Bank, **2015**. PPP Conversion Factor. International Comparison Program database. Available at: http://data.worldbank.org/.

WWAP (United Nations World Water Assessment Programme), **2015**. The United Nations World Water Development Report 2015: Water for a Sustainable World. Paris, UNESCO.

Yang, C., Ogden, J., **2007**. Determining the lowest-cost hydrogen delivery mode. *International Journal of Hydrogen Energy*, 32(2), 268–286.

Yang, X.M., Morita, A., Nakano, I., Kushida, Y., Ogawa, H., **2011**. History and Current Situation of Night Soil Treatment Systems and Decentralized Wastewater Treatment Systems in Japan. *Water Practice & Technology*, 5(4), 1–18.

Zeferino, J.A., Cunha, M.C., Antunes, A.P., **2012**. Robust optimization approach to regional wastewater system planning. *Journal of Environmental Management*, 109, 113–122.

Zeppini, P., Frenken, K., Kupers, R., **2014**. Thresholds models of technological transitions. *Environmental Innovation and Societal Transitions*, 11, 54–70.

Zhao, Y. W., Qin, Y., Chen, B., Zhao, X., Li, Y., Yin, X. A., Chen, G. Q., **2009**. GIS-based optimization for the locations of sewage treatment plants and sewage outfalls - A case study of Nansha District in Guangzhou City, China. *Communications in Nonlinear Science and Numerical Simulation*, 14(4), 1746–1757.

Zurbrügg, C., Drescher, S., Patel, A., Sharatchandra, H.C., **2004**. Decentralised composting of urban waste - An overview of community and private initiatives in Indian cities. *Waste Management*, 24(7), 655–662.

Zvoleff, A., Kocaman, A.S., Huh, W.T., Modi, V., **2009**. The impact of geography on energy infrastructure costs. *Energy Policy*, 37(10), 4066–4078.

Chapter 2

To connect or not to connect? Modelling the optimal degree of centralisation for wastewater infrastructures

*published in **Water Research**, 84, 218–231.*

To connect or not to connect? Modelling the optimal degree of centralisation for wastewater infrastructures

Eggimann Sven[1, 2*], Truffer Bernhard[1,3], Maurer Max[1, 2]

[1] Eawag, Swiss Federal Institute of Aquatic Science and Technology, 8600 Dübendorf, Switzerland.

[2] Institute of Civil, Environmental and Geomatic Engineering, ETH Zürich, 8093 Zurich, Switzerland.

[3] Faculty of Geosciences, Utrecht University, Heidelberglaan 2, NL-3584 CS Utrecht, The Netherlands

Keywords

Sustainable Network Infrastructure Planning, Geographic Information System, Sewer Modelling, Algorithmic Network Generation, Wastewater Infrastructure, Degree of Centralisation

Abstract

The strong reliance of most utility services on centralised network infrastructures is becoming increasingly challenged by new technological advances in decentralised alternatives. However, not enough effort has been made to develop planning tools designed to address the implications of these new opportunities and to determine the optimal degree of centralisation of these infrastructures. We introduce a planning tool for sustainable network infrastructure planning (SNIP), a two-step techno-economic heuristic modelling approach based on shortest path-finding and hierarchical-agglomerative clustering algorithms to determine the optimal degree of centralisation in the field of wastewater management. This SNIP model optimises the distribution of wastewater treatment plants and the sewer network outlay relative to several cost and sewer-design parameters. Moreover, it allows us to construct alternative optimal wastewater system designs taking into account topography, economies of scale as well as the full size range of wastewater treatment plants. We quantify and confirm that the optimal degree of centralisation decreases with increasing terrain complexity and settlement dispersion while showing that the effect of the latter exceeds that of topography. Case study results for a Swiss community indicate that the calculated optimal degree of centralisation is substantially lower than the current level.

2.1 Introduction

2.1.1 Sustainable Network Infrastructure Planning (SNIP)

In the last two centuries, many physical network infrastructures of various types have been built worldwide.[1] This implementation of extensive networks was accompanied by a widely shared conviction in expert and policy circles that technological centralisation would generally lead to superior solutions (Graham and Marvin 2001). As a consequence, an "expand and upgrade" philosophy became predominant (Moss 2001). This approach leads to biased economic incentives because actors tend to base their decisions on economies of scale in the cost of a centralised wastewater plant, while neglecting the economies of scale at the level of the entire network, which are, as a rule, much more difficult to assess (Maurer et al. 2012). As a consequence, centralisation always seems to be the preferred solution for decision makers. More recently, however, new context conditions have led to this generally received wisdom being questioned (Marlow et al. 2013). Reasons for questioning the sustainability of the centralised approach include shrinking public budgets and subsidies as well as the massive maintenance and restoration costs of centralised systems (Maurer and Herlyn 2006). Furthermore, new technological advances such as remotely operating measuring devices and membrane technology challenge the centralised approach as they increasingly help decentralised technology to be considered as a fully functional substitute for centralised infrastructures (Libralato et al. 2012).

We assume that decentralised alternatives can already, or will soon be able to, deliver utility services of comparable quality, which means that the superiority of the centralised paradigm can no longer be taken for granted, and questions about the optimal degree of centralisation (ODC) need to be addressed. A shift to a decentralised approach has broad economic, technical and environmental implications (e.g. environmental risks) which need to be addressed elsewhere in the literature (inter alia Libralato et al. 2012, Poustie et al. 2014). In the present paper, we introduce the Sustainable Network Infrastructure Planning (SNIP) approach, which consists of a single objective cost-optimisation algorithm designed to determine the ODC for wastewater systems. We start from the assumption that we do not have to choose either a purely centralised or a purely decentralised service structure for a given region but that the optimum configuration will generally be defined by some sort of hybrid constellation (Poustie et al. 2014, Sapkota et al. 2015), also referred to as a distributed wastewater infrastructure (inter alia Tchobanoglous and Leverenz

[1] Examples can be found in the field of transportation (Rodrigue et al. 2013), in heating and energy systems (Hughes 1983, Gochenour 2001, Hawkey 2012) as well as drinking and wastewater systems (Lofrano and Brown 2010, Geels 2006).

2013). We define a system as being increasingly centralised as more elements are linked to it and interconnected (for an elaborate definition, see Section 2.3.1). As a result, we are able to determine to what degree economies of scale in wastewater treatment drive infrastructural centralisation, or whether distributed systems may result in lower total system costs.

Finding the ODC is methodologically challenging because of the large number of system alternatives that have to be considered. Very recently, scholars have started to tackle these complexities in integrated strategic planning by means of exploratory modelling techniques (Urich and Rauch 2014). Still, only few tools (for exceptions see inter alia Zeferino et al. 2010, Sitzenfrei et al. 2013, Urich and Rauch 2014) are currently available to determine optimal combinations of these alternatives, especially if we consider real-world data. The main focus of the present paper is to introduce the SNIP methodology and apply it to the case of wastewater management. These systems are highly suitable infrastructures for studying ODC. The sector has developed a strongly centralised paradigm in many industrialised countries, which has frequently led to connection rates above 95%. However, fully functional decentralised alternatives have emerged only recently and their longer-term contribution to wastewater treatment is still unknown. Finally, centralised infrastructures are coming to the end of an investment cycle, and many communities in the industrialised world have to consider whether and how they want to reinvest in their existing systems (OECD 2006/7, Urban Land Institute and Ernst&Young 2007). This question is also relevant for other network infrastructures such as electricity, heating or water supplies.

The current SNIP approach comprises a single-objective framework focussing exclusively on the minimisation of total system costs (compare inter alia Weber et al. 2007, Sapkota et al. 2013). SNIP could very well be expanded in a multi-objective approach, where a broader set of objectives could be included in the cost or objective function. However, many of the key objectives, such as performance, failure frequency or environmental effects of distributed wastewater systems are not trivial to assess and their inclusion in the text would greatly exceed the scope of this paper. Our approach limits itself to determining the ODC only from a cost efficiency point of view.

The manuscript is structured as follows: in the remainder of Section 2.1 we further specify the state of the literature on determining ODCs for network infrastructures. In Section 2.2 we present the SNIP model in detail. Sections 2.3 and 2.4 present real-world and virtual case studies to illustrate the performance of the approach. Section 2.5 concludes this study specifying the further development steps of the methodology.

2.1.2 Location Problem in the Field of Wastewater Management

Finding the ODC for wastewater infrastructures involve questions of optimal geographical placement, sizing and number of facilities and can be seen as a location model. Different types of location models exist, whereas a model designed to minimize total facility and transportation costs is defined as a fixed-charge location problem (Current et al. 2002).[2] For an application in wastewater management, we define the facilities as wastewater treatment plants (WWTP) and understand sewer-related infrastructures as a means of transporting wastewater. It is extremely difficult to solve these kinds of optimum location models because they are NP-complete. The most important aspect of NP-complete problems is that we cannot solve them deterministically in polynomial time (Garey and Johnson 1979). Therefore finding solutions results in a high computational burden for any application that involves realistic data sets. One way to solve these problems is by looking for approximate solutions with the aid of heuristics. Given the complexity of the problem of determining the ODC, finding approximate solutions with the aid of heuristics is already a big step forward. Approximate solutions may still be very useful for decision makers at those points in time when strategic decisions must be made.

Compared to other network infrastructures, the management of wastewater has some specific characteristics:

- There exists a long-known economic trade-off between installing wastewater treatment plants and extending the sewer network (inter alia Converse 1972). The literature suggests high economies of scale in the treatment of wastewater but a tendency for diseconomies of scale in the construction of sewer networks. This trade-off is further aggravated as typically more than 80% of the investment costs have to be spent on sewer infrastructures (Maurer et al. 2006). These cost calculations are based on typical infrastructure lifetimes of 25 years for WWTP and 80 years for sewers.
- Water is quite bulky and heavy per source (household) and wastewater generation rates vary depending on the geographical context (UNEP 2015). As a consequence, topography has a strong influence on network costs, especially as gravity-driven sewers are the preferred type of transportation.
- Sewers are usually considered to have a relatively high average

[2] Fixed costs are assumed for locating a facility at a candidate site. For a detailed problem formulation, see Daskin (1995).

life-span of about 80 years compared to approximately 25 years for large scale WWTP. Larger uncertainties are attributed to the life expectancy of smaller WWTP.

2.1.3 Critical Literature Review

Despite the fact that the problem of finding the ODC has been raised repeatedly (inter alia by Downing 1969, Abd el Gawad and Butter 1995, Ambros 1996) in various technological fields, only little research has actually been conducted into this topic. However, we notice that researchers are increasingly focussing on the transition to more decentralised systems (inter alia Sitzenfrei and Rauch 2014, Bach et al. 2013) and the question of the sustainability of the degree of centralisation (inter alia Poustie et al. 2014).

The issue of the optimal degree of centralisation is crucial for many network based infrastructures. Therefore, before focussing on the literature in the field of wastewater we will take a look at the available literature in other fields, especially that of electricity infrastructures. Although a comparison with other infrastructures such as water distribution systems (inter alia Ostfeld 2015) would be interesting, we believe that the link to the energy literature is especially fruitful given its extensive use of heuristic approaches.

Recently, discussions about centralised versus decentralised technologies have taken place in the fields of electricity network infrastructures (Kocaman et al. 2012, Levin and Thomas 2012, Sanoh et al. 2012, Parshall et al. 2009, Deichmann et al. 2011), hydrogen distribution networks (Johnson et al. 2008, Stiller et al. 2010, Baufumé et al. 2013) and district heating (Möller and Lund 2010, Gils et al. 2013, Nielsen and Möller 2013). Different types of methodological approaches such as mixed integer programming, branch and bound methods or heuristic algorithms are used to determine the optimal outlays for these infrastructures (Kocaman et al. 2012).

Zvoleff et al. (2009) use a heuristic network algorithm to access the impact of geography on infrastructure costs and suggest a linkage between the increasing distance per building connection (marginal distance) and the increasing percentage of the connected population. The marginal distance indicates when connection expenses become unreasonable, thus making a decentralised option economically preferable. Levin and Thomas (2012) use similar techniques and create a least-cost transmission network for connecting a given fraction of the population. Even though the authors include decentralised technologies, they do not consider multiple disaggregated networks. In contrast, Sanoh et al. (2012) and Parshall et al. (2009) start from a pre-existing network and try to determine whether specific still-unconnected nodes are better served with a decentralised option or a network extension.

The most comprehensive approach so far considers multiple transformer stations and network sizes to determine the optimal infrastructure outlay (Kocaman et al. 2012). The authors use an agglomerative hierarchical clustering method to find optimal locations of transformers and minimize overall grid costs. This approach consequently results in networks of various sizes and thus produces hybrid solutions. Its limiting factor is the large computation burden when the restrictions are more complex or the algorithm is not based on straight-line distances alone.

For wastewater management, network infrastructures (simulated or pre-existing) are also needed to estimate centralised and decentralised costs. For a recent overview of integrated urban water modelling techniques we refer to Bach et al. (2014). Even though a number of innovative methods are available to design and automatically generate different kinds of network infrastructure such as drinking water (inter alia Urich et al. 2010) or sewer networks (inter alia Blumensaat et al. 2012, Bach et al. 2014),[3] they are not used to address the question of the ODC. With the few exceptions listed below, no geographically explicit analysis of where to treat wastewater in a more decentralised or centralised manner has yet been systematically elaborated. Brand and Ostfeld (2011) point out the general lack of optimisation models incorporating all the most basic system components such as sewers, WWTP and pumps at the same time, and Sitzenfrei et al. (2013) observe that tedious handling and processing of explicit geographic data is required to generate cost estimates for centralised infrastructures.

Nevertheless, there are important exceptions in the literature which cover the optimisation of wastewater infrastructure: Schiller (2010) uses GIS to determine where to start a transition towards decentralised wastewater management systems from existing sewer networks in case of a shrinking population. Zeferino et al. (2010) consider hybrid solutions and use simulated annealing to determine different optimal system configurations in a multi-objective framework. Leitão et al. (2005) compare a drop and a add algorithm to solve a location model at regional level.

[3] Two sewer modelling approaches can be distinguished, namely those that model actual case-specific sewer systems and those that estimate the material stock of the sewer infrastructures with the aid of virtual network layouts. As we focus on the optimisation process, and the detailed network design is of secondary interest, we refer to Maurer et al. (2012) for an overview.

2.1.4 Original contribution of the presented SNIP model

A brief overview of the literature on heuristic network optimisation shows that only few approaches consider hybrid constellations. In combination with sewer modelling, we can deduce four main shortcomings in the literature that the SNIP approach takes as a starting point:

- Even though a number of innovative methods exist to model sewer systems, only few of them explicitly address the ODC.
- Most optimisation approaches apply a dichotomic perspective, whereas real cases require hybrid constellations such as distributed wastewater systems with self-contained wastewater networks for any given landscape.
- The optimisation rule in most ODC models is limited to investment costs and straight-line distance calculations on flat terrain. Further costs are calculated independently of the position in the network and (dis-)economies of scale are not considered.
- A common limitation of all the approaches to network infrastructures (wastewater or other networks) mentioned so far is that they do not consider changes occurring in the physical network properties as the size of the network changes.

2.2 Model Description

2.2.1 Optimisation Function

The SNIP algorithm is based on cost and sewer-design assumptions and aims to determine the ODC by minimizing the overall system costs (C) of a wastewater system by considering the costs of WWTP of varying sizes, pumping and sewer costs. We solve the cost objective function (Eq. 2.1) by numerical computation.

$$Min\ C\ (N_{WWTP,}\ V_{WWTP}, l, d, V_{PUMP}, H) \qquad (2.1)$$

where the total system costs C depend on the number of WWTP (N_{WWTP}), the wastewater volume treated per WWTP (V_{WWTP}), the sewer network length (l), the sewer diameters (d), the pumped volume (V_{PUMP}) and the pump head at the duty point (H).

In each iteration step i, the values of the variables are changed and the new cost function C_{i+1} is generated and compared to C_i. The iteration stops when $C_{i+1} \geq C_i$ (see Fig. 2.1).

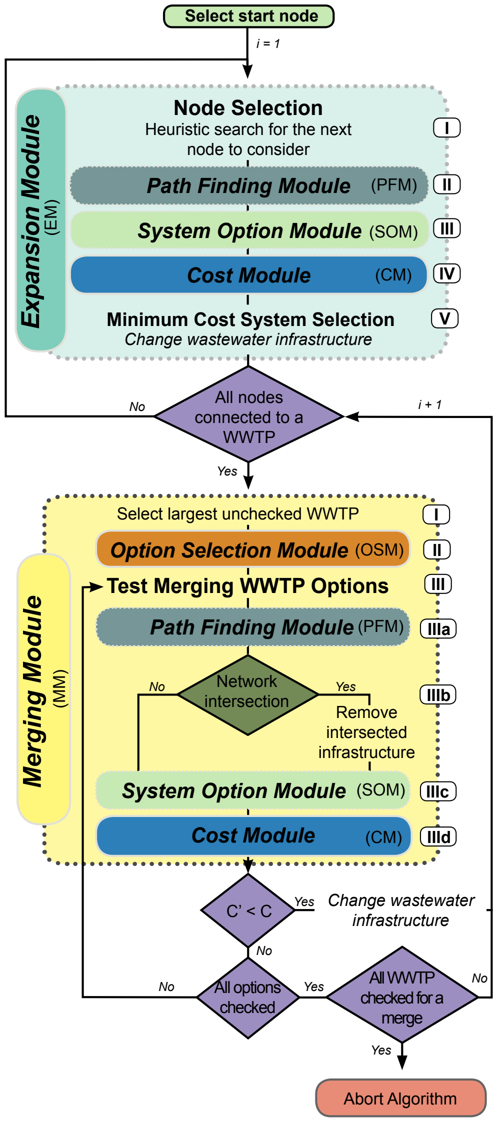


Figure 2.1: *SNIP algorithm workflow. The EM calculates an initial network layout until all nodes have a sanitation solution, while the MM optimises the infrastructure layout generated by the EM.*

2.2.2 SNIP Algorithm Modules

The SNIP algorithm is partitioned into two main consecutive functional modules, namely the expansion module (EM) and the merging module (MM) (Fig. 2.1). The EM is responsible for calculating a first system outlay whereas the MM improves overall cost savings by merging or agglomerating WWTP.

In a first step, the EM determines an initial set of WWTP and sewers which are defined from the bottom-up with shortest path-finding algorithms. In a second step, the MM looks for further cost savings by checking the potential merging of WWTP by means of heuristic agglomerative hierarchical clustering (Kaufman and Rousseeuw 2005).

Both modules execute sub-modules: the path-finding module (PFM) determines the path along which sewers are constructed. The system option module (SOM) identifies potential system options and the cost module (CM) determines the overall costs of each option. The algorithm terminates when no further cost decreases can be achieved by merging any WWTP.

The two main modules use greedy algorithms: these are characterized by the assumption that selecting the best-looking choice at each iterating step of the optimization procedure will yield an optimal global solution (Cormen et al. 2009). The assumption that local optimal choices result in a globally optimal solution is not generally true, even though it may be valid for many problems (Cormen et al. 2009). Given the problem complexity, finding reasonably approximate solutions is the only way forward given the restrictions of computation time. As decisions made in the EM can be altered in the MM, SNIP is neither an add nor a drop algorithm (Daskin 1995), but a mixture of both.

In the following sections, we describe the algorithm workflow with all sub-processes in more detail.

2.2.2.1 Expansion Module (EM)

The EM is based on Prim's algorithm (1957), which is well-known and widely applied in infrastructure planning and graph theory. It represents the sewer network as edges and houses, and WWTP as nodes. It then calculates a graph which connects all nodes with minimal edge weights to produce a minimum spanning tree (MST). Edge weights are generally derived from straight-line distances between nodes, but they can represent any metric such as time or costs. Prim's algorithm thus allows a least-cost network connecting all nodes to be found.

The use of gravity-driven sewer lines means that the actual path between two nodes may not be a straight line. So costs cannot be derived linearly from straight-line distances, and this makes it a complex task to attribute real costs to each edge. Thus sewer costs may depend on the direction of flow, the trench depth and any height differences encountered. More sophisticated methods are consequently needed for estimating costs.

We choose the following five-step approach to build a minimum network representing sewers and WWTP in a simplified manner (cf. Fig. 2.1):

Step I: We first select a starting node (household).[4] We then determine the minimum connection costs between this node and all still

[4] Due to the heuristic nature of the algorithm, the result is dependent on the starting node. Therefore we recommend that the algorithm be run with different starting nodes even though our case study results indicate low effects (Appendix B). Due to the logic of the algorithm, it makes sense to start at a node which lies in an area of high node density. These areas offer a greater chance that the total system costs will decrease by connecting nodes.

unconnected nodes. As the distance is important, the classical Prim-based approach of approximating connection costs between two nodes with straight-line distances seems plausible. Thus the assumption is made that the closest node is the best one for iteratively considering a network connection. In contrast to Prim's algorithm, we ask in each iteration whether a connection leads to cost minimisation, an approach which resembles the clustering idea of Zahn (1971), who removes edges from a fully calculated MST.

Step II: The sewers between the two detected nodes from Step I are designed with the path-finding module. The PFM determines the path with the aid of the street network and a digital terrain model (DTM). The motivation to use the street network is the close linkage between the two networks that is often found (Blumensaat et al. 2012, Nielsen and Möller 2013). However, this assumption may not always be true, especially if the distance along the street network is significantly longer where no street exists.
Our algorithm first identifies the direct distance d_{direct} between the two nodes from step I. The Dijkstra Algorithm (Dijkstra 1959) is applied to a street network to find the shortest distance between the next node to connect and the existing sewer network (d_{street}). The decision as to which sewer path to take is based on the ratio f_{street} between the direct distance (d_{direct}) and the distance along the street (Eq. 2.2).

$$f_{street} = \frac{d_{street}}{d_{direct}} \quad (2.2)$$

We derive f_{street} by comparing existing connection ratios in a given sewer network for an area of interest. So by changing this ratio, we can adapt the sewer design to local design practice. If f_{street} is larger than the derived ratio, an alternative sewer path following the local topography is calculated with help of the a* algorithm (Hart et al. 1968).

For the 3D path-finding methodology along the terrain, we build a graph from the raster-based DTM on which each centre raster point links all neighbouring cell centre points (queen neighbourhood) (Leitão et al. 2005). We derive the edge weights of the resulting graph from the height difference Δh between the raster cells and a weighting factor f_{topo} used to calculate a weighted distance d_w (Eq. 2.3).

$$d_w = d_{direct} \, |\Delta h|^{f_{topo}} \quad (2.3)$$

where f_{topo} can be altered depending on how closely the sewers should follow the topography. More sophisticated methods, such as land data use, could be applied to determine the weighting on anisotropic surfaces (Yu et al. 2003). However, the weighting is not of primary interest in this paper and the only restriction is that sewers cannot cross raster cells of the DTM containing buildings.

Step III: After the sewer path has been determined, three system options are always identified with the System Option Module (SOM, explained in Section 2.2.2.2), namely an option without sewer expansion and two options with a sewer expansion in either direction. We use the term system option to describe one system configuration. As different system options are available for selection in each iteration, this allows a cost-optimised system to be selected locally.

Step IV: Operation costs and replacement costs are attributed to the design alternatives defined in step III with the aid of the cost module (Section 2.2.2.4).

Step V: The choice for one of the options designed in Step III is made by considering reasonable costs (cf_{rc}). These costs are politically defined per capita cost values, which decide whether a decentralised option may be legally considered. Below that value, sewer connections are enforced. Similar criteria, such as distance measures, are used in many countries in what is known as the mandatory connection rule (e.g. Switzerland, Germany and Austria).

2.2.2.2 System Option Module (SOM)

The SOM creates different system options on the basis of the two nodes considered for connection in each iteration of the EM. A local competitive choice is then made from these options on the basis of cost calculations relating to all system elements. The modelled system elements are gravity driven and pressurized sewage pipes and WWTP. See Table 2.1 for all parameters influencing the design of the sewage system.

In each iteration, only two nodes are considered for designing system alternatives: this results in three possible options (Fig. 2.2). For two of these, the two nodes are connected and the network is consequently expanded. The existing WWTP is then either enlarged (option A), or else abandoned and a new one is built in the new node (option C). Alternatively, the new node is not connected and serviced by a separate WWTP (option B).

2.2.2.3 Merging Module (MM)

In the second step of the algorithm (see lower part in Figure 2.1), the MM optimises the configuration found by the EM by merging WWTP based on agglomerative hierarchical clustering (HAC), where we consider each WWTP with the corresponding network as a cluster. The motivation to merge plants lies in the economies of scale achieved as the per capita treatment costs decrease with growing networks and consequently larger WWTP.

Table 2.1: *Cost and design-related model parameters. The considered standard pipe diameters are (in m): 0.25. 0.3, 0.4, 0.5, 0.6, 0.7, 0.8, 0.9. 1, 1.2, 1.5, 2, 2.5, 3, 4, 6, 8.*

	Symbol	Unit	Base scenario value	Considered limits in eFAST analysis	
				Lower	Upper
Design Parameter					
Maximum trench depth	T_{max}	m	8	8	12
Minimum trench depth	T_{min}	m	0.25	-	-
Minimum slope	$f_{minslope}$	%	1	1	3
Sewer design factor	f_{street}	-	1.7	1	5
Sewer design factor	f_{topo}	-	1.4	1	2
Merging factor	f_{merge}	-	3	1	5
Wastewater production	Q_{ww}	$m^3 d^{-1}$ $capita^{-1}$	0.162	0.1	0.4
Strickler coefficient	k_{st}	$m^{1/3}s^{-1}$	85		
Pipe diameter	d	m	standard values		
Cost Parameter					
Sewers					
Sewer operating costs (VSA 2011)		$\$m^{-1}$	3.6		
Sewer pipe lifespan (Maurer and Herlyn 2006)	$cf_{sewerlifespan}$	y	80	60	100
Sewer replacement value (AWA 2001)	cf_{sewer}	%	0	- 20	+ 20
Sewage pumps					
Electricity costs (BFE 2011)		$\$kWh^{-1}$	0.14		
Pumping operation cost function (Grundfos 2014)		kWh	Section 2.2.4.2		
WWTP					
WWTP operating cost (VSA 2011)	$cf_{wwtpopex}$	%	0	- 20	+ 20
WWTP replacement value (VSA 2011)	$cf_{wwtpcapex}$	%	0	- 20	+ 20
WWTP lifespan (Maurer and Herlyn 2006)	$cf_{wwtplifespan}$	y	33	30	40
Other Parameters					
Real interest rate (Maurer and Herlyn 2006)	$cf_{interest}$	%	2	0	6
Reasonable costs (AWEL 2005)	cf_{rc}	\$	5357	0	14286

HAC is a distance-based bottom-up clustering algorithm in which each single object is treated as a cluster and then iteratively agglomerated until all objects are either merged or the algorithm is aborted on the basis of defined criteria (Manning et al. 2008). A typical property of HAC algorithms is that the number of clusters does not need to be defined a priori, which suits our need to find the optimal number of plants. The challenge of HAC methods is finding dissimilarity coefficients for cluster building. These coefficients reflect the dissimilarity between clusters and are often obtained from distance calculations or more complex computations (Kaufman and Rousseeuw 2005). For this study, we define the connection costs between WWTP as dissimilarities.

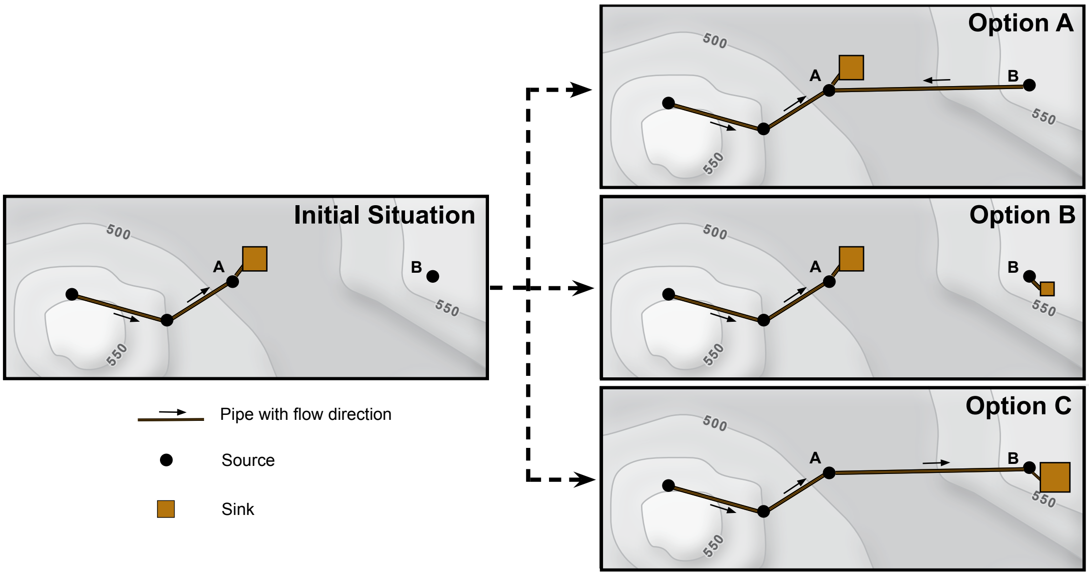


Figure 2.2: *System design options (SOM module) for an exemplary initial situation. Options A and C show a network expansion in combination with a WWTP enlargement. In option B the network is not enlarged and a new WWTP is installed instead.*

Because of the high calculation intensity of testing all merging possibilities or calculating the dissimilarity coefficients of all WWTP in each iteration, a heuristic selection of possible merges is made in the MM. The selection takes place in three major steps (compare Fig. 2.1):

Step I: As possible economies of scale can most probably be exploited by merging larger plants, each merge check is always started with the largest WWTP and is terminated as soon as all plants have been checked.

Step II: The three most promising WWTP to be considered for merging are determined with the aid of the SOM. The SOM finds the closest WWTP, the WWTP of the closest sewer network and the network with the highest merging potential $f_{MergePot}$. This potential is a distance-to-WWTP size ratio and is expressed as (Eq. 2.4)

$$f_{MergePot} = d\ (WWTP_{size})^{-f_{merge}} \quad (2.4)$$

where d is the distance between two nodes, f_{merge} the weighting factor and $WWTP_{size}$ the size of a WWTP given in population equivalents. The exponent f_{merge} allows us to increase the weighting for the size of the WWTP, thus decreasing the importance of the distance when choosing a WWTP to merge. This means that a higher merging potential is assigned to larger and more distant WWTP. We consider distance and size to be good criteria for selecting WWTP as the high cost of connecting more distant WWTP could be compensated thanks to economies of scale in wastewater treatment. Figure 2.3 explains the various possibilities of the SOM. Let us consider facility C in the illustrated example and determine the three WWTP to be checked for a merge. The closest facility is B, the facility with the closest sewer D and the facility with the best merging potential index is A because of its larger size.

Step III: The WWTP identified in Step II are tested for a merge. The sewer path between two WWTP is derived from the PFM (IIIa), the sewage system options found (IIIc) and the costs calculated (IIId). In the process of finding interconnecting sewer paths between WWTP, other sewer networks may be crossed. In such cases, the intersected network elements are removed from the current network (IIIb) and are reconnected with the EM in case of reduced system costs.

2.2.2.4 Cost Module (CM)

The SNIP algorithm finds an optimal wastewater management configuration by minimizing operation and capital replacement costs, which are calculated with help of the CM. In order to compare the different costs, we calculate the total replacement costs and convert them to equivalent uniform annual cash flows or annuities. The annuities A can be calculated from a net present value (NPV) written as (Eq. 2.5) (Crundwell 2008).

$$A = NPV \frac{q^n(q-1)}{q^n-1} \qquad (2.5)$$

where q is the (real) interest rate + 1 and n the number of years for depreciation. All local currencies are converted to US$ using purchase power parities for the year 2013 (World Bank 2014). All cost factors used are listed in Table 2.1.

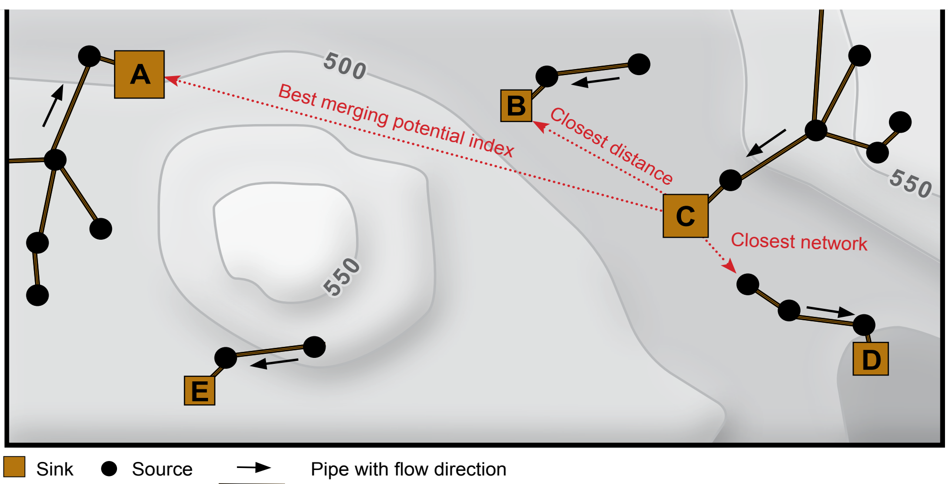


Figure 2.3: *Exemplary representation of the WWTP selection by the SOM heuristic for WWTP C. B is closest to C, D has the closest network to C whereas A has the best merging potential for C due to its size (see Equation 4).*

Sewers

As sewer construction costs depend on numerous factors, it is problematic to derive general costs. We reduce the cost factors to the trench depth, pipe diameter and sewage pipe length in accordance with a cost model from the case study area (AWA 2001) which relies on Swiss sewer construction standards. The sewage replacement costs c are calculated with the aid of the average trench depth T_{avg} and the cost coefficients a and b relating to the pipe diameter (Eq. 2.6):

$$c = aT_{avg} + b \qquad (2.6)$$

We calculate the sewer diameters using a standard engineering approach according to Manning-Strickler (compare for example Maurer et al. 2012). A maximum trench depth restriction TD_{max} prevents the construction of sewage pipes too deep underground. If the minimum slope restriction ($f_{minslope}$) cannot be maintained because of TD_{max}, the wastewater is pumped. The parameter $f_{minslope}$ describes the slope of the sewers which need to be constructed in order to allow gravity-driven flow. Therefore $f_{minslope}$ does not represent the slope of the terrain. In case of steep terrain, the sewer slope is similar to the terrain slope. In flat terrain, the slope corresponds to the value given by $f_{minslope}$. Sewer operation costs are taken from the literature and set to average costs per metre per year (VSA 2011) (see Appendix A).

Pumps

Wastewater is pumped wherever the topography does not provide enough downward gradients. We use a very simplified approach for calculating pumping costs. Given the genericness of the plain model design, we do not consider costs resulting from the need to provide pumping redundancy, potential wastewater storage costs for pump sumps, or cost differences depending on the pump size. Furthermore, we do not consider economies of scale, but only assign a fixed cost for a pumped volume. As a consequence, SNIP does not minimize the number of pumps but only the sewer length where pumping is required. Further SNIP generally neglects different kinds of implications such as odour problems or hygienic challenges resulting from long residence times.

We choose a methodology to estimate the needed power input P_{gr} from a standard engineering sewage pumping handbook (for example Grundfos 2014) (Eq. 2.7):

$$Pg_r = \frac{gQH}{n_{gr} * 1000} \qquad (2.7)$$

Pgr: motor power input [kW]
Q: pump volume flow at duty point [l/s]
H: pump head at duty point [m]
g: gravitational constant [m/s²]
n_{gr}: overall energy conversion efficiency

The total cost of the energy consumption for one year is calculated by multiplying P_{gr} with the running time per year and the specific average pumping costs.

Wastewater treatment plants
According to Friedler and Pisanty (2006), WWTP cost functions are best expressed by a power law (Eq. 2.8)

$$c = ax^b \qquad (2.8)$$

where the costs c are estimated by defining x as the plant capacity in population equivalents and using the cost coefficients a and b.

We found it challenging to determine a single generic cost function over the entire range of possible WWTP sizes. The available data indicate that smaller package treatment plants show a different cost scaling behaviour than the larger custom-built ones. The operating-cost and replacement-cost functions for the WWTP used in this paper are taken from VSA (2011) derived from larger WWTP.

2.3 Materials and Methods

In order to test the adequacy of the SNIP algorithm, we carried out the following analysis steps. First we defined the degree of centralisation. Second we determined the influence of SNIP variable changes with the aid of a sensitivity analysis in order to determine whether we could distinguish between important and less important variables. Third, we conducted a total of 250 model runs for different topographies in order to determine whether SNIP gives reasonable representations of possible WWTP and sewer outlays.

2.3.1 Defining the Degree of Centralisation

The current discussion about central or decentral infrastructure planning is often fuzzy due to a lack of clear definitions. In practice, simple measures, such as the dimension (e.g. treated volume) or vague terms relating to the served area (e.g. small) or distance (e.g. close) are often used to define decentralised treatment plants (cf. Makropoulos and Butler 2010, DIN 4261 2010, EPA 2005, Cook et al. 2009). However, such a definition is problematic in two ways: first, the understanding of the terms "centralised" or "decentralised" depends on the chosen system boundaries, as we can define a continuum of different wastewater system scales (Hamilton et al. 2004). Second, the definition of the ODC is often limited to two categories: a source is either fully connected or entirely decentralised. Such a dichotomic definition of system alternatives is unrealistic as a whole range of intermediate solutions may be possible.

A more systematic definition taking into account the continuum of possible facility sizes is adapted from Ambros (1996) (Eq. 2.9):

$$DC = \frac{\sum_{i=1}^{n} N_i - \sum_{j=1}^{m} \frac{M_j}{B_j}}{\sum_{i=1}^{n} N_i} \qquad (2.9)$$

where we define a weighted degree of centralisation (DC). For this paper, M denotes the volume of wastewater which needs to be treated at a sink (treatment plant), N the volume of wastewater originating from a source (household) and B the number of sources connected to a sink. We sum over all sources $(i = 1,\dots,n)$ and sinks $(j = 1,\dots,m)$. Compared to the original definition, the DC allows us to consider different source weights, as the required wastewater quantity to be treated at the sources may differ. If DC is 0, we find complete decentralisation with a sink placement at each source. If treatment takes place only outside the considered area, the DC reaches 1 (Fig. 2.4).

2.3.2 Case Studies

In order to test SNIP under varying system conditions, we introduce virtual case studies (Section 2.3.2.1) and apply SNIP to a real-world case (Section 2.3.2.2). It is problematic to validate the model results with real world data because existing systems have grown historically and mostly constitute combined sewer systems. This means that even newly designed

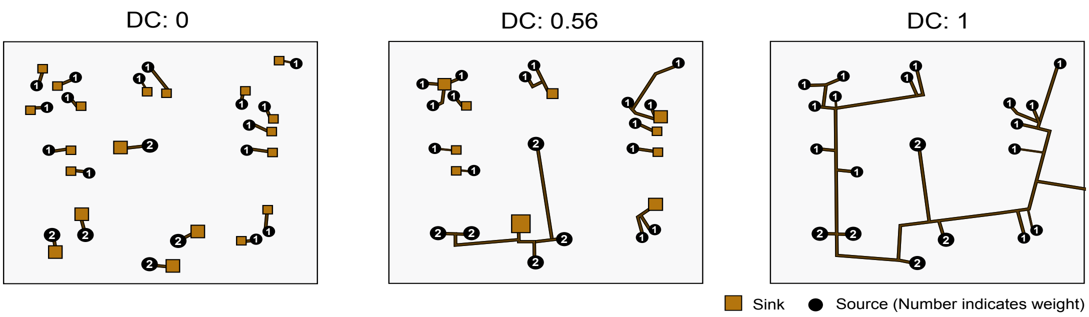


Figure 2.4: *Example calculations of DC. The characteristic of DC can be seen in the situation in the middle, where on average two nodes are connected to a plant, but we calculate a value higher than 0.5 because of the merging of nodes with higher weights.*

systems would look different. An advantage of the virtual case study approach is that we can easily generate and test SNIP for a broad set of different conditions. On the basis of the real world application, we can show the potential of SNIP for a given Swiss context in an exemplary way.

2.3.2.1 Virtual Case Studies

In order to better understand our algorithm, we generate contrasting virtual cases with real world topographies but virtual settlement distributions and use face validation to see whether the input-output rela-

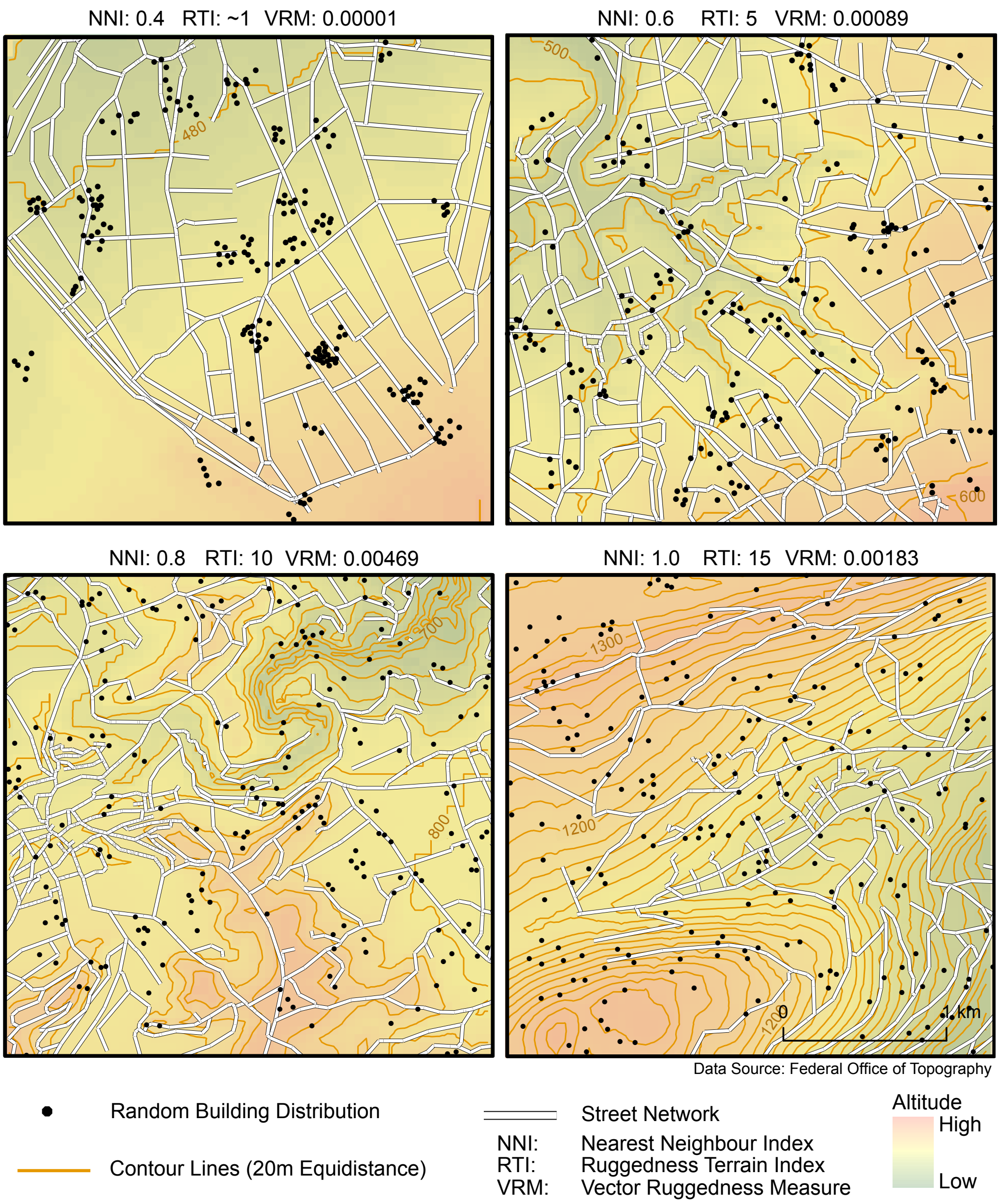


Figure 2.5: *Overview of virtual case studies. A different exemplary settlement distribution is displayed for each topography. We use real world topography and street networks but redistribute the buildings in order to achieve a different source clustering.*

tionships of the model are reasonable (Sargent 1991). The virtual case study allows us to observe whether the model can be sensibly applied in different contexts considering completely different topographies or settlement distributions. We use the ruggedness terrain index (RTI) (Riley et al. 1999) and the vector ruggedness measure (VRM) (Sappington et al. 2007) to quantify terrain complexity, and the nearest neighbour index (NNI) (Clark and Evans 1954) to quantify the degree of clustering of the inhabited buildings.

The virtual case studies (Fig. 2.5) are created as follows: we select four clippings (of 9 km^2 each) from the digital elevation model of Switzerland and the respective street networks. By calculating the RTI and VRM, we are able to select topographically contrasting cases. We then create different virtual settlement distributions (with 200 buildings) on the selected clippings with nearest neighbour indices ranging from 0.2 to 1. We assume that the amount of wastewater flow is equal for each building.

2.3.2.2 Real World Case Study

The SNIP model was applied to the community of Trubschachen (~1500 inhabitants, 365 buildings) in the Emmental region of western Switzerland. This region is hilly, relatively sparsely populated and makes network infrastructure planning challenging because of its complex topography and settlement distribution. Today's relatively high presence of on-site solutions in this region already indicates a borderline situation for the central network paradigm. Based on the current distribution of small WWTP and network outlay of Trubschachen, we calculate the actual DC as 0.85.

We assign an average wastewater production to the number of people living in a building. Access to population distribution data on a high spatial scale is often problematic either because of missing data or due to privacy concerns. Therefore we spatially disaggregate the population with the aid of a dasymetric mapping technique developed by Lwin and Murayama (2009).

We run a variance-based sensitivity analysis in order to quantify the total effect of each parameter on the model output for the real world case study. The extended Fourier Amplitude Sensitivity Test (eFAST) of Saltelli et al. (1999) allows us to cope computationally with a large number of factors and take the interactions between them into account (Crosetto et al. 2000). The analysis is performed in R with the R package "sensitivity" of Pujol (2014). As there is no exact rule for finding an adequate sample size of eFAST, we use a number close to the minimum known value (Marino et al. 2008). For eFAST, we do not consider changing starting nodes and start with a node located in a densely populated area.

2.3.3 Data and Software

SNIP was developed to be as economical as possible with regard to data requirements. All data are generally easily accessible and were obtained from the Swiss Federal Office of Topography (see Appendix C). SNIP is implemented in Python 2.7.3. ArcGIS® 10.2 is used for reading and visualisation purposes.

2.4 Results and Discussion

2.4.1 Sensitivity Analysis

The result of the sensitivity analysis in Table 2.2 for the real world case study shows that sewer design factors have a predominantly greater effect on the ODC even though the differences between individual factors are generally not very distinct. The analysis shows that the sewer design factor f_{street} (main effect of 0.34) that characterises when to follow the street and when to build sewers along the terrain has a particularly large impact on the ODC. This emphasises the importance of determining the relationship between the given street network and the sewer outlay for each case study. Similarly, other sewer-related design factors such as the minimal slope, f_{street} (main effect of 0.20), or the maximum trench depth T_{max} (main effect of 0.16) are also sensitive. The high general interaction effects of all parameters, indicating a high correlation between them, are not unexpected, as many of these parameters have a direct influence on costs, and thus to a change of DC. As many of these parameters relate to real-world characteristics, it is possible to treat them as input parameters

Table 2.2: *eFAST results (sample size = 70). See Table 2.1 for a more detailed description of the parameters.*

Parameter	**Description**	**Main Effect**	**Interaction Effect**
Q_{ww}	Wastewater production	0.0364	0.4390
$c_{fwwtplifespan}$	WWTP lifespan	0.0665	0.4928
$cf_{wwtpopex}$	WWTP replacement value	0.0881	0.4104
cf_{sewer}	Sewer replacement value	0.0884	0.5283
$cf_{sewerlifespan}$	Sewer pipe lifespan	0.0886	0.4113
$cf_{interest}$	Real interest rate	0.0973	0.8000
f_{topo}	Sewer design factor	0.0993	0.5585
$cf_{wwtpcapex}$	WWTP replacement value	0.1318	0.4111
f_{merge}	Merging factor	0.1518	0.6279
T_{max}	Maximum trench depth	0.1567	0.5760
cf_{rc}	Reasonable cost	0.1762	0.6142
$f_{minslope}$	Sewer design factor	0.1977	0.5927
f_{street}	Sewer design factor	0.3408	0.8657

and obtain sensible values for a given application case. As a consequence, only three 'real' model parameters remain, f_{topo}, f_{merge}, and f_{street}, all three of which are sensitive and correlated with other parameters.

2.4.2 Face Validation Virtual Case Studies

We are testing the proposed SNIP algorithm in the four virtual case studies shown in Fig. 2.5. They differ with respect to terrain ruggedness and source clustering. We expect lower degrees of centralisation (lower DC values) wherever we encounter high terrain complexity and distributed sources due to higher network construction costs. We find this

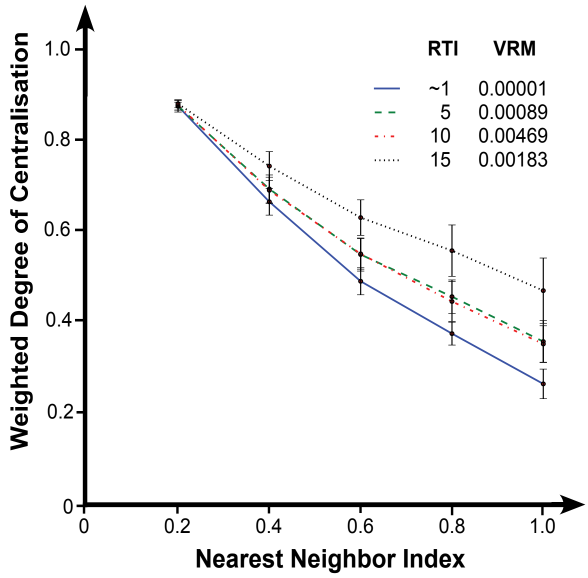


Figure 2.6: *SNIP results for virtual case studies with different degrees of source clustering and different topographic complexities. We distributed 200 buildings and generated 50 model runs in each case. The error bars show the standard deviation of the 50 settlement distributions for each situation.*

general pattern to be true for our virtual case studies. Figure 2.6 shows a very distinctive dependency of DC on the NNI. The effect of the terrain complexity is much less visible.

We notice that the DC does not always decline with increasing RTI values. Despite high RTI values due to large even flanks, such a topography favours gravity-driven sewer construction. This is reflected in the VRM index, which we use to distinguish steep even terrain from steep uneven terrain (Sappington et al. 2007). Therefore the choice of index matters when relating topographical complexity to DC.

2.4.3 Real World Case Study

We ran our algorithm for the community of Trubschachen and calculated an ODC of 0.76 (Appendix B). Figure 2.7 shows annuities for different DC for this catchment. We see that the overall costs decrease

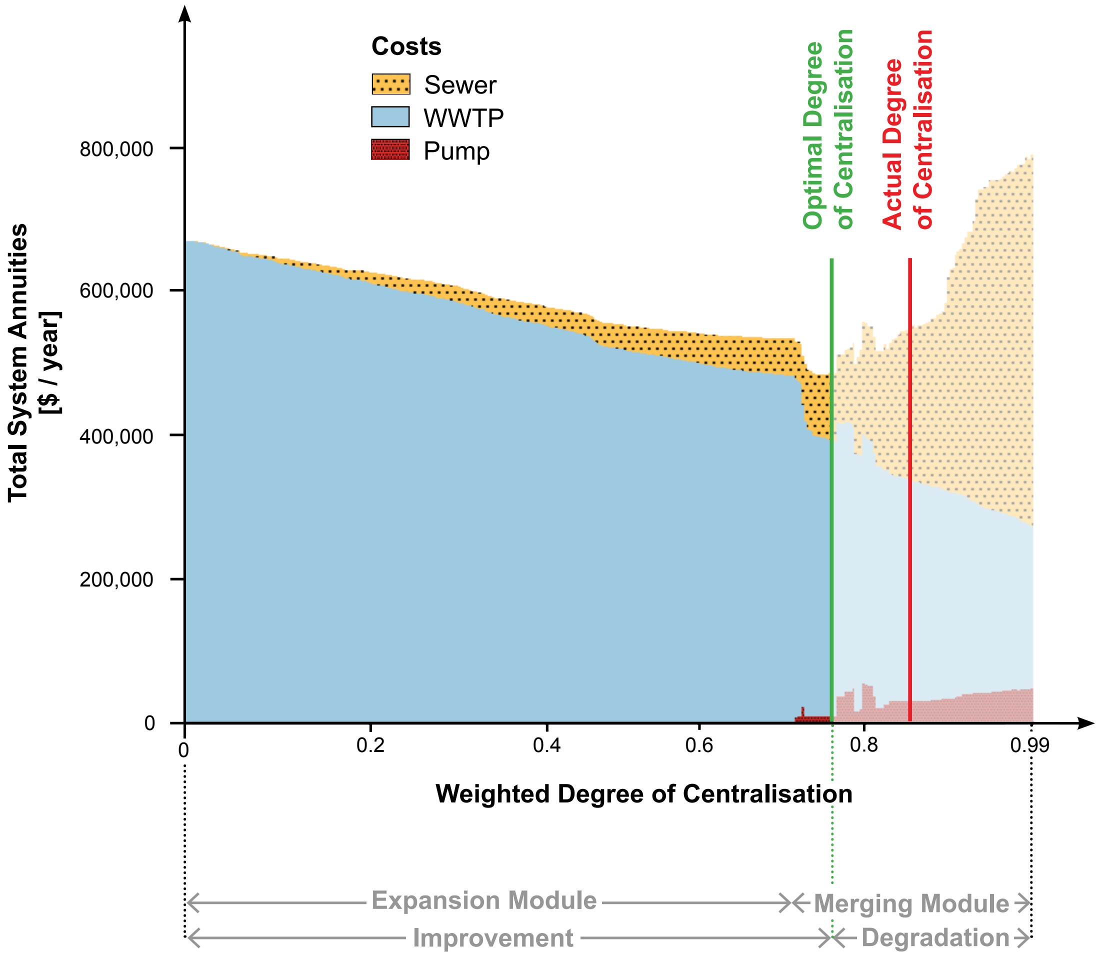


Figure 2.7: *Total system annuities of Trubschachen as a function of DC. The cost shares of the different system elements shift with increasing DC from WWTP costs towards sewer and pumping costs until minimum total system costs are reached at DC = 0.76.*

with increasing centralisation due to a decrease of WWTP costs and a relatively slow increase in sewerage costs. This is valid to the proposed optimal centralisation degree where DC = 0.76. After this, the costs for sewer lines and pumping costs exceed the economies of scale of the WWTP. We have extended the calculations of the total system costs represented in Fig. 2.7 beyond the ODC in order to illustrate the consequences of forced centralisation and as well as to allow a comparison with the actual degree of centralisation. The initial gradual decrease takes place in the EM whereas the cost drop at about 0.72 results from merging (agglomerating) WWTP within the MM. The increasing marginal sewer connection costs are particularly noticeable where DC is close to 1, which shows the high costs of connecting the most remote settlements.

The calculated DC is lower than the effective centralisation achieved in Fig. 2.8. We observe that sewers follow the street network in the urban area more closely and deviate more for single rural buildings, which is plausible and corresponds to the real situation (compare Blumensaat et

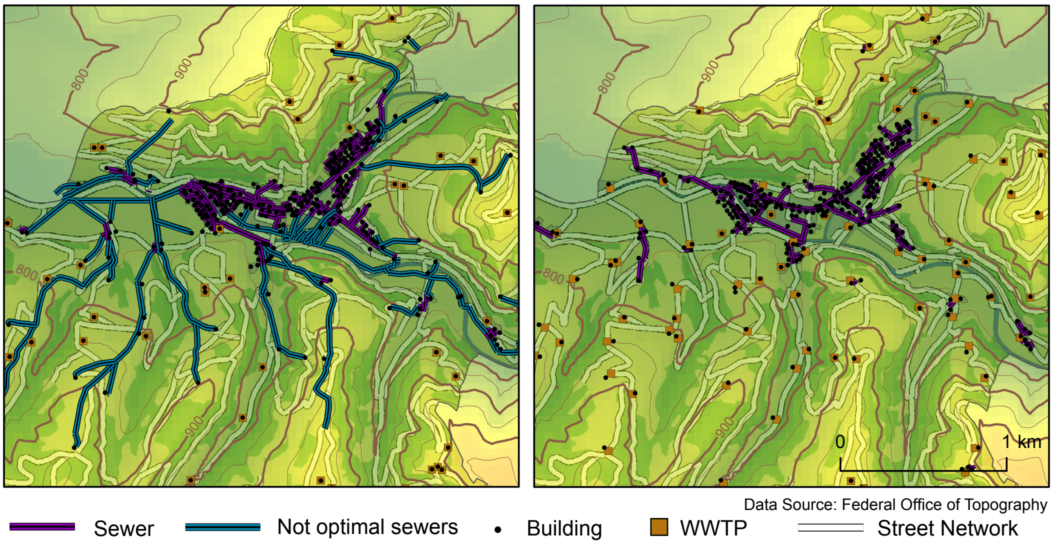


Figure 2.8: *Today's wastewater system connecting the inhabited buildings (left) and optimum system design calculated with SNIP using the base parameters (right). We assume that all inhabited buildings which are not connected to the sewers currently have an on-site treatment solution.*

al. 2012). Figure 2.8 indicates that in reality more buildings were connected to the central system than the economically optimal number. In the real case, the implementation of sewer lines stopped only when pumping costs substantially increased. Visual inspection of Fig. 2.8 confirms that the two system settings differ mostly by quite remote settlements (blue sewers in Fig. 2.8).

Nonetheless, the difference between today's DC and the ODC fits well for Switzerland in general as well as for Trubschachen, whose wastewater infrastructure was largely built during the economic boom of the 1960s, 70s, and 80s, when on average 37% of wastewater evacuation costs was subsidized (Müller and Kramer 2000, Maurer and Herlyn 2006). Additionally, a lot of infrastructure was planned and built at a time when small treatment plants had a distinctly worse performance compared to large ones, which was the reason for the subsidies. So it is not surprising that today's network system is over-dimensioned from a cost efficiency point of view. We see that SNIP allows decision makers to re-asses the economic efficiency of a given system and to consider disconnecting certain households or at least delay rehabilitation projects until decentralised systems can be implemented.

2.4.4 Limitations and Research Needs

These results highlight an important aspect of the SNIP approach, namely that it is a single-objective approach exclusively focussing on cost minimisation and thus ignores other performance or sustainability goals that a wastewater system could fulfil. An important assumption underlying the current approach is that all possible system configurations (from fully centralised to fully decentralised) achieve the same performance. There are good indications that this last strong assumption might become superseded by current research efforts on small-scale treatment systems (see also Larsen et al. 2013). Other important limitations of the SNIP approach are:

- The presented cases contained only foul sewers. For storm sewers, it is less a question of treatment than of transportation, and is dealt with in the literature (inter alia Urich et al. 2013, Bach et al. 2014). Expanding SNIP with combined sewers is fairly simple, as it only requires the design rain input for each source and the identification of suitable combined sewer overflow points.
- It does not consider the currently existing network infrastructure. SNIP provides a pseudo- or quasi optimal situation for a given catchment, ignoring any transition scenarios needed to transform an existing infrastructure.
- SNIP is static, ignoring dynamic changes in settlement patterns or changing input parameters. The results for the presented case studies show that changing settlement structures are of particularly great importance for the ODC.

The last two points (transitions and scenario planning) in particular need to be addressed if SNIP is to serve as a more realistic planning tool. It is important to realise that SNIP cannot currently be seen as a prescriptive tool for system implementation, but more as a form of guidance about the momentary sensible extent of the network infrastructure. SNIP can contribute an additional perspective in a system planning process by providing cost-effective alternatives. We believe that SNIP not only has value for planning new infrastructure but also in guiding or stimulating infrastructure transitions for existing sewer networks. This is increasingly important in contexts where major investments need to be made in existing infrastructures.

Additionally, more research is needed to determine better cost functions depending on the particular case study. Whereas we consider model uncertainty as a minor problem, the standard deviation of our random distribution in Fig. 2.6 and the starting node uncertainty in Fig. B.2.1 indicate that different results may be obtained depending on the chosen input parameters. But we argue that such uncertainty could even serve as a valuable input for a planning process.

There are a number of other ways in which the SNIP approach may be further developed. We especially see potential in broadening the set of criteria to address the sustainability of network infrastructure planning in a holistic way.

2.5 Conclusions

We present the heuristic SNIP algorithm as a tool to model the optimal degree of centralisation (ODC) for wastewater infrastructures. We consider the optimal number, placement and sizing of wastewater treatment facilities, gravity-driven and pressurised sewer networks as a fixed-charge location problem and use heuristics to find cost-minimised solutions.

SNIP is generic and uses only basic data input, thus allowing easy transfer to other case studies. We find that the SNIP algorithm can generate interesting plausible suggestions for sewer networks on a small scale and also produce face-value plausibility in virtual case studies. In-depth analyses will need to follow in the event of possible implementation. The approach presented here considers economies of scale, calculates costs depending on network position and considers the influence of the topography on sewer design when addressing the question of ODC. Most importantly, it takes into account different sizes of treatment plants and is applicable to local scale analysis. It also allows us to go beyond the often fruitless discussion about the appropriateness of on-site versus fully centralised solutions. Moreover, the combination of quantitative measures for settlement distribution and topographic complexity used for the calculated ODC allows us to quickly derive estimates of the ODC for different case studies. The real-world application of SNIP to a Swiss community suggests that the prevailing sewer system is over-centralised. Thus the SNIP-ODC may guide decision-makers to ask the right questions about the cost-efficiency of the current infrastructure layout and demonstrates that questions relating to current planning approaches need to be addressed in more detail. Knowing the ODC represents valuable information, especially in those cases in which new infrastructure needs to be built or already built infrastructure has to be redeveloped.

SNIP is based on heuristics, so the ODC solutions found are (pseudo-) optimal with regard to a rather restricted set of criteria. Even though its artificially generated wastewater systems are based on real world sewer-design principles, our model in no way replaces detailed engineering decisions on the ground. SNIP depends on generic design and cost parameters, and in combination with the model uncertainty it is obvious that DC values obtained can only be approximate.

The application of tools such as SNIP is especially promising in the context of changing futures such as changing settlement patterns and shrinking or growing populations. SNIP has so far been applied on a local scale and needs to be extended to a regional scale. We believe that further improvement of our static one-dimensional optimisation process towards a multi-objective framework taking into account different context conditions will deliver insights into a possible sustainability transition (Coenen and Truffer 2012).

Acknowledgements

We thank Christoph Egger for his support with the cluster calculations and Andreas Scheidegger for his help with the sensitivity analysis. We are grateful for general advice from Robert Weibel. We also thank Ruefer Ingenieure AG for providing their data and Richard Michell for proof reading. Furthermore, we are grateful for the fruitful feedback from our three anonymous reviewers.

Source Code

The source code and an ArcGIS-Toolbox are available from: https://github.com/eggimasv/SNIP ■

Appendix A

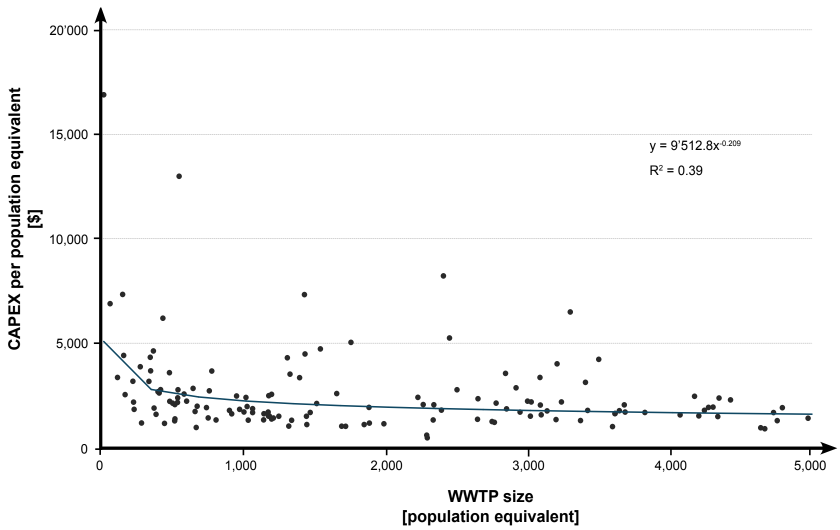


Figure A.2.1: *WWTP capital expenditure cost curve from* VSA (2011).

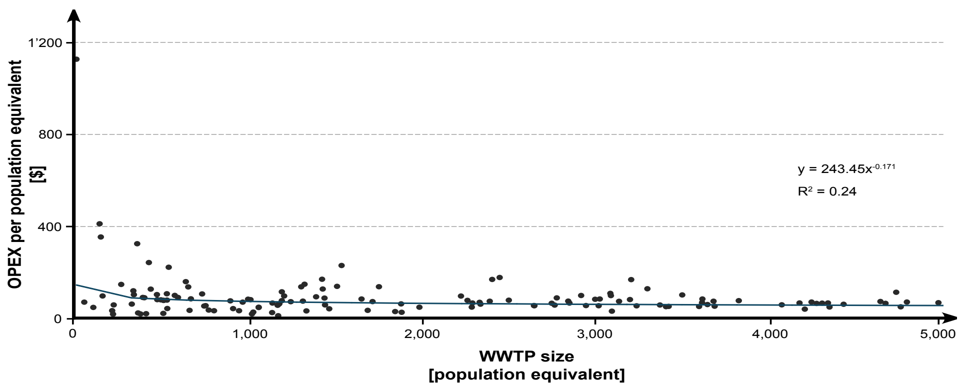


Figure A.2.2: *WWTP operation expenditure cost curve from* VSA (2011).

Appendix B

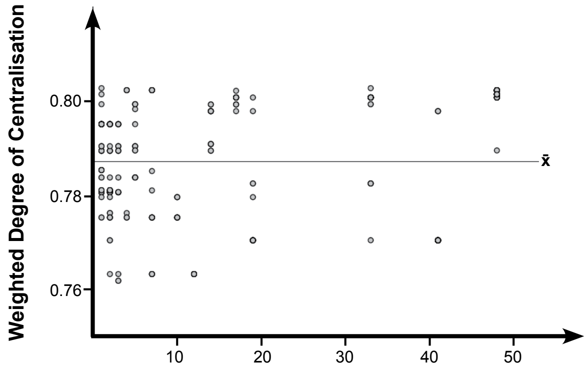


Figure B.2.1: *Case study results for Trubschachen. We run SNIP from each start node (n = 362), which results in a DC ranging from 0.76 to 0.80 ($\bar{x}$ = 0.787, σ= 0.01)*

Appendix C

Data	Description	Source
Digital terrain model with a resolution of 25m x 25m	Raster	swisstopo
Population data on community level	-	swisstopo
Street network	Vector	swisstopo
Buildings	Vector	swisstopo

Table C.2.1: *Data sets used for SNIP.*

2.6 References

Abd El Gawad, H., Butter, J., **1995**. Clustering of towns and villages for centralized wastewater treatment. *Water Science and Technology*, 32(11), 85–95.

Ambros, R., 1996. Application of mathematical optimization methods for variant calculations (Anwendung mathematischer Optimierungsmethoden in der Variantenrechnung). Wiener Mitteilungen, Band 130, 107–133 (in German).

AWA, **2001**. Approach for calculating costs of sewers (Vorgehen zur Bestimmung der Kosten von Abwasserkanälen), Amt für Abwasser und Abfall des Kantons Bern, Bern, Switzerland (in German).

AWEL, **2005**. Guidelines concerning mandatory connection of property to the private and public sewerage system (AWEL-Richtlinien betreffend die Anschlusspflicht von Liegenschaften an die private und öffentliche Kanalisation.) Accessed: 05.05.2015 Archived by WebCite® http://www.webcitation.org/6Y-IKrL8cV (in German).

Bach, P.M., McCarthy, D.T., Urich, C., Sitzenfrei, R., Kleidorfer, M., Rauch, W., Deletic, A., **2013**. A planning algorithm for quantifying decentralised water management opportunities in urban environments. *Water Science and Technology*, 68(8), 1857–1865.

Bach, P.M., Rauch, W., Mikkelsen, P.S., McCarthy, D.T., Deletic, A., **2014**. A critical review of integrated urban water modelling – Urban drainage and beyond. *Environmental Modelling & Software*, 54, 88–107.

Baufumé, S., Grüger, F., Grube, T., Kroeg, D., Linssen, J., Weber, M., Hake, J.-F., Stolten, D., **2013**. GIS-based scenario calculations for a nationwide German hydrogen pipeline infrastructure. *International Journal of Hydrogen Energy,* 38 (10), 3813–3829.

BFE, **2011**. Development of the Swiss electricity price (Strompreisentwicklung in der Schweiz). Neuchâtel, Switzerland (in German).

Blumensaat, F., Wolfram, M., Krebs, P., **2012**. Sewer model development under minimum data requirements. *Environmental Earth Sciences,* 65(5), 1427–1437.

Brand, N., Ostfeld, A., **2011**. Optimal Design of Regional Wastewater Pipelines and Treatment Plant Systems. *Water Environment Research,* 83(1), 53–64.

Clark, P.J, Evans, F.C., **1954**. Distance to Nearest Neighbor as a Measure of Spatial Relationships in Populations. *Ecological Society of America*, 35(4), 445–453.

Coenen, L., Truffer, B., **2012**. Places and Spaces of Sustainability Transitions: Geographical Contributions to an Emerging Research and Policy Field. *European Planning Studies,* 20(3), 367–374.

Cook, S., Tjandraatmadja, G., Ho, A., Sharma, A., **2009**. Definition of Decentralised Systems in the South East Queensland Context. Urban Water Security Research Alliance, Technical Report No. 12. City East, Australia.

Converse, A.O., **1972**. Optimum number and location of treatment plants. *Water Pollution Control Federation,* 44(8), 1629–1636.

Cormen, T.H., Leiserson, C. E., Rivest, R. L., Stein, C., **2009**. Introduction to Algorithms. MIT Press, Cambridge, Massachusetts, London, UK.

Crosetto, M., Tarantola, S., Saltelli A., **2000**. Sensitivity and uncertainty analysis in spatial modelling based on GIS. *Agriculture, Ecosystems and Environment,* 81, 71–79.

Crundwell, F.K., **2008**. Finance for Engineers. Evaluation and Funding of Capital Projects. Springer, London, UK.

Current, J., Daskin, M., Schilling, D., **2002**. Discrete network location models. Chapter 3. In: Drezner, Z., Hamacher, H. (eds), *Facility Location Theory: Applications and Methods*. Springer-Verlag, Berlin, 81–118.

Daskin, M.S., **1995**. Network and discrete location – models. In: *Algorithms and Applications.* Wiley, New York, USA.

Deichmann, U., Meisner, C., Murray, S., Wheeler, D., **2011**. The economics of renewable energy expansion in rural Sub-Saharan Africa. *Energy Policy,* 39(1), 215–227.

Dijkstra, E.W., **1959**. A note on two problems in connexion with graphs. *Numerische Mathematik,* 1, 269–271.

DIN **4261**, 2010. Small sewage treatment plants, English Version of DIN 4261:2010-10. DIN, Germany.

Downing, P. B., **1969**. The economics of urban sewage disposal. Praeger, New York, USA.

EPA, **2005**. Handbook for Managing Onsite and Clustered (Decentralized) Wastewater Treatment Systems. Environmental Protection Agency. Cincinnati, United States.

Friedler, E., Pisanty, E., **2006**. Effects of design flow and treatment level on construction and operation costs of municipal wastewater treatment plants and their implications on policy making. *Water Research*, 40, 3751–3758.

Garey, M.R., Johnson, D.S., **1979**. Computers and Intractability: A Guide to the Theory of NP-Completeness. W.H. Freeman & Co. New York, NY, USA.

Geels, F.W., **2006**. The hygienic transition from cesspools to sewer systems (1840–1930): The dynamics of regime transformation. *Research Policy*, 25, 1069–1082.

Gils, H.C., Cofala, J., Wagner, F., Schöpp, W., **2013**. GIS-based assessment of the district heating potential in the USA. *Energy,* 58, 318–329.

Gochenour, C., **2001**. District energy, trends, issues, and opportunities. The Role of the World Bank. World Bank Technical Paper No. 493. Washington, D.C., USA.

Graham, S., Marvin, S., **2001**. Splintering urbanism: Networked infrastructures, technological mobilites and the Urban Condition. Routledge, London, UK.

Grundfos (eds) **2014**: The sewage pumping handbook: Grundfos wastewater. Accessed at: 30.04.2015. Archived by WebCite® http://www.webcitation.org/6YB8zRK65

Hart, P.E., Nilsson, N.J., Raphael, B., **1968**. A Formal Basis for the Heuristic Determination of Minimum Cost Paths. *IEEE Transactions on Systems Science and Cybernetics SSC4* (2), 100–107.

Hamilton, B.A., Pinkham, R.D., Hurley, E., Watkins, K., Lovins, A.B., Magliaro, J., Etnier, C., Nelson, V., **2004**. Valuing Decentralized Wastewater Technologies: a Catalog of Benefits Costs and Economic Analysis Techniques. Rocky Mountain Institute, Snowmass, USA.

Hawkey, D.J.C., **2012**. District heating in the UK: A technological innovation systems analysis. *Environmental Innovation and Societal Transitions,* 5, 19–32.

Hughes, T.P., **1983**. Networks of power: electrification in western society, 1880-1930. JHU Press, London, UK.

Johnson, N., Yang, C., Ogden, J., **2008**. A GIS-based assessment of coal-based hydrogen infrastructure deployment in the state of Ohio. *International Journal of Hydrogen Energy*, 33 (20), 5287–5303.

Kaufman, L., Rousseeuw, P.J., **2005**. Finding Groups in Data. An Introduction to Cluster Analysis. An Introduction to Cluster Analysis, John Wiley & Sons, Hoboken, NJ, USA.

Kocaman, A.S., Huh, W.T., Modi, V., **2012**. Initial layout of power distribution systems for rural electrification: A heuristic algorithm for multilevel network design. *Applied Energy*, 96, 302–315.

Larsen, T. A., Udert, K. M., Lienert, J., **2013**. Source Separation and Decentralization for Wastewater Management. IWA Publishing, London, UK.

Leitão, J.P., Matos, J.S., Gonçalves, A.B., Matos, J.L., **2005**. Contribution of Geographic Information Systems and location models to planning of wastewater systems. *Water Science and Technology,* 52(3), 1–8.

Levin, T., Thomas, V.M., **2012**. Least-cost network evaluation of centralized and decentralized contributions to global electrification. *Energy Policy,* 41, 286–302.

Libralato, G., Ghirardini, A.V., Avezzù, F., **2012**. To centralise or to decentralise: An overview of the most recent trends in wastewater treatment management. *Journal of Environmental Management,* 94, 61–68.

Lofrano, G., Brown, J., **2010**. Wastewater management through the ages: A history of mankind. *Science of the Total Environment,* 408, 5254–5264.

Lwin, K., Murayama, Y., **2009**. A GIS Approach to Estimation of Building Population for Micro-spatial Analysis. *Transactions in GIS,* 13(4), 401–414.

Makropoulos, C.K., Butler, D., **2010**. Distributed Water Infrastructure for Sustainable Communities. *Water Resources Management*, 24(11), 2795–2816.

Manning, C. D., Raghavan, P., Schütze, H., **2008**. Introduction to Information Retrieval. University Press, Cambridge, UK.

Marino, S., Hogue, I.B., Ray, C.J., Kirschner, D. E., **2008**. A methodology for performing global uncertainty and sensitivity analysis in systems biology. *Journal of Theoretical Biology,* 254, 178–196.

Marlow, D.R., Moglia, M., Cook, S., Beale, D.J., **2013**: Towards sustainable urban water management: A critical reassessment. *Water Research,* 47, 7150–7161.

Maurer, M., Herlyn, A. **2006**. Status, costs and investment needs of wastewater removal in Switzerland (Zustand, Kosten und Investitionsbedarf der schweizerischen Abwasserentsorgung). Eawag, Zurich, Switzerland (in German).

Maurer, M., Rothenberger, D., Larsen, T. A., **2006**. Decentralised wastewater treatment technologies from a national perspective: at what cost are they competitive? *Water Science and Technology*, 5(6) 145–154.

Maurer, M., Scheidegger, A., Herlyn, A., **2012**. Quantifying costs and lengths of urban drainage systems with a simple static sewer infrastructure model. *Urban Water Journal*, 1–13.

Möller, B., Lund, H., **2010**. Conversion of individual natural gas to district heating: Geographical studies of supply costs and consequences for the Danish energy system. *Applied Energy*, 87, 1846–1857.

Moss, T., **2001**. Flow management in urban regions. In: Guy, S., Marvin, S., Moss, T. (eds), *Urban infrastructure in transition*. Earthscan, London, UK.

Müller, A., Kramer, D., **2000**. Market economic instruments of sewage disposal (Marktwirtschaftliche Instrumente in der Abwasserentsorgung). Ecoplan. Bern, Switzerland (in German).

Nielsen, S., Möller, B., **2013**. GIS based analysis of future district heating potential in Denmark. *Energy*, 57, 458–468.

OECD, **2006**. Infrastructure to 2030: Telecom, Land Transport, Water and Electricity. OECD Publishing, Paris, France.

OECD, **2007**: Infrastructure to 2030 (Volume 2): Mapping Policy for Electricity, Water and Transport, OECD Publishing, Paris, France.

Ostfeld, A., **2015**. Water Distribution Networks. In: E. Kyriakides and M. Polycarpou (eds.), *Intelligent Monitoring, Control, and Security of Critical Infrastructure Systems, Studies in Computational Intelligence*. Springer-Verlag, Berlin and Heidelberg, Germany.

Parshall, L., Pillai, D., Mohan, S., Sanoh, A., Modi, V., **2009**. National electricity planning in settings with low pre-existing grid coverage: Development of a spatial model and case study of Kenya. *Energy Policy*, 37(6), 2395–2410.

Poustie, M.S, Deletic, A., Brown R.R., Wong, T., J. de Haan, R., Skinner, R., **2014**. Sustainable urban water futures in developing countries: the centralised, decentralised or hybrid dilemma. *Urban Water Journal*, 1–16.

Prim, R.C., **1957**. Shortest connection networks and some generalizations. *Bell System Technical Journal*, 36, 1389–1401.

Pujol, G., **2014**. Package 'sensitivity', Version 1.7. Available from: http://cran.r-project.org/.

Riley, S.J., DeGloria, S.D., Elliot R., **1999**. A terrain ruggedness index that quantifies topographic heterogeneity. I*ntermountain Journal of Sciences*, 5(1-4), 23–27.

Rodrigue, J. P., Comtois, C., Slack, B., **2013**. The geography of transport systems. 3rd ed. Routledge, London, UK.

Saltelli, A., Tarantola, S., Chan, K. P. S., **1999**. A quantitative model-independent method for global sensitivity analysis of model output. *Technometrics,* 41, 39–56.

Sanoh, A., Parshall, L., Sarr, O.F., Kum, S., Modi, V., **2012**. Local and national electricity planning in Senegal: Scenarios and policies. *Energy for Sustainable Development,* 16(1), 13–25.

Sapkota, M., Arora, M., Malano, H., George, B., Nawarathna, B., Sharma, A., Moglia, M., **2013**. Development of a framework to evaluate the hybrid water supply systems. 20th International Congress on Modelling and Simulation, Adelaide, Australia, 1–6 December.

Sapkota, M., Arora, M., Malano, H., Moglia, M., Sharma, A., George, B., Pamminger, F., **2015**. An Overview of Hybrid Water Supply Systems in the Context of Urban Water Management: Challenges and Opportunities. *Water*, 7(1), 153–174.

Sappington, J.M., Longshore, K.M, Thompson, D.B., **2007**. Quantifying landscape ruggedness for animal habitat analysis: A case study using bighorn sheep in the Mojave desert. *The Journal of Wildlife Management,* 17(5), 1419–1426.

Sargent, G.R., **1991**. Simulation model verification and validation. *Proceedings of the 1991 Winter Simulation Conference*.

Schiller, G., **2010**. Cost evaluation of the adaptation of waste water treatment systems under shrinkage (Kostenbewertung der Anpassung zentraler Abwasserentsorgungssysteme bei Bevölkerungsrückgang). IÖR Schriften, Band 51. Rhombos, Berlin, Germany (in German).

Sitzenfrei, R., Möderl, M., Rauch, W., **2013**. Assessing the impact of transitions from centralised to decentralised water solutions on existing infrastructures e Integrated city-scale analysis with VIBe. *Water Research*, 47, 7251–7263.

Sitzenfrei, R., Rauch, W., **2014**. Integrated hydraulic modelling of water supply and urban drainage networks for assessment of decentralized options. *Water Science and Technology*, 70(11), 1817–1824.

Stiller, C., Bünger, U., Møller-Holst, S., Svensson, A.M., Espegren, A., Nowak, M., **2010**. Pathways to a hydrogen fuel infrastructure in Norway. *International Journal of Hydrogen Energy*, 35(7), 2597–2601.

Tchobanoglous, G., Leverenz, H., **2013**. The rationale for decentralization of wastewater infrastructure. In: Larsen, T. A., Udert, K. M., Lienert, J. (eds), *Source Separation and Decentralization for Wastewater Management*. IWA Publishing, London, UK.

UNEP, **2015**, Wastewater characteristics. Section 7.1.1 Wastewater Generation. Accessed: 30.04.2015. Archived by WebCite® http://www.webcitation.org/6YB8j1ifN

Urban Land Institute and Ernst & Young (eds), **2007**. Infrastructure 2007: A Global Perspective. ULI – the Urban Land Institute, Washington, D.C., USA.

Urich, C., Sitzenfrei, R., Möderl, M., Rauch, W., **2010**. An agent-based approach for generating virtual sewer systems, 62(5), 1090–1097.

Urich, C., Bach, P.M., Sitzenfrei, R., Kleidorfer, M., McCarthy, D.T., Deletic, A., Rauch, W., **2013**. Modelling cities and water infrastructure dynamics. *Engineering Sustainability,* 166(ES5), 301–308.

Urich, C., Rauch, W., **2014**. Exploring critical pathways for urban water management to identify robust strategies under deep uncertainties. *Water Research,* 66, 374–389.

VSA, **2011**: Costs and performances of the sewage disposal (Kosten und Leistungen der Abwasserentsorgung). Verband Schweizer Abwasser- und Gewässerschutzfachleute. Glattbrugg, Switzerland (in German).

Weber, B., Cornel, P., Wagner, M., **2007**. Semi-centralised supply and treatment systems for (fast growing) urban areas. *Water Science and Technology*, 55(1-2), 349–356.

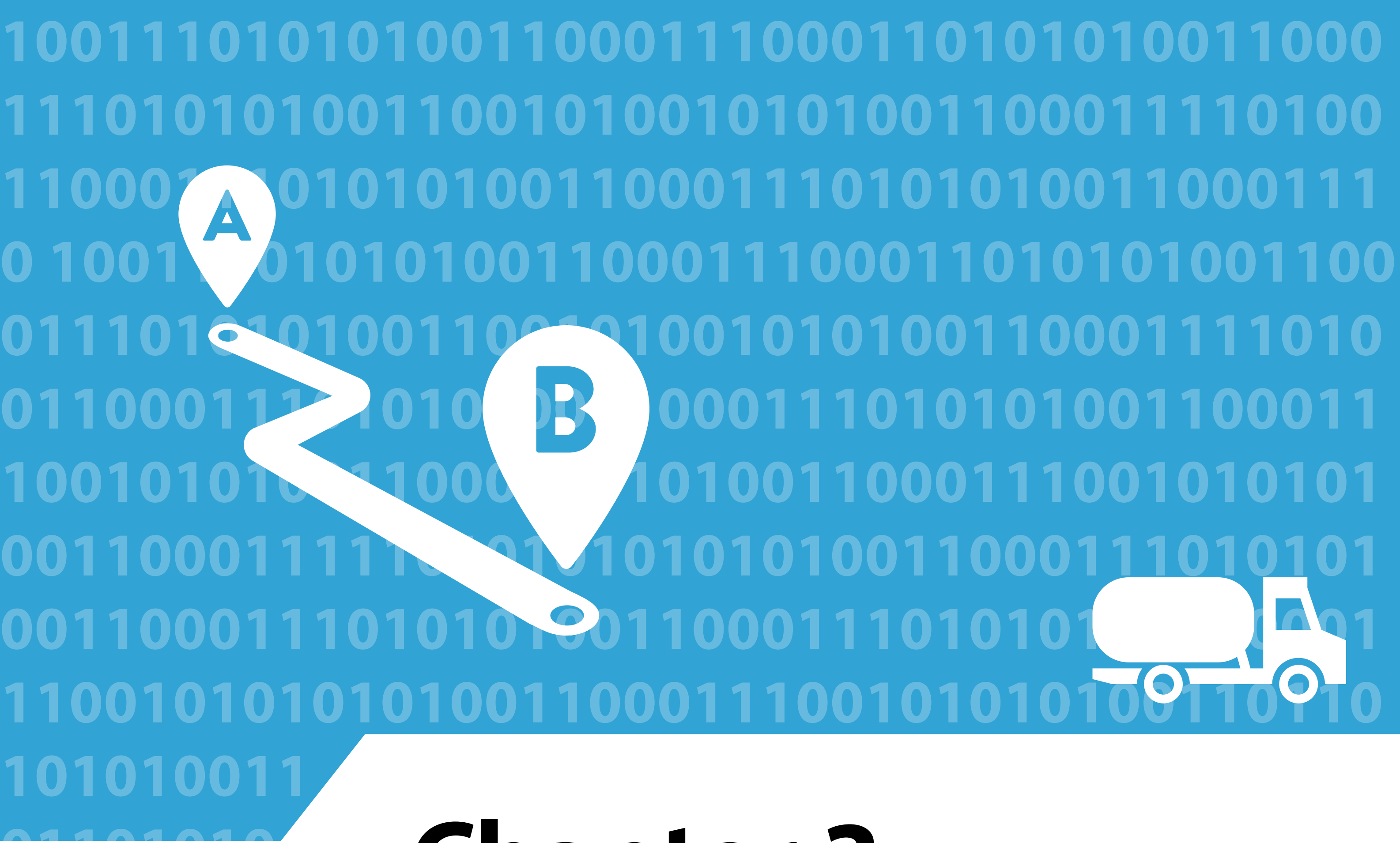
World Bank, **2014**. PPP conversion factor. International Comparison Program database. Available at: http://data.worldbank.org/

Yu, C., Lee, J., Munro-Stasiuk, M. J., **2003**. Extensions to least-cost path algorithms for roadway planning. *Geographical Information Science,* 17 (4), 361–376.

Zahn, C.T., **1971**. Graph-theoretical methods for detecting and describing gestalt clusters. *IEEE Transactions on computers,* C-20(1), 68–86.

Zeferino, J.A., Antunes, A.P., Cunha, M.C., **2010**. Multi-objective model for regional wastewater systems planning. *Civil Engineering and Environmental Systems*, 27(2), 95–106.

Zvoleff, A., Kocaman, A.S., Huh, W.T., Modi, V., **2009**. The impact of geography on energy infrastructure costs. *Energy Policy,* 37(10), 4066–4078.



Chapter 3

Economies of density for on-site waste water treatment

published in ***Water Research****, 101, 476–489.*

Economies of density for on-site waste water treatment

Eggimann Sven[1, 2*], Truffer Bernhard[1,3], Maurer Max[1, 2]

[1] Eawag, Swiss Federal Institute of Aquatic Science and Technology, 8600 Dübendorf, Switzerland.

[2] Institute of Civil, Environmental and Geomatic Engineering, ETH Zürich, 8093 Zurich, Switzerland.

[3] Faculty of Geosciences, Utrecht University, Heidelberglaan 2, NL-3584 CS Utrecht, The Netherlands.

Keywords

on-site treatment costs, sewage sludge logistics, degree of centralization, network externality, economies of scale, decentralized wastewater infrastructure

Abstract

Decentralised wastewater treatment is increasingly gaining interest as a means of responding to sustainability challenges. Cost comparisons are a crucial element of any sustainability assessment. While the cost characteristics of centralised waste water treatment (WMS) have been studied extensively, the economics of decentralised WMS are less understood. A key motivation for studying the costs of decentralised WMS is to compare the cost of centralised and decentralised WMS in order to decide on cost-efficient sanitation solutions. This paper outlines a model designed to assess those costs which depend on the spatial density of decentralised wastewater treatment plants in a region. Density-related costs are mostly linked to operation and maintenance activities which depend on transportation, like sludge removal or the visits of professionals to the plants for control, servicing or repairs. We first specify a modelled cost-density relationship for a region in a geometric two-dimensional space by means of heuristic routing algorithms that consider time and load-capacity restrictions. The generic model is then applied to a Swiss case study for which we specify a broad range of modelling parameters. As a result, we identify a 'hockey-stick'-shaped cost curve that is characterised by strong cost reductions at high density values which level out at around 1 to 1.5 plants per km^2. Variations in the cost curves are mostly due to differences in management approaches (scheduled or unscheduled emptying). In addition to the well-known diseconomies of scale in the case of centralised sanitation, we find a similar generic cost behaviour for decentralised sanitation due to economies of density. Low densities in sparsely populated regions thus result in higher costs for both centralised and decentralised system. Policy implications are that efforts to introduce decentralised options in a region should consider the low-density/high-cost problem when comparing centralised and decentralised options.

3.1 Introduction

3.1.1 Comparing central and decentral sanitation costs

Costs are an integral criterion for decisions on suitable wastewater management systems (WMS) for both centralised and decentralised scenarios (inter alia Hamilton et al. 2004, Maurer et al. 2006, Libralato et al. 2012, Truffer et al. 2013). Decentralised WMS are increasingly considered as potential substitutes for centralised WMS with sewer networks (inter alia Tchobanoglous et al. 2004, Massoud et al. 2009, Larsen et al. 2013, OECD 2015). Typically, decentralised WMS – also called on-site (OST) – treat small wastewater flows in individual residences or residential clusters (cf. Tchobanoglous and Leverenz 2013), which can, as a consequence, save on extensive sewer networks (Libralato et al. 2012). However, it is a complex task to determine the optimal degree of centralisation in water and wastewater management (Eggimann et al. 2015, Poustie et al. 2014, Adams et al. 1972, Guo and Englehardt 2015, Lee et al. 2013) because the overall costs in a region depend not only on the sum of the costs of all individual technological components but also on how they are spatially distributed. This implies that besides the usual cost-driving factors like context uncertainties, economies of scope, economies of scale or high network infrastructure life-spans (Hansman et al. 2006, Markard 2009, Starkl et al. 2012), space-dependent cost items such as economies of density and network externalities have to be taken into account.

In the case of centralised WMS, space-dependent cost effects play out in the form of major economies of scale at the level of the wastewater treatment plant (i.e. per capita costs decrease with the number of people in a catchment connected to it), whereas the costs of building up a sewer system show diseconomies of scale (i.e. to reach full connection more distant settlements need to be connected). These cost characteristics have been intensively discussed in the literature (cf. Townend 1959, Downing 1969, Adams et al. 1972, Haug 2004, Friedler and Pisanty 2006, Maurer et al. 2006/2010). Nevertheless, the cost characteristics of OST systems are much less well known. In general, unit prices of OST plants do not depend on the number of units installed in a specific region. However, management, maintenance and regulation schemes may turn out to be very costly, because travel costs for service teams may become important (inter alia Kennedy-Walker et al. 2014, Semiyaga et al. 2015, Hamilton et al. 2004, Kaminsky and Javernick-Will 2013). An integrated assessment of these different cost components for determining the optimal degree of centralisation in a region is however lacking (Hamilton et al. 2004, OECD 2015, Eggimann et al. 2015). The optimal degree of centralisation is directly linked to the OST plant density, as this increases in response to growing population percentages serviced by on-site treatment plants. In this paper we examine an essential parts of such an integrated cost assessment, which are arguably the least well understood, namely those that are related to spatial density of OST plants. We present a model-based approach to examining the economies of density[1] of OST plants and conduct a sensitivity analysis of different management approaches. A model-based approach is needed because cost-data collection is challenging and there is a lack of available data to carry out a systematic comparison of the costs of different WMS in a region.

3.1.2 How space and transportation influence costs

In the field of spatial economics, the important influence of spatial dispersion on service provision has long been postulated (Wegener 2011): many different theoretical models based on transportation-cost considerations have been developed, such as von Thünen's (1875) ring model, Christallers' (1933) model of optimal provision or the optimal city-size model of Arnott (1979). Such studies highlight the fact that the transportation of material or personnel are critical for efficient service provision. Much research has consequently evolved around space-dependent cost efficiencies in many different infrastructure fields,[2] including the water and wastewater sector (cf. Guerrini et al. 2013, Álvarez et al. 2014). The finding that the operation and maintenance (O&M) costs of point-type infrastructures are particularly dependent on the settlement or population density is especially interesting with respect to OST systems (inter alia Schiller and Siedentop 2005, Wenban-Smith 2009). As a consequence, we expect the haulage distance to be crucial for assessing the O&M costs of OST systems (Semiyaga et al. 2015). Despite this long-known influence, the spatial cost effects concerning the O&M of OST plants have not been systematically estimated. Furthermore, the literature often focuses on single cost aspects of decentralised wastewater O&M such as monitoring (inter alia Hug and Maurer 2012) or sludge transportation (inter alia Steiner et al. 2002). Nevertheless, there are some notable exceptions explicitly focusing on the road-based transportation needed in the case of OST plants: Steiner et al. (2002) propose a simple method for estimating the haulage costs on the basis of geometrical and economic criteria, and have used

[1] González-Gómez and García-Rubio (2008) differentiate between economies of product density and economies of customer density. The former denotes the marginal cost savings of a fixed number of consumers due to increased consumption. The latter refers to the cost savings achieved by the higher efficiency resulting from a larger number of consumers. We focus on economies of customer density, implying that the marginal costs of providing services decrease with an increasing number of customers in a spatially defined area. We refer to Holmes (2011) for an overview of the literature focusing on economies of density in other thematic fields.

[2] Typically, examples can be found in solid waste management (inter alia Zamorano et al. 2009, Tavares et al. 2009, Ghose et al. 2006). See Section 3.4.3 for further applications.

it to find decreasing costs with higher population densities. Flotats et al. (2009) show that minimising transportation costs is vital for manure management, a factor that is highly relevant to wastewater transportation in OST plants. The authors compare on-farm and centralised treatments and conclude that transportation costs are crucial for deciding between centralised and decentralised strategies. Marufuzzaman et al. (2015) present a method to compare pipeline and truck-based transportation of wastewater sludge and perform a cost analysis based on transported volumes and distances. Whereas different treatment options might result in different operating and maintenance requirements, Etnier et al. (2000) note that cost differences can be expected to result from the different strategies of collecting and maintaining WMS.

We believe the paucity of literature about O&M for OST systems to be responsible for rather speculative and vague overall cost claims (Hamilton et al. 2004, Dodane et al. 2012, Singh et al. 2015, Hendrickson et al. 2015, Truffer et al. 2013, Etnier et al. 2000). As a result, many authors conceive O&M of OST systems as costly, which adds to the conventional wisdom that decentralised WMS are challenging to operate and manage (inter alia Bakir 2001, Parkinson and Tayler 2003, Maurer et al. 2006, Buchanan et al. 2014). The methodological framework introduced in this paper enables the systematic assessment of cost effects relating to OST plant density by examining the most important space-related costs (residual transportation, service and repair costs), and in doing so prepares the ground for an integrated assessment of the optimal degree of centralisation in the provision of regional wastewater infrastructure. It is not the aim of this paper to perform a comprehensive overall cost analysis.

3.2 Materials and methods

We first identify those cost items which depend on the spatial density of plants in a region and differentiate between two management approaches for sludge emptying at OST plants. We then give a general methodological overview and explain the routing algorithms in detail. Section 3.2.5 presents the distance parameter estimation, followed by information on cost parameters and a sensitivity analysis. Section 3.2.8 introduces the case study.

3.2.1 Tasks sensitive to economies of density

We do not intend to perform a full cost comparison of OST systems or a complete analysis of O&M costs, but only aim to identify space-related costs. Therefore we do not consider investment or capital costs or all fixed costs, and particularly not costs independent of space. By the same logic, we also treat variable costs which depend on the chosen OST system or specific external conditions being constant, such as sludge treatment, energy consumption, chemical acquisition or other expenses such as taxes (see i.e. Fletcher et al. 2007, WERF 2015). Such costs can simply be added as fixed baselines to the costs calculated in this paper, depending on the chosen technological solution. Further items such as regulatory costs may also be included in this broad conceptualisation. However, we maintain that these items follow the same logic and could therefore be easily added to an overall cost assessment.

We consider three typical tasks that exhibit cost characteristics which are space-dependent; namely, i.) residual (sludge and scum) emptying, ii.) service and iii.) repairs, as detailed below. Specific task execution may differ depending on the technical details of the chosen OST plant. As we outline below for each specific task, depending on the decentralised WMS, less service and repairs may be needed or the amount of sludge and scum may differ[3] (we exemplarily refer to Singh et al. (2015) and Crites and Tchobanoglous (1998) for various technology options). We assume that these tasks are carried out by specialised external contractors (operator model) who have to travel to the treatment plants, as proposed for example by Massoud et al. (2009).

i.) *Residual emptying*: Wastewater treatment produces sludge and scum which needs to be disposed of within certain time intervals. We assume that this disposal is performed by a specialised contractor. The accumulated volume per population equivalent (PE) depends on the given technical system and the sludge residence time. The haulage of these residuals is context-dependent (Mikhael et al. 2014) but is commonly road-based and is typically carried out by a suction truck with a specific load capacity and an average travelling speed for collection. The process of emptying is time-consuming, as the treatment plants need to be accessed, the sludge pumped, and further tasks such as filling out paperwork completed.

ii.) *Service*: The long-term reliability of OST systems depends on maintenance and reporting (Bradley et al. 2002). In order to perform maintenance work (such as membrane regeneration or similar) or simply

[3] The same is true for the amount of time needed for system maintenance, which might differ considerably (e.g. the maintenance time of a reed-bed treatment plant system differs from that of a sequencing batch reactor). However, these three tasks must be performed in some way, irrespectively of the choice of OST system. For this paper, the chosen parameter values apply especially to membrane bio-reactors (MBR) or sequencing batch reactors (SBR). However, the previously outlined operationalisation of space-dependent costs can be specifically adapted to other systems such as septic tanks or reed-bed based systems.

to check functionality, OST systems typically need to be visited by a competent service contractor. How often this has to be done depends on the complexity of the OST system, the sensor technology equipment and the required level of monitoring (ibid.). Depending on the management model, legal situation and treatment system, the number of visits or the type of tasks to be performed differ. We make the simplifying assumption that the service task is independent of the residual emptying and performed within certain time intervals by a mobile technician spending an average amount of service time per OST system plus travelling time.

iii.) *Repairs*: If an OST system fails, a technician needs to visit the plant and perform specific repairs. We assume that enough funds are available for these repairs and that failing systems require mandatory repair. How often the system fails depends on its type and the quality of its service and control. In reliability engineering, the frequency of failure is commonly expressed as a failure rate (Finkelstein 2008). We consider the repair tasks to be similar to service tasks by assuming that a technician has to visit the plant in a vehicle and needs an average repair time to do the job. Given the scope of this paper, we do not relate service frequency with failure rates.

3.2.2 Differentiating management approaches

Efficiency and cost-effectiveness are major goals of logistics management. Therefore it is important to consider different management approaches in order to assess their impact. In this section, we present two ideal types of management for residual emptying which represent a worst (unscheduled) and best-case (scheduled) emptying approach.

- *Scheduled*: For scheduled emptying, we assume that the plants are evacuated periodically and an optimal routing plan can be set up. We make the simplifying assumption that all OST plants are full at the time of emptying. This assumption allows each OST plant to be visited along an optimal collection tour and the truck's storage capacity to be exploited to the maximum at all times. We consider this emptying approach to be the most efficient one from a logistics point of view, yielding the best theoretical solution.
- *Unscheduled*: For unscheduled emptying, the collection tour is determined on the basis of whether the OST plant emptying is needed due to critical tank filling. We assume that the plant owners will call the operator (or the plant will send a signal) when the tank capacity limits are reached. This results in different daily collection tours depending on the number of people calling and their geographical position.

In real world situations, tank filling rates will in general not be constant. As a consequence, some tanks will have to be evacuated earlier than expected or would not be full at the time of emptying. Differences in filling rates may be due to variations in the number of users per plant, particularly over time. Examples with highly varying rates would include tourist regions with many part-time residences. The two ideal approaches of scheduled and unscheduled emptying therefore represent best and worst-case scenarios. More realistic situations can be considered as lying somewhere between these two extremes.

3.2.3 Modelling setup

The proposed modelling procedure derives a cost-curve for different treatment plant densities. The basic idea is to assess the costs of the tasks outlined in Section 3.2.1 for a specific number of OST plants with a given treatment volume. For modelling purposes, we identify density values by calculating the space-dependent costs for a sequence of circular catchment areas. Different densities result from an incremental decrease of the catchment area diameter (see Fig. 3.1).[4] For each catchment, we randomly

[4] A more intuitive operationalization of densities would have been to increase the number of OST plants in a given region. Instead we chose a fixed number of OST plants in a decreasing set of smaller areas. The main reason for this is that we wanted to correct for influences of specific geographical clustering, while still

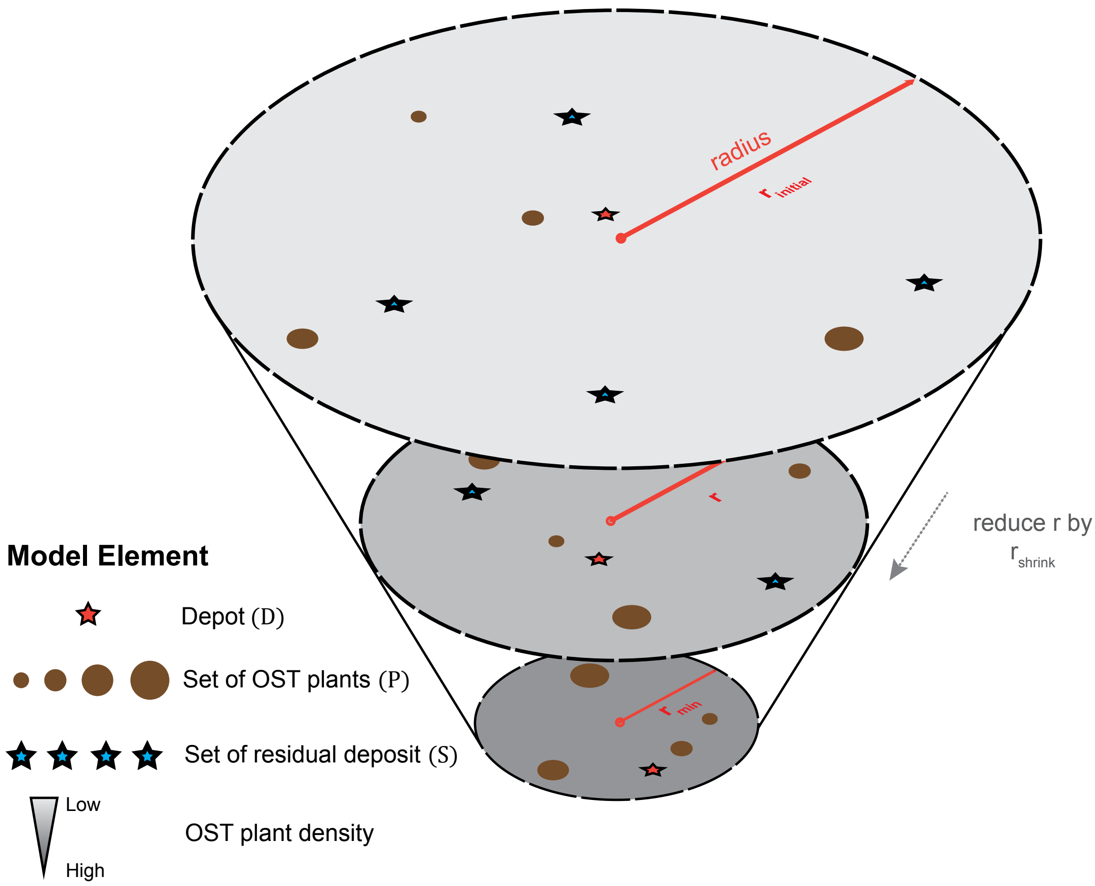


Figure 3.1: *Visualisation of the schematic modelling approach.*

distribute a fixed number n of OST plants, and a relative number m of residual deposits (places where the sludge can be discharged).[5] Additionally, one depot (the operating basis for the emptying vehicle and its driver) is positioned near the most central residual deposit. The following cost calculation steps are executed for each catchment area (cf. Fig. 3.2 for more

being able to run meaningful routing procedures. Holding the number of OST plants constant has the advantage that different density values remain largely comparable. Geographical variations will be taken into consideration by the values of parameter f_d.

5 We assume that the sludge can be treated at large conventional treatment plants, which allows us to easily calculate $\rho_{deposit}$ based on the number of today's existing plants of this kind. Depending on the specific application case, the rationale for defining the deposit density may be different (for example whether we assume that additional special sludge treatment units are constructed or that all residuals are transported to existing treatment plants).

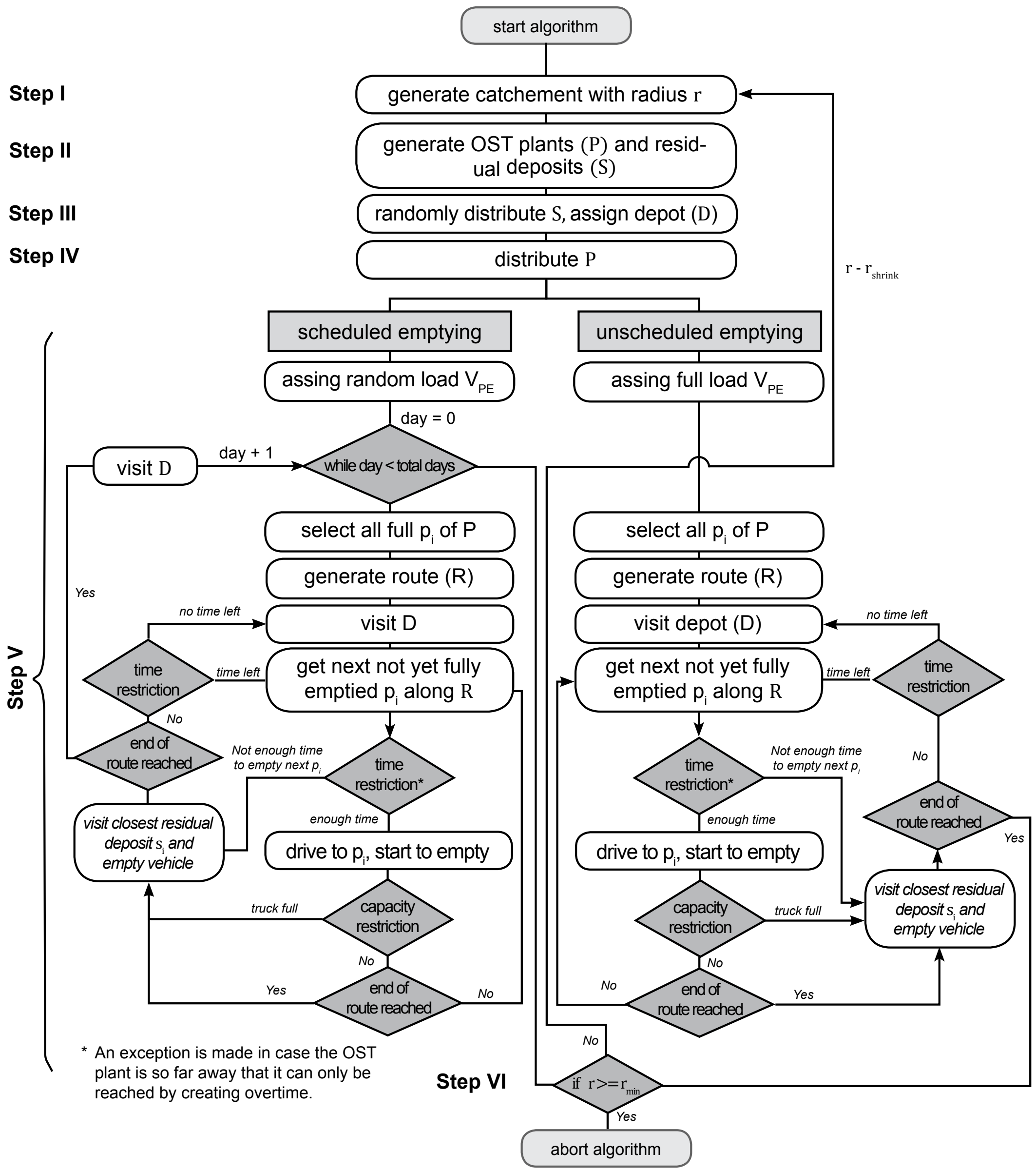


Figure 3.2: *ULM diagram of the methodological approach.*

details and Table 3.1 for a overview of all model parameters):

Step I: In each iteration step, the catchment area of the system is defined as a circle of radius r, which we denote in the first iteration as $r_{initial}$. We define the initial radius as the maximum distance a vehicle can reach within one working day (Eq. 3.1):

$$r_{initial} = \frac{v_{truck} * t_{drive}}{2} \quad (3.1)$$

where: v_{truck} = average vehicle speed and t_{drive} = maximum possible driving time per day. The rationale for choosing this maximum radius is that even the most distant OST plants can be reached from the depot within one working day.[6]

Step II: We generate a set of OST plants $P = \{p_1, p_2, \ldots, p_n\}$ consisting of a fixed number (n) of individual plants (p_i). To each of these individual plants we attribute a load (within a given range) given in population equivalents V_i^{PE}. On the basis of this load V_i^{PE} and the sludge accumulation rate (r_{acc}), we then calculate the total sludge and scum accumulation for all p_i. In the case of scheduled emptying we attribute the maximum possible load V_i^{PE} to each plant. In the unscheduled mode, V_i^{PE} are set at a random level.

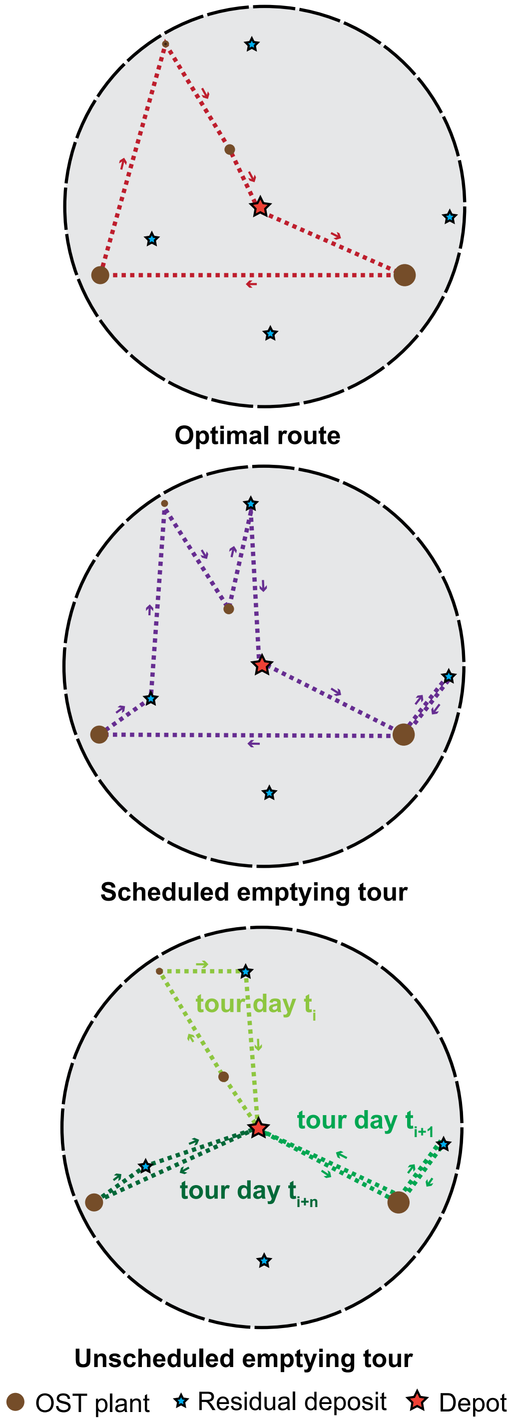


Figure 3.3: *Schematic example of different tours depending on the chosen emptying approach with shown deviations from the optimal tour due to time and capacity restrictions.*

Additionally, we create a set $S = \{s_1, s_2, \ldots, s_m\}$ of individual residual deposits (s_j). The number of deposits m is calculated on the basis of the

[6] Because of distance weighting, travelling times might be longer than the daily working time. Including the time needed to empty the sludge at the disposal point might therefore result in daily overtime.

Table 3.1: *Model parameter overview.*

	Symbol	Unit	Parameter Description	Base Value	Source	Selection		Parameter for Sensitivity Analysis
						First	Second	
Deposit	$\rho_{deposit}$	deposits*(km²)⁻¹	Residual deposit density	0.015	AWA and UWE (2010)	•	•	Base, 0.0075, 0.00325
OST plant	V^{PE}	PE	Range of PE for stochastic OST plant dimensioning	4-20		•	•	4 – 10, base, 4 – 30
	r_{acc}	$m3*PE^{-1}*y^{-1}$	Faecal sludge and scum accumulation rate	0.25	Franceys et al. (1992)	•	•	0.1, base, 0.4
	$p_{failure}$	$\%*year^{-1}$	Linear OST failure probability rate	10	EPRI (2000)	•		
	$t_{emptyOST}$	$h*OST^{-1}$	Average time used for emptying an OST plant	0.33		•		0.166, base, 0.50
	$t_{service}$	h	Average time to service an OST plant	0.5		•		
	t_{repair}	h	Average time to repair an OST plant	1.0		•		
	n_{visits}	$visits*year^{-1}$	Number of times an OST plant gets visited per year	1.0	AWA (2014)			
	n_{OST}		Number of OST plants in set P	500		•	•	100, base, 900
	p_{emp}	%	Probability that an OST plant can be served	0.99		•	•	0.97, 0.98. 0.99
Vehicle (truck)	v_{truck}	$km*h^{-1}$	Average driving speed of suction vehicle	50	Zamorano et al. (2009)	•	•	20, base, 80
	cl_{truck}	m^3	Load capacity	14	Amphitec (2015)	•	•	10, base, 18
	$ct_{ruckrent}$	$\$*h^{-1}$	Rental cost	70	Amphitec (2015)			
	$c_{truckfix}$	$\$*km^{-1}$	Fixed costs	0.45	See Section 3.2.6			0.166, base, 0.50
	$t_{emptyTruck}$	$h*event^{-1}$	Average time used for vehicle unloading	0.33		•	•	

Table 3.1: *(continued).*

	Symbol	Unit	Parameter Description	Base Value	Source	Selection		Parameter for Sensitivity Analysis
						First	Second	
Vehicle (car)	v_{car}	$km*h^{-1}$	Average speed	50		•		
	$c_{carrent}$	$\$*h^{-1}$	Rental costs	2.3	Mobility (2015)			
	c_{carfix}	$\$ *km^{-1}$	Fixed costs	0.36	Mobility (2015)			
Personnel	t_{pmax}	$h*day^{-1}$	Maximum daily working hours for drivers	8.0	BFS (2013)	•	•	7, base, 9
	c_p	$\$*month^{-1}$	Monthly personnel payroll	5366	BFS (2015)			
Other	f_d	-	Ratio between road distance and straight-line distance		See Section 3.2.5	•	•	-10%, base, +10%
	f_{NN}	-	Average nearest neighbour clustering ratio	1		•		
	r	km	Radius to generate catchments		See Section 3.2.3			
	Δr	km	By how much the catchment is decreased in each iteration	10				
	r_{min}	km	Minimum radius to generate catchments	10				
	y	-	Number of years	5				
	t_{drive}	h	Maximum daily driving time assuming a minimum of one OST plan scheduled daily $t_{pmax} - t_{emptyTruck} - t_{emptyOST}$		See Section 3.2.3			

deposit density factor ($\rho_{deposit}$) for each radius. This way we assume that the total number of deposits in a region is set largely constant despite a density increase.

Step III: We randomly distribute all s_j within the catchment area and assign a single depot to the s_j closest to the centre of this area.

Step IV: We randomly rearrange all p_i until we reach a Nearest Neighbour Index (Clarke et al. 1964) of $f_{NN} = 1$ in order to ensure the same degree of clustering for all r. This allows a like-for-like comparison, independent of the catchment radii, and ensures that the cost differences found do not result from different degrees of clustering. An alternative to a random distribution of OST plants within the catchments would be a selection of possible OST plant sites with the aid of real settlement structures. However, the formulation of detailed (spatial) technological transition models is complex (cf. Zeppini et al. 2014) and not appropriate given the generality of the modelling approach.

Step V: We calculate the costs for a contractor travelling to the plant by means of a routing algorithm (described in Section 3.2.4). This provides us with a point estimate for the respective density measure.

Step VI: We decrease the catchment radius r by Δr, keeping $r \geq r_{min}$ and loop back to Step II. Δr and r_{min} are technical parameters for determining the number of iterations in the algorithm. Choosing smaller values for Δr simply increases the number of iterations, while r_{min} denotes the maximum density beyond which we no longer observe any significant cost reductions.

3.2.4 Routing algorithms

In this section, we explain our cost calculations for the logistics of OST plant O&M services in detail for the case of residual emptying (cf. Fig. 3.2, Step V). We apply different routing algorithms depending on the chosen management approach (Section 3.2.3). A schematic example of scheduled and unscheduled emptying approaches is visualised in Fig. 3.3.

We generate collection tours for both emptying modes with the aid of route optimisation techniques. We use algorithms based on heuristic routing (Cormen et al. 2009) with the aim of finding a minimum path between a given set of destinations. This is commonly referred to as the Vehicle Routing Problem, which adds capacity constraints to the common Travelling Salesman Problem (Lawler et al. 1985). We refer to Gendreau et al. (1996) for an overview of the numerous approaches to addressing the Vehicle Routing Problem. We use heuristics in order to avoid a heavy computational burden and choose the classical Nearest Neighbour Algorithm because of its intuitiveness (Johnson and Papadimitriou 1985). Heuristically determined solutions rarely constitute an optimum: however, they give good approximations in many applications with a reasonable computational burden (Michaelewicz and Fogel 2004). For validation purposes, we compare the applied Nearest Neighbour algorithm with a computationally more expensive algorithm developed by Clarke and Wright (1964).

The various tasks such as emptying an OST plant ($t_{emptyOST}$), travelling to the depot or to an OST plant and emptying a suction truck ($t_{emptyTruck}$) require a time t. We determine the required travel time on the basis of the distances travelled and average travelling speed (v_{truck}) and choose reasonable value ranges for the remaining time parameters. We then use cost parameters (c_p, $c_{truckrent}$, $c_{truckfix}$) to convert the time taken or distances travelled into total costs.

The routing algorithms for the different tasks are based on the following logic:

- *Scheduled emptying tour*: We assume that the collector can set up an optimal route for the entire catchment, so all OST plants are considered for optimal tour calculations and are always filled to a maximum V_i^{PE} (compare Fig. 3.2). The tour starts at the depot and continues along the optimal route until all OST plants are emptied.[7] For the case of residual emptying, we consider two restrictions which influence the theoretical optimal route, namely the maximum vehicle load capacity (cl_{truck}) and the maximum number of working hours per day (t_{pmax}).These restrictions primarily influence the degree to which the suction truck deviates from its optimal route. For every OST plant visit, only as much sludge is emptied as fits the free capacity of the suction truck. If the maximum load is reached, the truck visits the closest residual deposit in order to dispose of its load. If some daily working time is left to continue the tour, the truck resumes its emptying work, otherwise it returns to the depot for the night. If the time restriction is reached even though the truck is not completely filled, it first visits the closest residual deposit and then returns to the depot in order to start a new working day. Cost calculations for scheduled emptying are based on a single tour where all OST plants are visited only once (see Fig. 3.2).

[7] We do not differentiate between the directions of the tour and randomly decide between a clockwise or anticlockwise direction starting from the depot.

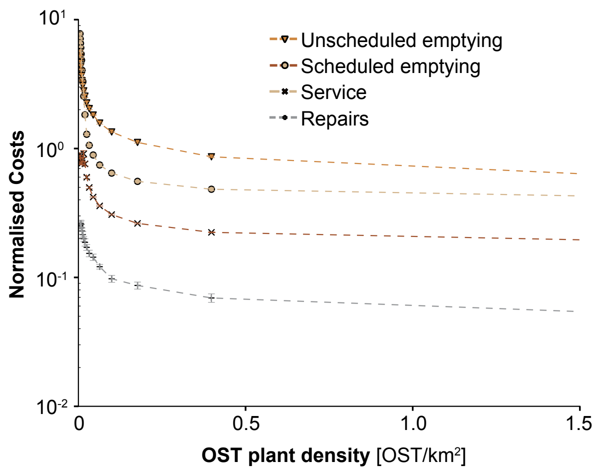


Figure 3.4: *Normalised costs (1 = $6.2 PE-1 yr-1) for the scheduled and unscheduled emptying approaches, and for the service and repair tasks (base parameters, n = 10). The error bars show the uncertainty resulting from different OST plant placement in the catchments.*

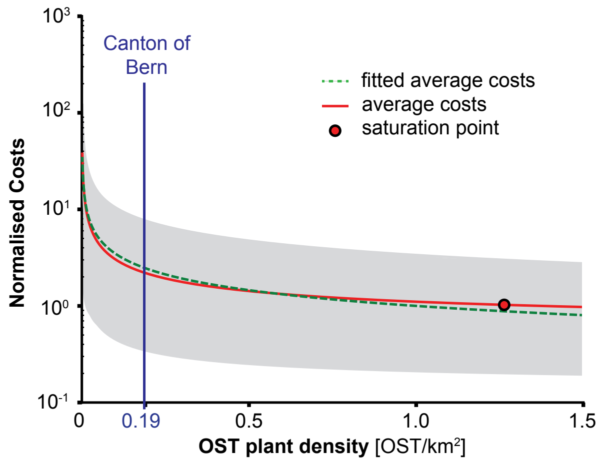


Figure 3.5: *The average normalised cost (NC) curve (1 = $5.6 PE^{-1} yr^{-1}) is indicated in red: it can be approximated with NC = $x^{-0.90}$ (n=100). The grey area shows the result range. Today's average OST plant density for the whole case study area is indicated with the blue line.*

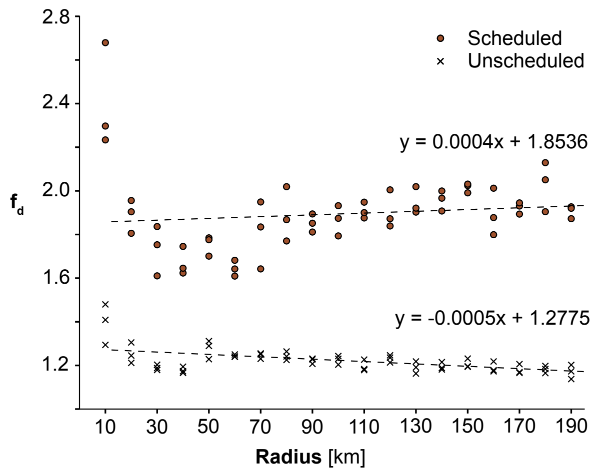


Figure 3.6: *Distance factor estimation for the scheduled and unscheduled emptying approach.*

We further assume that within a certain probability (p_{emp}) an OST plant may be not emptied due to unexpected events (such as the absence of the house owner making the property inaccessible). This results in distances travelled and time spent without task accomplishment and leads to tour deviations, since OST plants need to be visited again in a later tour.

- *Unscheduled emptying tour*: In the case of unscheduled emptying we initially assign a random tank filling level to each OST plant. Then we increase the level iteratively by a constant daily sludge and sum accumulation rate (r_{acc}) over a number of years (y). Finally, we calculate collection tours for each day based on the OST plants filled on each respective day. The procedure of collection and transport for the individual daily tours is analogous to the scheduled tour calculation. However, itis repeated for unscheduled emptying over several years (y) in order to calculate average costs.[8]
- *Service tour*: The algorithm for the service tour is very similar to that for scheduled emptying. The sole methodological difference is that we only consider the restriction on maximum working hours and use different values for the corresponding parameters (v_{car}, $c_{carrent}$, c_{carfix}, n_{visits}, $t_{service}$).
- *Repair tour*: The repair task tour calculation is similar to that for unscheduled emptying. We set up the tours daily only on the basis of the failed OST plants. In order to calculate the average repair costs per year, we also perform an iteration over a number of years (y). As only limited knowledge is available about the failure rates of OST plants, we use linear failure rates ($p_{failure}$) taken from the literature (EPRI 2000). The only restriction on the repair task is the maximum number of working hours per day for the tour generation. Corresponding time and cost parameters are also used (t_{repair}).

3.2.5 Distance factor estimation

The model distance calculations are based on straight-line distances. In order to derive more realistic distances, we introduce a weighting factor (f_d). This f_d enables the model to be adapted to different geographies where distance calculations differ because of topographical

[8] Most management approaches probably lie somewhere between the scheduled and unscheduled emptying tours, as we often find intermediate approaches where for example certain geographical regions are scheduled to be served within a certain time window and OST plant owners need to call if they want to have their OST serviced or emptied within this time frame. Furthermore, due to discontinuous OST plant filling rates, it may not be feasible to set up a purely scheduled approach.

characteristics and the existing road network (e.g. objects like lakes or mountains which require detours). We use different weighting factors for scheduled and unscheduled emptying and estimate the distance factor for different distance radii. For the case study (see Section 3.2.8) we estimate f_d for scheduled and unscheduled emptying with the aid of the Dijkstra (1959) algorithm by comparing the road-based and straight-line distances for all calculations on the basis of the actual road network (Figs. 3.6 and A.3.1).

3.2.6 Cost parameters

The estimation of transportation costs is central for the proposed modelling approach. Fuel consumption, wear, repairs, insurances, payload, truckload, routine maintenance and depreciation are typically considered in estimating transportation costs (inter alia Barnes and Langworthy 2003, Cambridge Systematics 1995). For this study, we calculate fixed transportation costs per km by assuming constant fuel consumption independently of the load ($c_{truckfix}$). We further assume that our vehicles are rented in order not to incorporate idle capacities in our study. This means that all costs relating to maintenance, repair or insurance of the vehicle etc. are included in the rental costs ($c_{truckrent}$). All costs are given per capita and year and we convert local currencies to US$ using purchase power parities for the year 2013 (World Bank 2014).

Table 3.2: *Comparison of the Clarke and Wright (CW) algorithm with the Nearest Neighbour (NN) algorithm for sludge emptying (n = 10).*

	Scheduled Emptying			Unscheduled Emptying		
r	CW [$/PE/year]	NN [$/PE/year]	[%]	CW [$/PE/year]	NN [$/PE/year]	[%]
10	3.3	3.3	0	4.9	4.8	1
20	3.9	3.7	4.3	6.9	6.7	3.4
30	4.4	4.3	3	8.8	8.6	1.5
40	5	5	0.8	11	10.4	1.4
50	5.9	5.8	1.4	12.4	12.2	1.6
60	7	6.9	1.7	14.2	14.1	0.8
70	8.4	8.2	2	16	15.8	0.7
80	10.4	10	3.6	18.2	17.5	3.8
90	14.7	14.2	3.2	20	19.7	1.2
100	20.5	19.7	4.2	22.2	21.7	2.2
110	27	25.7	5	24.2	23.9	1.3
120	32.9	31.3	5.2	26.4	26	1.7
130	37.5	37.5	0	28.2	28.5	-1.1
140	43	41.9	2.6	31	30.6	1
150	48.6	46.8	3.8	34.2	33.4	2.1
160	52.9	50.6	4.6	37.1	36	3
170	57	55	3.6	40.4	39.8	1.4
180	62.7	60.2	4.1	44.5	43.5	2.3

3.2.7 Sensitivity analysis

We assess the model sensitivity by a fixed sampling-based approach (Saltelli et al. 2004). We assess the model sensitivity by determining all model parameters resulting in possible routing changes and thus cost changes in a first step (first selection in Table 3.1). Parameters influencing only the absolute costs for all modelled catchments are merely of secondary interest, as they only increase or decrease costs proportionally over all OST plant densities (for example, by changing the fixed travel costs per km).These parameters need to be adapted to the case study. In a second step, we select all parameters relating to residual management, as this is the most cost-intensive task (compare Fig. 3.5). This leaves us with eleven parameters for which we define reasonable value ranges and use the extreme and base values to create a set of parameter configurations (see Table 3.1). From this set, we randomly select and calculate fifty parameter configurations for each emptying approach. This results in a range of different cost curves (Fig. 3.5) representing the sensitivity of the spatial dependence of the costs.

3.2.8 Case study Canton of Bern (Switzerland)

We apply our model to an administrative area in Switzerland (Canton of Bern) (Fig. A.3.1). This area is located in western Switzerland and covers ~6000 km^2, contains 65 large (>1000 PE) wastewater treatment plants and roughly 1000 OST systems (AWA and UWE 2010). We therefore calculate an overall density of ~0.19 OST plants per km^2 with locally varying densities (Fig. A.3.2).[9] We chose this case study because this is an area where the provision of centralised wastewater services has already reached its limits today. This is confirmed by the higher OST plant density compared to the whole of Switzerland. In the case study area, each community is responsible for the operation and maintenance of OST plants and the regulatory agency is responsible for ensuring that the laws are observed. For instance, local regulations require that OST plants are emptied at least once a year (AWA 2014).

The calculation steps outlined in Section 3.2.8 are applied to the case study area, with the city of Bern being close to the catchment's centre (Fig. A.3.1). We calculate truck-based transportation costs for the case study on the basis of parameters collected by the Swiss Federal Statistical Office (BFS 2009, BFS 2013, BFS 2015). For the suction truck, we assume average fuel consumption of 33 litres per 100 km at a diesel price of $1.34/litre for the year 2014 (BFS 2015b). The average speed is set at 50 km per hour, which corresponds to the official speed limit within settlement areas and is similar to average speed assumptions

[9] Because there are significant areas covered with glaciers or without vegetation on the borders of the case study area, we subtract these for the density calculation (see Appendix A).

in other logistics studies (inter alia Zamorano et al. 2009). The accumulation of sludge per year typically depends on the residence time in the OST plants, and we refer here to the literature for the accumulation rates (inter alia Franceys et al. 1992). We did not perform a detailed time analysis assessment (e.g. with method-time measurements) but chose a time interval for each time parameter on the basis of our own expertise. The chosen standard parameter values can be used as a starting point for further detailed analysis (cf. Table 3.1). We use typical transportation costs per km and Swiss hourly rental costs for a minivan used to perform repair and service tasks (Mobility 2015), and calculate hourly rental costs for the suction truck with the aid of cost data from a rental company for these trucks (Amphitec 2015).

3.3 Results

We normalise all costs in order to highlight the cost relationship and not the absolute cost level, which depends on cost- and technology assumptions specific to the case study.[10]

3.3.1 Costs of space-sensitive tasks

Fig. 3.4 shows that costs decrease exponentially with increasing plant density and that emptying costs are the highest for the defined tasks. The shape of this density-cost relationship can be described as a 'hockey-stick'. As expected, scheduled emptying is more efficient than unscheduled emptying, especially for low OST plant densities. For lower densities, the scheduled and unscheduled curves approximate as more time is spent driving to the depot, making less time available for possible route optimisation by visiting multiple OST along the same route.

3.3.2 Model sensitivity

Fig. 3.5 shows the average cost curve of the different input parameter sets with respective sensitivity bands. We generally find the highest model sensitivity in the range where the cost-density relationship starts to level out (~0.1 – 0.5 OST/km^2). Comparing the current average OST density for the whole case study area, we notice that we are close to the range where costs start to level out (see blue line in Fig. 3.5), even though regional differences exist (cf. Fig. A.3.2).

3.3.3 Routing algorithm comparison

The calculated cost differences arising from selecting either the Clarke and Wright algorithm or the Nearest Neighbour algorithm are very minor. On average, the differences between these two algorithms are less than 3% (see Table 3.2).

[10] Results with absolute costs are given in Fig. A.3.3.

3.3.4 Distance factor f_d

Fig. 3.6 shows the results of the estimation of the distance factor for the case study area (compare Section 3.2.5). The street network (shown in Fig. A.3.2) needed for this parameter estimation tends to show slight star-like behaviour with more streets oriented towards the capital city in the centre of the catchment. Street density is also higher for central areas and those with higher population densities. Two things stand out from this parameter estimation: first we see a significant difference between the two emptying approaches and note that the straight-line distance approximation is more realistic for the case of unscheduled emptying; second, the factors are relatively constant for larger areas and we have more variation for smaller areas.

We explain the differences between the cost factors by the geography of the case study. There is a tendency for the centre of the catchment (where the depot is located) to have better straight road connections to reach its periphery. Scheduled emptying requires vehicles to travel across more single-tour segments in parallel to the catchment centre, resulting in longer distances travelled.

3.4 Discussion

3.4.1 Economies of density and emptying approaches

There exists a technological variety of different decentralised WMS (cf. Tchonablous et al. 2004) such as for example MBR systems, constructed wetlands, SBR systems and many more. In this paper, we however argue that generic space-dependent expenditures arise in form of residual emptying, service and repair costs which are mainly irrespectively of the system choice (see Section 3.2.1 for an more comprehensive discussion). We see a highly non-linear relationship between space-dependentcosts and low OST densities, but low spatial cost dependencies at high densities. Sewer-based centralised sanitation shows diseconomies of scale for scattered settlements and has well been studied (inter alia Townend 1959, Downing 1969). We also find decreasing economic efficiency for decentralised sanitation, although this is due to economies of density. The knowledge gained about economies of density becomes indispensable in integrated cost comparisons of centralised and decentralised WMS (cf. Eggimann et al. 2015).

The numbers show clearly that the sludge disposal costs dominate the spatial cost behaviour. We define a saturation point of the economies of density as the point at which costs do not decrease by more than 5% of the minimal costs calculated on the basis of the minimum catchment radius r_{min}. This percentage seems plausible given the noise of the modelling approach. Considering the average cost calculation of all potential input parameters in Fig. 3.5, we find that the average economies of density diminish at a plant density of equal to or greater than 1.3 OST/km^2 (decreasing returns to density). This saturation point will shift depending on the emptying approaches and input parameters: we see that saturation is generally reached more quickly with scheduled than unscheduled emptying.

The identification of a saturation point and the 'hockey-stick' shaped cost relationship in the economies of density are the major findings of this investigation. They indicate that costs will be exceedingly high in the initial phase of introducing OST systems in a region until the density surpasses the saturation point. The saturation point enables the "true" costs of OST systems to be estimated if they are implemented in a certain number. This could be used as an argument for centrally regulating the introduction of OST systems (at least in an early introduction phase). Economies of density may also be moderated by the number of operators competing for service contracts in a region. As a result, effective OST plant densities will be reduced by each additional company entering the market. The number of competitors will therefore increase the necessary numbers of OST plants in a given region before their costs can be considered to be constant. This could be used as an argument to limit the number of competitors in a region (or alternatively to put out a call for tenders for servicing contracts for the entire region) in order to reap economies of densities more quickly.

A further major result relates to the outlined emptying approaches to test our model for worst and best case scenarios. On the basis of Fig. 3.4, we conclude that the choice of emptying approach is particularly cost-relevant in low plant densities. Scheduled emptying is greatly preferable to the unscheduled alternative for low-density situations, whereas the differences decrease in high-density situations. However, even though a scheduled emptying approach is more cost-efficient, setting up a scheduled emptying tour may not be realistic because of factors such as highly fluctuating filling rates (e.g. in tourist regions). We notice that despite scheduled emptying being generally more efficient, scheduled costs are higher at very low densities because of the greater distance weighting factor (cf. Section 3.3.4). This preliminary result helps us to identify the optimisation potential in choosing the appropriate technology and management approaches as the specific sludge production of an OST technology seems to be more relevant than maximising the robustness of the plant.[11]

[11] We would need to assume very high failure rates or a high number of service visits in order for this cost relationship to change.

3.4.2 Validation

A systematic validation of our results was not possible because no extensive cost data with respect to different treatment plant densities were available. We would need to know the costs for different point densities to validate the saturation point. It is therefore essential to use realistic values for the model parameters specific to the particular case study (see Table 3.1). Information provided by a local service operator allows us to estimate the real emptying cost of sludge and scum to be around \$50 to \$100 per m^3 (Fritzsche and Maurer 2013) and the service cost to be between \$10 and \$50 per year (Creabeton 2015). As expected (see the limitations outlined in Section 3.4.4), our model underestimates both values. Whereas values could be derived from the literature for most parameters (see Table 3.1), we had to work with reasonable ranges for the time parameters. If more detailed time parameter estimates were required, more elaborate method-time measurements (cf. Karger and Bayha 1987) could be carried out by splitting tasks into subtasks and systematically collecting and evaluating the respective required times. In terms of the chosen routing heuristics, we believe that the Nearest Neighbour algorithm is suitable for use here, as model uncertainties resulting from different routing algorithms are minor. The results obtained are very similar to those of the computationally much less efficient Clarke and Wright algorithm. We conclude from this that it is ineffective in improving the model by heuristic optimisation.

We have shown that we can deduce a robust and distinct relationship between the infrastructure layouts of OST plants and space-dependent costs despite the abstract model design. The cost-space relationship remains robust even when our parameter sets are randomly varied (cf. Section 3.2.7). This robustness improves the validity of our assessed 'hockey stick' shaped cost-density relationship, as this consequentially holds for many different case studies where the individual cost parameters vary depending on the context.

3.4.3 Different application contexts

Although we do not carry out a comparative analysis across different infrastructure domains (Hansman et al. 2006), we are convinced that, given the generality of the presented problem, our results are not of interest only for the case of waste water management. The issues highlighted in this work are generally encountered in solid-waste handling and management and many further application contexts involving similar problems of location-routing modelling (see e.g. Yang and Ogden 2007, Nagy and Salhi 2007 or Current et al 2002 for different application areas).The analogy to further management applications is evident, as many different household devices also depend to some degree on road-based operation and maintenance schemes (e.g. heating systems or washing machines). The findings concerning optimal logistics and costs relating to the distribution and density of such devices or systems are therefore applicable to fields beyond on-site sanitation.

3.4.4 Limitations and future research needs

The absolute resulting cost values are subject to several limitations: first and most important the cost structure used reflected Swiss conditions and might change in other markets. Other factors that influence costs are: first, we do not consider idle capacities of collection trucks and personnel but assume a rental system. We make this simplifying assumption as full utilisation of personnel or vehicles is often hard to achieve in real world applications if trucks are acquired and personnel is hired for sludge collection only. However, idle capacities become less important in the case of increasing densities and thus larger service organisations. This means that we generally underestimate the costs for lower densities. Second, our analysis uses underlying simplifying assumptions which need to be adapted for case study applications. For example, more sophisticated scheduling schemes could be set up taking into account increasing emptying demands after weekends. Additionally, more complex tank filling rates could be implemented or more detailed analysis could be made of the differences in space-dependency with respect to different on-site technologies. Furthermore, we neglect any potential profits made by the operators. In summary, we see many possible modifications or extensions of our analysis, such as increasing the level of realism of the modelling approach or improving the cost model. However, the main conclusions about the economies of density drawn from this model exercise still hold. Many of the mentioned limitations have an influence on the level but not on the shape of the cost curve (e.g. profits). Others are considered by using best and worst case assumptions (e.g. scheduling schemes).

Finally, we want to emphasise that in practice the discussion about on-site treatment often revolves around technology, performance and public acceptability (inter alia Massoud et al. 2009, Larsen et al. 2013). An analysis focusing on cost alone is therefore incomplete. However, costs specifically related to transportation and space are essential for decentralised WMS and it is therefore important to assess them (inter alia Kennedy-Walker et al. 2014).

3.5 Conclusion

In this study, we examine the spatially dependent costs of decentralised WMS. The main goal was to address the lack of knowledge about the density-related cost characteristics (i.e. economies of density) of OST systems. Economies of density are closely linked to the operation and maintenance of OST, and in-depth knowledge concerning this specific cost aspect of O&M is needed to obtain improved true cost comparisons of centralised and decentralised WMS. Our model-based approach allows us to calculate economies of density for non-sewer based WMS in a systematic way for the first time. The input parameters can be adapted to specific real-world applications, and the distance parameters can be estimated.

We summarise our main findings as follows:

- In sparsely populated regions, we not only find a reduced economic efficiency for centralised WMS but also for decentralised ones due to economies of density.
- Economies of density for OST plants are highly non-linear and take the form of a 'hockey stick'.
- Economies of density depend on the chosen management approach, i.e. whether optimisation of road-based transportation is feasible or not. This is especially true for residual emptying at low plant densities where high cost savings can be achieved by optimised routing through scheduled emptying.
- We argue that knowledge about economies of density is especially important in the early introduction phases of OST systems in a region. This is because very low densities are reached in the initial introduction phase of on-site technology as only a small number of systems are implemented by first-movers. At low densities, OST costs will therefore be unduly high compared to a centralised WMS. Only after a minimum number of plants have been installed will be space-dependent costs of OST plants decrease. This cost behaviour might provide a rationale for subsidising the first OST plants in a region in order to achieve attractive prices for these services.
- Based on today's number of OST plants in the Swiss case study region, potential cost savings could be realised if more plants were installed in order to reach higher densities treatment plants. We find that the saturation point for economies of density is between 1 and 1.5 OST plants per km^2. The current relatively high plant density in the overall region is close to the saturation point, even though differences exist between the various sub-regions, indicating locally distinctive costs for service, repairs and residual emptying. However, a comprehensive full-cost analysis of centralised and decentralised WMS would be needed to decide on the economically optimal number of OST plants in a region. The knowledge presented here prepares the ground for such an integrated cost assessment.

Source Code

The model is implemented in Python 2.7 and the source code is available under https://github.com/eggimasv/EcoDen.

Acknowledgement

We thank Eberhard Morgenroth, Heiko Gebauer and João Leitão for fruitful discussions and Richard Michell and Rachel Barrett for proof-reading. Finally, we acknowledge the feedback received from anonymous reviewers. ■

Appendix A

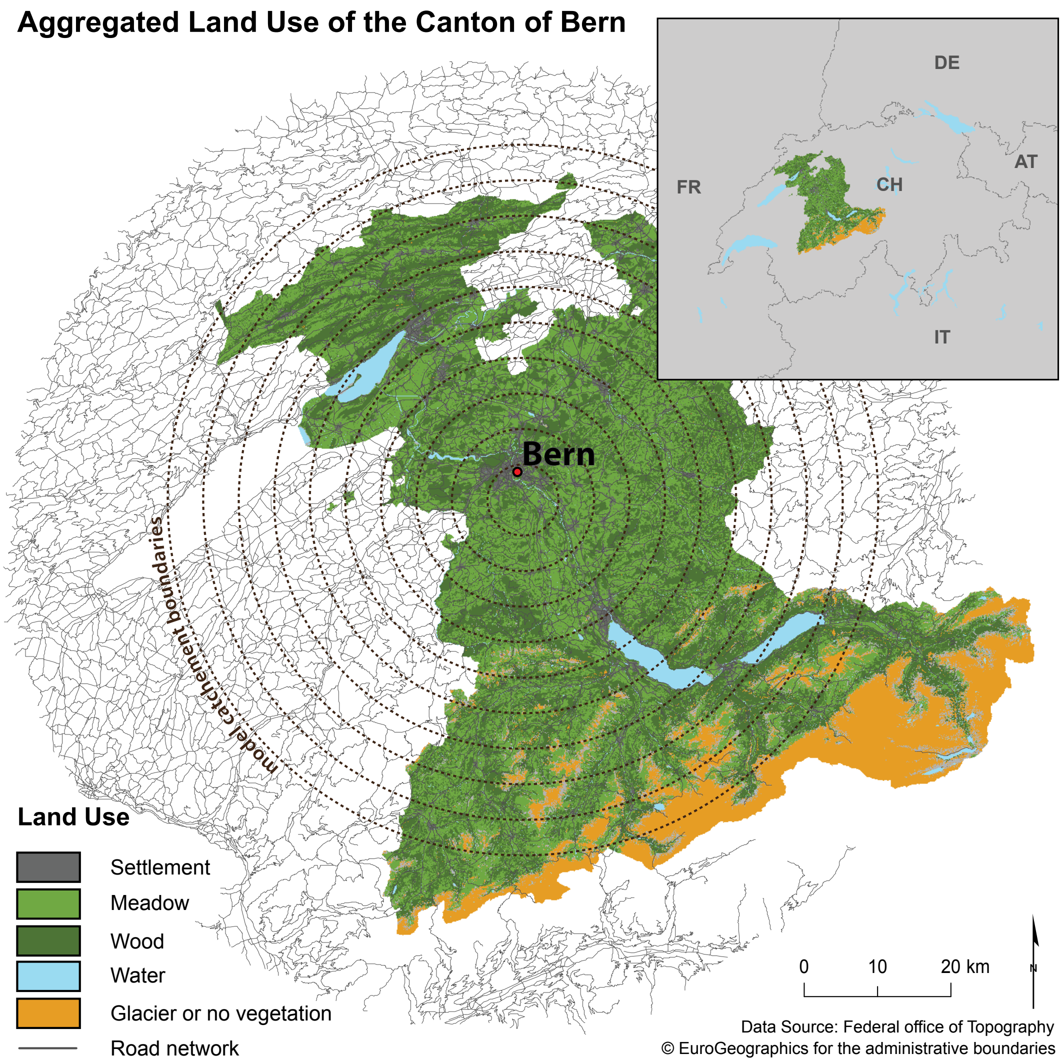


Figure A.3.1: *Land use of the Canton of Bern. For the density calculation we do not consider land covered by vegetation or glaciers. Model catchment boundaries of the case study application up to 100km are indicated with dotted circles.*

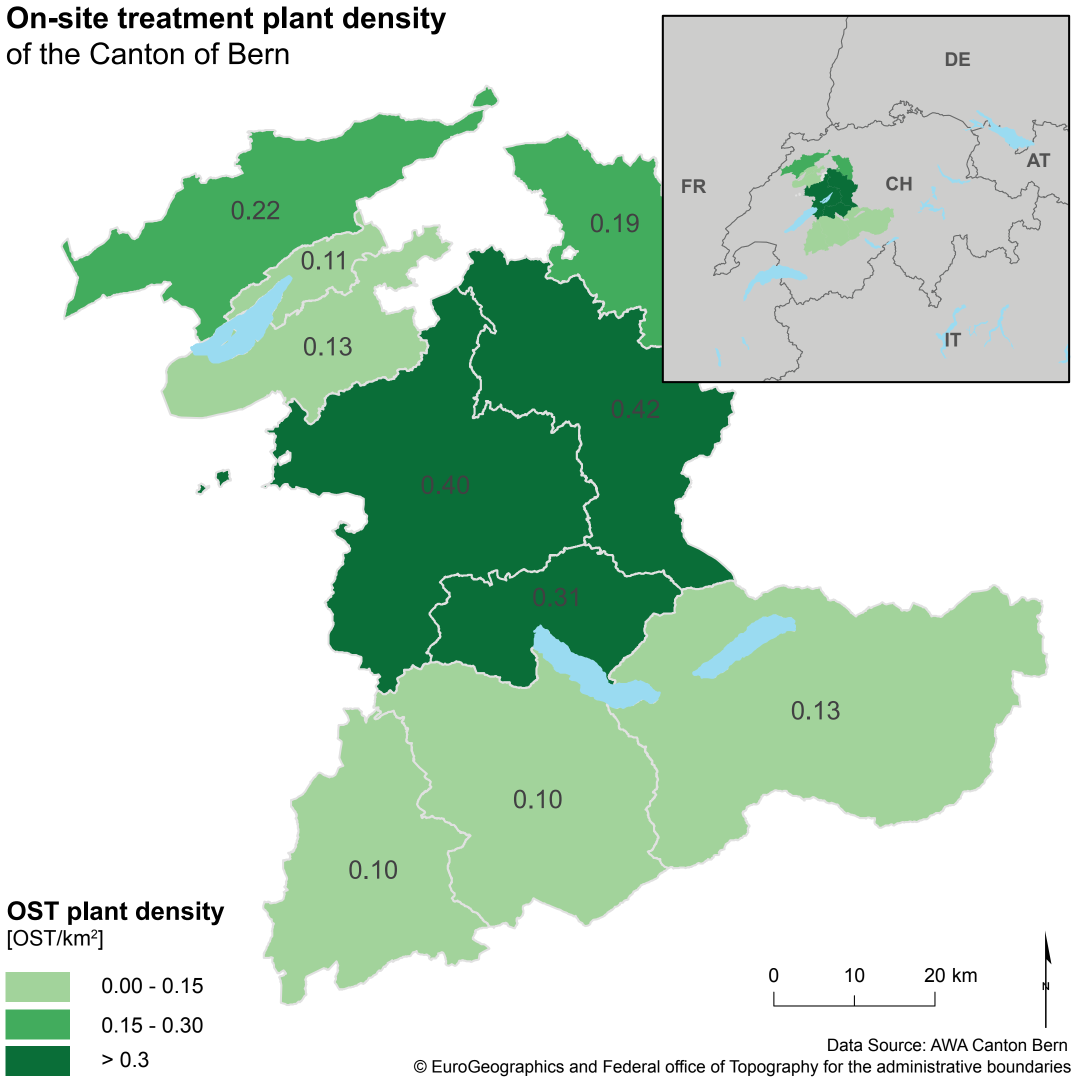


Figure A.3.2: *Current regional OST plant density estimates within the Canton of Bern.*

3.6 References

Adams, B. J., Dajani, J. S., Gemmell, R. S., **1972**. On the centralization of wastewater treatment facilities. *Water resources bulletin,* 8(4), 669–678.

Álvarez, I. C., Prieto, Á. M., Zofío, J. L., **2014**. Cost efficiency, urban patterns and population density when providing public infrastructure: a stochastic frontier approach. *European Planning Studies*, 22(6), 1235–1258.

Amphitec, **2015**. Rental Brochure (Rental Broschüre). Link: Accessed: 30.05.2015. Archived by WebCite® http://www.webcitation.org/6YuYVDPm4 (in German).

Arnott, R., **1979**. Optimal City Size in a Spatial Economy. *Journal of urban economics*, 6, 65-89.

AWA, UWE, **2010**. Sachplan Siedlungsentwässerung 2010 der Kantone Bern und Solothurn (Sectoral Plan Urban Drainage Systems 2010 of the Canton Bern and Solothurn). Switzerland. (in German).

AWA, **2014**. Entsorgung von Rückständen aus dezentralen Abwasseranlagen (Disposal of residues of decentralised wastewater treatment plants). Koordination Nordwetschweiz, Bern, Switzerland (in German).

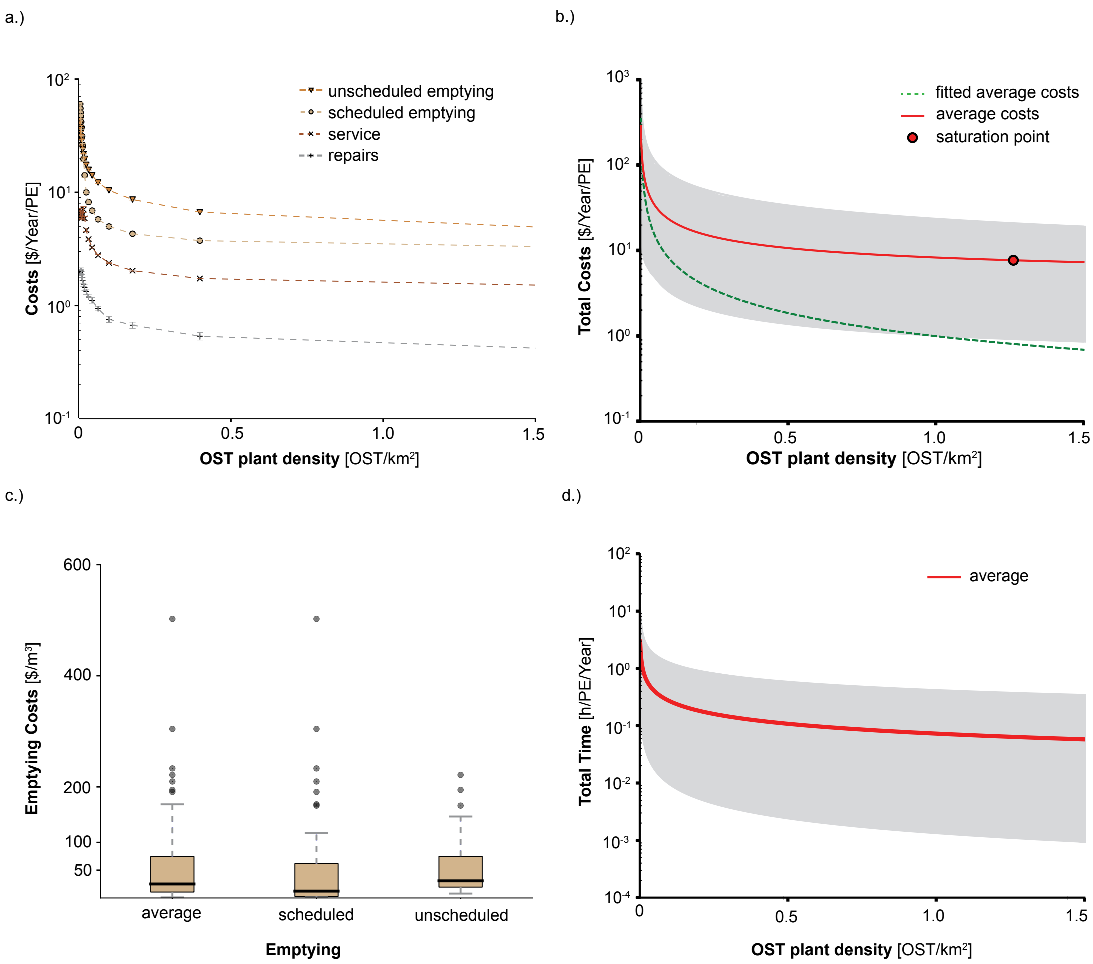


Figure A.3.3: *Non-normalised costs for Figure 3.4 (a), Figure 3.5 (b), non-normalised total emptying costs (c) and total time results (d) (n = 100).*

Bakir, H. A., **2001**. Sustainable wastewater management for small communities in the Middle East and North Africa. *Journal of environmental management*, 61, 319–328.

Barnes, G., Langworthy, P., **2003**. The per mile costs of operating automobiles and trucks. Final Report. Minnesota Department of Transportation, MN, USA.

BFS, **2009**. Transportrechnung – Jahr 2005 (Transport Accounting – Year 2005). Neuchâtel, Switzerland (in German).

BFS, **2013**. Betriebsübliche Arbeitszeit nach Wirtschaftsabteilungen (Typical working times for different economic devisions). Neuchâtel, Switzerland (in German).

BFS, **2015**. Schweizerische Lohnstrukturerhebung 2012 (Swiss Earnings Structure Survey 2012). Neuchâtel, Switzerland (in German).

BFS, **2015**b. Treibstoff - Durchschnittspreise pro Liter in Franken (Fuel - Average prices per litre in Francs), Neuchâtel, Switzerland, (in German).

Bradley, B., Daigger, G., Rubin, R., Tchobanoglous, G., **2002**. Evaluation of onsite wastewater treatment technologies using sustainable development criteria. *Clean Technologies and Environmental Policy*, 4, 87–99.

Buchanan, J. R., **2014**. Decentralized Wastewater Treatment. Book chapter, volume three, 'Remediation of Polluted Water', in the Series 'Comprehensive

Water Quality and Purification.' Elsevier, Oxford, UK.

Cambridge Systematics, **1995**. Characteristics and Changes in Freight Transportation Demand: A Guidebook for Planners and Policy Analysts. Appendix F. Prepared for NCHRP Project 8-30. Washington, USA.

Christaller, W., **1933**. Die zentralen Orte in Süddeutschland. Reprinted in: Wissenschaftliche Buchgesellschaft (eds), 1980, Third Edition, Darmstadt, Germany.

Clarke, G., Wright, J.W., **1964**. Scheduling of vehicles from a central depot to a number of delivery points. *Operations Research*, 12, 568–581.

Cormen, T. H., Leiserson, C. E., Rivest, R. L., Stein, C., **2009**. Introduction to Algorithms. MIT Press, Cambridge, Massachusetts, London, UK.

Creabeton, **2015**. Mechanisch-biologische Kleinkläranlagen (Mechanical and biological small-scale wastewater treatment plants). Einingen, Switerland. Link: Accessed: 30.06.2015. Archived by WebCite® http://www.webcitation.org/6ZfjCunUO

Crites, R., Tchobanoglous, G., **1998**. Small and decentralized wastewater management systems. WCB McGraw-Hill, Boston, USA.

Current, J., Daskin, M.D., Schilling, **2002**, "Discrete Network Location Models," Chapter 3 in facility location theory: applications and methods , Z. Drezner and H. Hamacher (eds), Springer-Verlag, Berlin, 81–118.

Dijkstra, E. W., **1959**. A note on two problems in connexion with graphs. *Numerische Mathematik,* 1, 269–271.

Dodane, P.-H., Mbéguéré, M., Sow, O., Strande, L., **2012**. Capital and Operating Costs of Full-Scale Faecal Sludge Management and Wastewater Treatment Systems in Dakar, Senegal. *Environmental Science and Technology,* 46, 3705–3711.

Downing, P. B., **1969**. The economics of urban sewage disposal. Praeger, New York, NY, USA.

Eggimann, S., Truffer, B., Maurer, M., **2015**. To connect or not to connect? Modelling the optimal degree of centralisation for wastewater infrastructures. *Water Research*, 84, 218–231.

EPRI, **2000**. Advanced On-Site Wastewater Treatment and Management Market Study. Volume 2: State Reports, Palo Alto, CA, USA.

Etnier, C., Nelson, V., Pinkham, R., **2000**: Economics of Decentralized Wastewater Treatment Systems: Direct and Indirect Costs and Benefits. A white paper for the National Decentralized Water Resources Capacity Development Project. In: *National Research Needs Conference Proceedings: Risk-Based Decision Making for Onsite Wastewater Treatment*, St. Louis, MO, USA.

Finkelstein, M., **2008**. Failure Rate Modelling for Reliability and Risk. Springer Series in Reliability Engineering, London, UK.

Fletcher, H., Mackley, T., Judd, S., **2007**. The cost of a package plant membrane bioreactor. *Water Research*, 41, 2627–2635.

Flotats, X., Bonmatí, A., Fernández, B., Magrí, A., **2009**. Manure treatment technologies: on-farm versus centralized strategies. NE Spain as case study. *Bioresource Technology*, 100, 5519–26.

Franceys, R., Pickford, J., Reed, R., **1992**. A Guide to the Development of on-Site Sanitation. WHO.

Friedler, E., Pisanty, E., **2006**. Effects of design flow and treatment level on construction and operation costs of municipal wastewater treatment plants and their implications on policy making. *Water Research*, 40, 3751–3758.

Fritzsche, C., Maurer, M., **2013**. Selected Prices and Costs for Swiss Urban Water Infrastructure. Eawag, Dübendorf, Switzerland.

Gendreau, M., Laporte, G., Séguin, R., **1996**. Stochastic vehicle routing. *European Journal of Operational Research*, 88, 3–12.

Ghose, M.K., Dikshit, A.K., Sharma, S.K., **2006**. A GIS based transportation model for solid waste disposal - A case study on Asansol municipality. *Waste Management*, 26(11), 1287–1293.

González-Gómez, F., García-Rubio, M. A., **2008**. Efficiency in the management of urban water services. What have we learned after four decades of research? *Hacienda Pública Española*, 185, 39–67.

Guerrini, A., Romano, G., Campedelli, B., **2013**. Economies of Scale, Scope, and Density in the Italian Water Sector: A Two-Stage Data Envelopment Analysis Approach. *Water Resources Management*, 27, 4559–4578.

Guo, T., Englehardt, J., **2015**. Principles for scaling of distributed direct potable water reuse systems: A modeling study. *Water Research*, 75, 146–163.

Hamilton, B. A., Pinkham, R. D., Hurley, E., Watkins, K., Lovins, A. B., Magliaro, J., Etnier, C., Nelson, V., **2004**. Valuing Decentralized Wastewater Technologies: a catalog of benefits costs and economic analysis techniques. Rocky Mountain Institute, Snowmass, USA.

Hansman, R. J., Magee, C., Neufville, R. de., Robins, R., Roos, D., **2006**. Research agenda for an integrated approach to infrastructure planning, design and management. *International journal of critical infrastructures,* 2(2/3), 146–159.

Haug, P., **2004**. Decreasing population and rising costs of providing water and sewage treatment for urban areas: a case study, Halle (Saale), Germany.

Hendrickson, T. P., Nguyen, M., Sukardi, M., Miot, A., Horvath, A., Nelson, K. L. **2015**. Life-cycle energy use and greenhouse gas emissions of a building-scale wastewater treatment and non-potable reuse system. *Environmental Science and Technology*, 49, 10303–10311.

Holmes, T. J., **2011**. The diffusion of walmart and economies of density. E*conometrica*, 79(1), 253–302.

Hug, T., Maurer, M., **2012**. Stochastic modeling to identify requirements for centralized monitoring of distributed wastewater treatment. *Water Science and Technology*, 65(6), 1067–1075.

Johnson, D. S., Papadimitriou, C. H., **1985**. Performance guarantees for heuristics. In: Lawler, E.L., Lenstra, J.K., Rinnooy Kan, A.H.G., Shmoys, D.B. (eds). *The Traveling Salesman Problem. A Guided Tour of Combinatorial Optimization*. Wiley, Chichester, UK.

Kaminsky, J., Javernick-Will, A., **2013**. Contested Factors for Sustainability: Construction and Management of Household On-Site Wastewater Treatment Systems. *Journal of Construction Engineering & Management*, 139(12), A4013004.

Karger, D. W., Bayha, F. H., **1987**. Engineered Work Measurement. Fourth Edition, Industrial Press, NY, USA.

Kennedy-Walker, R., Holderness, T., Barr, S., Alderson, D., Evans, B., **2014**. Network modelling for road-based faecal sludge management. *Proceedings of the ICE - Municipal Engineer*, 167(3), 157–165.

Larsen, T. A., Udert, K. M., Lienert, J., **2013**. Source Separation and Decentralization for Wastewater Management. IWA Publishing, London, UK.

Lawler, E. L., Lenstra, J. K., Rinnooy Kan, A. H. G., Shmoys, D. B. (eds), **1985**. The Traveling Salesman Problem. A Guided Tour of Combinatorial Optimization. Wiley, Chichester, UK.

Lee, E. J., Criddle, C. S., Bobel, P., Freyberg, D. L., **2013**. Assessing the scale of resource recovery for centralized and satellite wastewater treatment. *Environmental Science and Technology*, 47, 10762–10770.

Libralato, G., Ghirardini A. V., Avezzù, F., **2012**. To centralise or to decentralise: An overview of the most recent trends in wastewater treatment management. *Journal of Environmental Management*, 94, 61–68.

Markard, J., **2009**. Characteristics of Infrastructure Sectors and Implications for Innovation Processes. Discussion Paper for the Workshop on Environmental Innovation in Infrastructure Sectors, Karlsruhe Sept. 29–Oct. 1, Germany.

Marufuzzaman, M., Eksioglu, S. D., Hernandez, R., **2015**. Truck versus pipeline transportation cost analysis of wastewater sludge. *Transportation Research Part A: Policy and Practice*, 74, 14–30.

Massoud, M. A., Tarhini, A., Nasr, J. A., **2009**. Decentralized approaches to wastewater treatment and management: applicability in developing countries. *Journal of Environmental Management*, 90, 652–659.

Maurer, M., Rothenberger, D., Larsen, T. A., **2006**. Decentralised wastewater treatment technologies from a national perspective: at what cost are they competitive? *Water Science and Technology*, 5(6), 145–154.

Maurer, M., Wolfram, M., Herlyn, A., **2010**. Factors affecting economies of scale in combined sewer systems. *Water Science and Technology*, 62(1), 36–41.

Michaelewicz, Z., Fogel, D. B., **2004**. How to Solve It: Modern Heuristics. 2nd revised and extended edition, Springer, Berlin, Germany.

Mikhael, G., Robbins, D. M., Ramsay, J. E., Mbéguéré, M., **2014**: Methods and Means for Collection and Transport of Faecal Sludge. In: Strande, L., Ronteltap, M., Brdjanovic, D. (eds). *Faecal Sludge Management Systems Approach for Implementation and Operation.* IWA Publishing.

Mobility, **2015**. Overview of mobility rates. Link: Accessed: 30.05.2015. Archived by WebCite® http://www.webcitation.org/6YuZ3WJAv

Nagy, G., Salhi, S., **2007**. Location-routing: Issues, models and methods. *European Journal of Operational Research*, 177(2), 649–672.

OECD, **2015**. Water and Cities: Ensuring Sustainable Futures, OECD Studies on Water, OECD Publishing, Paris, France.

Parkinson, J., Tayler, K., **2003**, Decentralized wastewater management in peri-urban areas in low-income countries. *Environment & urbanization*, 15(1), 75–90.

Poustie, M. S., Deletic, A., Brown, R. R., Wong, T., de Haan, J., Skinner, R., **2014**. Sustainable urban water futures in developing countries: the centralised, decentralised or hybrid dilemma. *Urban Water Journal*, 1–16.

Saltelli, A., Tarantola, S., Campolongo, F., Ratto, M., **2004**. Sensitivity analysis in practice. A guide to assessing scientific models. Wiley & Sons, Chichester, UK.

Schiller, G., Siedentop, S., **2005**. Follow-up costs for infrastructure of settlement development in the case of decrease in population (Infrastrukturfolgekosten der Siedlungsentwicklung unter Schrumpfungs-bedingungen). *DISP - The Planning Review*, 41(160), 83–93 (in German).

Semiyaga, S., Okure, M. A. E., Niwagaba, C. B., Katukiza, A. Y., Kansiime, F., **2015**. Decentralized options for faecal sludge management in urban slum areas of Sub-Saharan Africa: A review of technologies, practices and end-uses. *Resources, Conservation and Recycling*, 104, 109–119.

Singh, N. K., Kazmi, A. A., Starkl, M., **2015**. A review on full-scale decentralized wastewater treatment systems: techno-economical approach. *Water Science and Technology*, 71, 468–478.

Starkl, M., Parkinson, J., Narayana, D., Flamand, P., **2012**. Small is beautiful but is large more economical? Fresh views on decentralised versus centralised wastewater management. *Water21(June)*, 45–47.

Steiner, M., Agnès, M., Koné, D., Strauss, M., **2002**. Economic Aspects of Low-cost Faecal Sludge Management. Eawag, Dübendorf, Switzerland.

Tavares, G., Zsigraiova, Z., Semiao, V., Carvalho, M.G., **2009**. Optimisation of MSW collection routes for minimum fuel consumption using 3D GIS modelling. *Waste Management*, 29(3), 1176–1185.

Tchobanoglous, G., Ruppe, L., Leverenz, H., Darby, J., **2004**. Decentralized wastewater management: challenges and opportunities for the twenty-first century. *Water Science and Technology: Water Supply*, 4(1), 95–102.

Tchobanoglous, G., Leverenz, H., **2013**. The rationale for decentralization of wastewater infrastructure. In: Larsen, T.A., Udert, K.M., Lienert, J. (eds). *Source Separation and Decentralization for Wastewater Management*, IWA Publishing, London, UK.

Townend, C. B., **1959**. The economics of the disposal of sewage and trade effluents. *Bull. World Health Organ*, 20, 535–562.

Truffer, B., Binz, C., Gebauer, H., Störmer, E., **2013**. Market success of on-site treatment: a systemic innovation problem. In: Larsen, T.A., Udert, K.M., Lienert, J. (eds). *Source Separation and Decentralization for Wastewater Management*. IWA Publishing, London, UK.

Von Thünen, J. H. v., **1875**. Der isolierte Staat in Beziehung auf Landwirtschaft und Nationalökonomie. Reprinted in: Wissenschaftliche Buchgesellschaft (eds), 1966, Third Edition, Darmstadt, Germany.

Wegener, M., **2011**: Transport in spatial models of economic development. In: Palma, A.D., Lindsey, R., Quinet, E., Vickerman, R. (eds). *A handbook of transport economics*. Edward Elgar Publishing, Cheltenham, UK.

Wenban-Smith, H. B., **2009**. Economies of scale, distribution costs and density effects in urban water supply: a spatial analysis of the role of infrastructure in urban agglomeration. PhD thesis, The London School of Economics and Political Science (LSE), UK.

WERF, **2015**. Performance and cost of decentralized unit processes. Link: http://www.werf.org

World Bank, **2014**. PPP conversion factor. International Comparison Program database. Available at: http://data.worldbank.org/

Yang, C., Ogden, J., **2007**. Determining the lowest-cost hydrogen delivery mode. *International Journal of Hydrogen Energy*, 32(2), 268–286.

Zamorano, M., Molero, E., Grindlay, A., Rodríguez, M. L., Hurtado, A., Calvo, F. J., **2009**. A planning scenario for the application of geographical information systems in municipal waste collection: A case of Churriana de la Vega (Granada, Spain). *Resources, Conservation and Recycling*, 54, 123–133.

Zeppini, P., Frenken, K., Kupers, R., **2014**. Thresholds models of technological transitions. *Environmental Innovation and Societal Transitions*, 11, 54–70.

Chapter 4

The cost of hybrid waste water systems: a systematic framework for specifying minimum cost-connection rates

published in ***Water Research****, 103, 472–484.*

The cost of hybrid waste water systems: a systematic framework for specifying minimum cost-connection rates

Eggimann Sven[1, 2*] Truffer Bernhard[1,3], Maurer Max[1, 2]

[1] Eawag, Swiss Federal Institute of Aquatic Science and Technology, 8600 Dübendorf, Switzerland.

[2] Institute of Civil, Environmental and Geomatic Engineering, ETH Zürich, 8093 Zurich, Switzerland.

[3] Faculty of Geosciences, Utrecht University, Heidelberglaan 2, NL-3584 CS Utrecht, The Netherlands.

Keywords

Urban water management, engineering economics, urban structural unit, geographical information system, decentralised waste water management, sustainability transition

Abstract

To determine the optimal connection rate (CR) for regional waste water treatment is a challenge that has recently gained the attention of academia and professional circles throughout the world. We contribute to this debate by proposing a framework for a total cost assessment of sanitation infrastructures in a given region for the whole range of possible CRs. The total costs comprise the treatment and transportation costs of centralised and on-site waste water management systems relative to specific CRs. We can then identify optimal CRs that either deliver waste water services at the lowest overall regional cost, or alternatively, CRs that result from households freely choosing whether they want to connect or not. We apply the framework to a Swiss region, derive a typology for regional cost curves and discuss whether and by how much the empirically observed CRs differ from the two optimal ones. Both optimal CRs may be reached by introducing specific regulatory incentive structures.

4.1 Introduction

Sanitation services in a region may in principle be provided by centralised or decentralised on-site waste water management systems (WMS) (Libralato et al. 2012). On-site WMS enable waste water to be treated geographically close to the point of generation (Tchobanoglous and Leverenz 2013), making costly investments in sewer networks obsolete and potentially allowing cost savings. Despite the potential advantages, however, centralised WMS have gained much higher market shares in most OECD countries over the past century. The primary rationale for this was to assure high levels of 'urban hygiene' (O'Flaherty 2005, Sedlak 2014). Moreover, centralised WMS were promoted by public regulators because of compatibility with currently existing systems, known manageability, well-defined performance as well as economies of scale in both waste water treatment and sewer management (Townend 1959, Downing 1969, Abd El Gawad and Butter 1995, Libralato et al. 2012). Over the years, institutions, organisations and the technology have co-evolved, leading to shared values, a professional culture based on civil engineering competences, and particular organisational forms dominated by utilities under public ownership (Dominguez 2008, Kiparsky et al. 2013, Fuenfschilling and Truffer 2014, Fane and Fane 2005, Lieberherr and Truffer 2015, Lieberherr and Fuenfschilling 2016). These alignments created strong path dependencies (Arthur 1989), so that today's catchments are dominated by large centralised waste water treatment plants (WWTP) and extensive sewer networks connecting large percentages of the population. Empirically, we observe a wide variety of connection rates (CR): whereas most emerging economies and developing countries are characterised by very low (typically $<<$ 50%) CRs (UN 2015), some OECD countries (e.g. Switzerland, Austria, the Netherlands and the United Kingdom) have pushed for very high CRs ($CR_{present}$) of >95%, whereas other OECD countries (e.g. Ireland, Slovenia or Poland) have a $CR_{present}$ of between 60 and 70% (OECD 2015).

The long-term superiority of very high CR has lately been questioned, and this has led to a call for a 'sustainability transition' towards more hybrid configurations combining centralised and on-site WMS (Fane and Fane 2005, Daigger 2007, Truffer et al. 2010, Larsen et al. 2013, Marlow et al. 2013). A wide range of criteria (e.g. technical, environmental, public-health related, institutional, social, economic) can be used to determine the optimal mixing rate. In recent years, however, we can observe an increasing predominance of economic efficiency criteria in the planning of network-based infrastructures (Knops 2008). Economic assessments of optimal infrastructure dimensioning have also gained increasing attention in the field of water management, not only for waste water (Eggimann et al. 2015, Lee et al. 2013), but also for drinking water (Poustie et al. 2014, Guo and Englehardt 2015), hydro power (Kaundinya et al. 2009) and seawater desalination (Shahabi et al. 2015). This heightened interest is due to strained public budgets, often leading to infrastructural underinvestment (WEF 2010), the demand for more infrastructure flexibility and recent advances in on-site treatment technology. Furthermore, a modular approach to infrastructure planning is becoming increasingly cost competitive: new sensor and communication technologies allow automation and mass production which drive down the cost of small standardised units (Dahlgren et al. 2013). Determining the optimal connection rate (OCR) therefore remains a relevant question to reconsider.

In the present paper, we focus exclusively on cost assessments, as they often play an important role in designing WMS (Maurer et al. 2006). The goal is to develop an encompassing framework for assessing the total costs of hybrid WMS (Tchobanoglous and Leverenz 2013) in a given region. Even though much effort has been spent on the cost considerations of WMS (Townend 1959, Downing 1969, Adams et al 1972, Etnier et al. 2000, Hamilton et al. 2004, Maurer et al. 2010, Libralato et al. 2012, Eggimann et al. 2015), there is a paucity of conceptual work focusing on systematic total cost assessments. We build on an extensive body of work and present a framework within which we deduce generic cost curves for all key cost elements of a hybrid WMS. On the basis of these considerations, we will provide alternative interpretations of the OCR depending on specific institutional arrangements and organisational set-ups of providers of WMS services. This will enable us to discuss just what 'more sustainable' WMS configurations in specific regions could be, and in particular to discuss to what extent the $CR_{present}$ deviates from the various OCRs.

4.2 Material and methods

4.2.1 Framework for total cost assessment

This section starts by introducing the general assumptions of our framework (Section 4.2.1.1), and continues by identifying all key cost components of centralised and on-site WMS needed for a total cost assessment of hybrid WMS in a region (Section 4.2.1.2).

4.2.1.1 General assumptions

The framework for assessing total costs of hybrid WMS in a region presented here draws on the following general assumptions:

- Households and utility operators prefer each system only on the basis of average cost considerations.[1]
- All households have to be served either by being connected to the sewers or installing on-site WMS.
- The average regional total costs at each CR are defined by the average per capita costs of both systems as well as being annualised on the basis of the expected life-spans of the corresponding assets.
- We use average costs as a meaningful approximation for individual household sanitation costs. We are aware that actual tariff systems often diverge from these average costs, as they may include block tariffs, subsidies, base fees or connection fees (OECD 2010).
- To ensure human and environmental health, centralised and on-site WMS need to fulfil the same functionality and provide an equivalent service. This implies that on-site WMS have to be equipped with treatment performance comparable to that of centralised WWTPs, and that on-site effluent disposal is possible (e.g. infiltration or on-site discharge into waters). We consequently assume that the sewers are built exclusively for waste water transportation and no synergies with storm water evacuation have to be accounted for (cf. Section 4.4.3).
- We assume that our region consists only of households and aggregated households in urban structural units (see Section 4.2.1.2) and no industry.[2]

4.2.1.2 Total costs of hybrid WMS

The total WMS costs C_{tot} can be subdivided into waste water treatment $C_{treatment}$ and waste water transport $C_{transport}$ costs. For centralised WMS, treatment occurs in one large WWTP $C^{cen}_{treatment}$ whereas for decentralised WMS, treatment is on-site $C^{dec}_{treatment}$. Transportation is either road-based in case of decentralised WMS $C^{dec}_{transport}$ or sewer-based for centralised WMS $C^{cen}_{transport}$. The total regional cost C^{region}_{tot} of a WMS can thus be specified as:

$$C^{region}_{tot} = \underbrace{C^{cen}_{treatment} + C^{dec}_{treatment}}_{treatment} + \underbrace{C^{cen}_{transport} + C^{dec}_{transport}}_{transport} \qquad (4.1)$$

[1] We conduct the entire cost assessment procedure by means of average cost calculations for each system. As we are interested in long-term optimal equilibrium solutions, this assumptions may be justified.

[2] However, this restriction is not decisive for the general argument that we develop and merely implies that the cost curves of the centralised WMS have a much bumpier shape than our idealised representation.

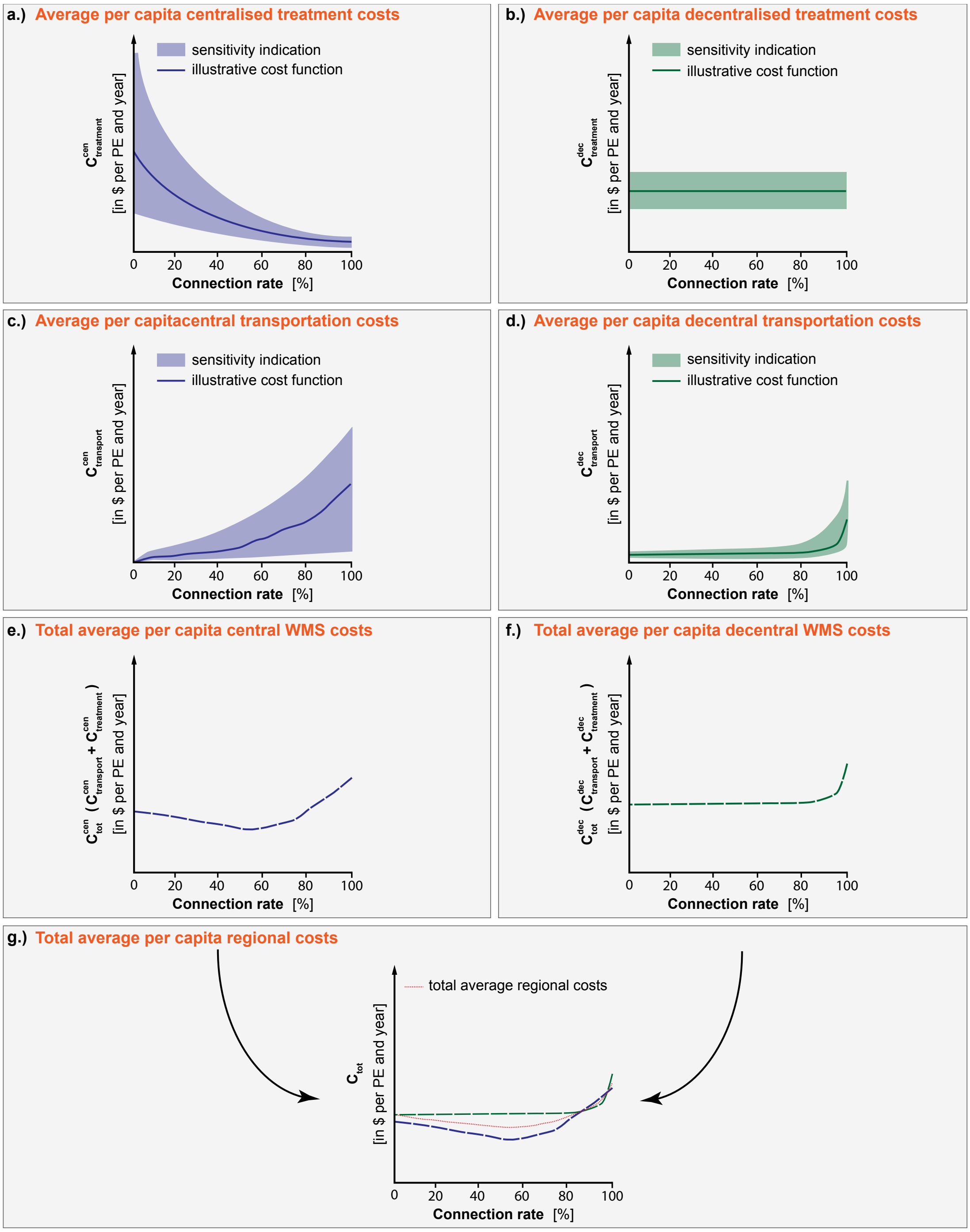


Figure 4.1: *Idealised average per capita cost functions over all CRs. The final cost curve (configuration g.) corresponds to the cost type C in Figure 4.10. See also Section 4.2.2.3 for underlying material and methods.*

Fig. 4.1 shows the generic functional forms of the cost components of C_{tot}^{region} as a function of the CR along the respective sensitivity bands. The average total costs of either system (e, f) can be calculated on the basis of the cost function of the centralised (a, c) and decentralised WMS (b, d). Finally, the average total regional costs at a specific CR (assuming that this CR corresponds to a share p of households connected to the centralised system, whereas 1-p have on-site treatment) can be expressed as the weighted sum (dotted red line in g):4

$$C_{tot}^{region}(CR) = C_{tot}^{cen}(CR) + (1\text{-}p) * C_{tot}^{dec}(CR) \quad (4.2)$$

Each of the specific shapes of the cost curves is based on different assumptions (outlined below): they are either derived directly from explicit cost data (a, b) or are model-based (c, d). The shapes are idealised, i.e. they vary depending on the specific case study. However, although we have varied the underlying assumptions to obtain cost ranges for each cost element, we find that the behaviour of each cost curve can be described in fairly generic terms:

Centralised treatment ($C_{treatment}^{cen}$, Fig. 4.1a): the prevailing key economic argument given in the literature to realise high CRs relates to economies of scale (inter alia Townend 1959, Downing 1969, Libralato et al. 2012). The likely decrease in average per capita costs for a large WWTP is inversely proportional to the number of households connected. In the literature, it is commonly implicitly assumed that the WWTP perfectly fits the demand of the connected users for each CR, and idle capacities are neglected. However, as investments in WWTP are typically based on the peak performance during the planning horizon (typically 20 to 30 years) (Hug et al. 2010), neglecting idle capacities underestimates treatment costs in catchments with positive or negative growth. The sensitivity indication in Fig. 4.1a reflects two extreme cases of idle capacities: the bottom border neglects idle capacities altogether while the top border indicates an investment scenario that considers maximum idle capacities. For the latter scenario, we calculate an initial expenditure of one WWTP serving the whole catchment and distribute this investment equally amongst the connected population at each CR.

Centralised transportation ($C_{transport}^{cen}$, Fig. 4.1c): sewer networks enable the transportation of waste water to the WWTP. Sewer construction and operation costs are heavily influenced by geography, settlement distribution or population density. We consequently find decreasing marginal costs for higher CRs and complex cost functions depending on the geography (cf. Adams et al 1972, Hamilton et al. 2004, Maurer et al. 2010, Eggimann et al. 2015). In reality, most sewer systems are built up iteratively, where each new settlement structure to be connected leads to particular cost curves in terms of shape and cost level. Thus clustered settlement structures prevent constant cost increases and lead to 'jumps' (Zvoleff et al. 2009) in the cost curve. Fig. 4.1c shows a generic sewer cost function with increasing average per capita costs for higher CRs as a result of heterogeneous settlement structures, and correspondingly higher costs for connecting more distant settlements.[3] The sensitivity indication reflects differences in the cost curve depending on the catchment context factors outlined above. Whereas cost curves can be derived from the detailed cost data of existing networks, model-based approaches allow us to overcome the lack of data or the influence of legacy infrastructures and to systematically assess sewer costs across different catchments.

Decentralised treatment ($C^{dec}_{treatment}$, Fig. 4.1b): the costs of on-site treatment are largely independent of specific CRs, and the generic cost function is thus constant (Fig. 4.1b). However, the installation costs may differ depending on local conditions (e.g. rural or urban setting) and on the system type (Singh et al. 2015). We also find economies of scale for on-site treatment, i.e. per capita costs are typically lower for a 20-person system than a 4-person one. The level of the cost curve can consequently differ, which we represent by the respective shaded area. We assume that the housing structure does not change for different CR, so that the cost curve remains constant over the entire CR range.

Decentralised transportation ($C^{dec}_{transport}$, Fig. 4.1d): in the case of on-site WMS, well-functioning operation and maintenance (O&M) schemes are necessary to achieve full functionality. Road-based transportation needs result from professionals having to access the plants as well as from residual evacuation. For operating and managing on-site WMS, we find economies of density (Eggimann et al. 2016), i.e. cost savings due to the numbers and spatial proximity of on-site WMS. However, this effect is limited to a rather small range of treatment plant densities. A generic cost function describing all transport-related costs is given in Fig. 4.1d, where the range of different cost functions is due to different O&M concepts.

Total costs: the total cost curve of centralised WMS is shown in Fig. 4.1e with its characteristic 'u-shaped' form (Adams et al. 1972). The total cost curve of on-site WMS results in a 'hockey-stick' shape as seen in Fig. 4.1f (Eggimann

[3] For the idealistic cost curve representation in Fig. 4.1c we assume that the dimension of the CR is ordered in a way that enables a monotonic presentation of the sewer cost curve, i.e. the x-axis proceeds from houses that are near the WWTP to those more distant from it. At the same time, we assume that the settlement density is highest around the WWTP and decreases over distance.

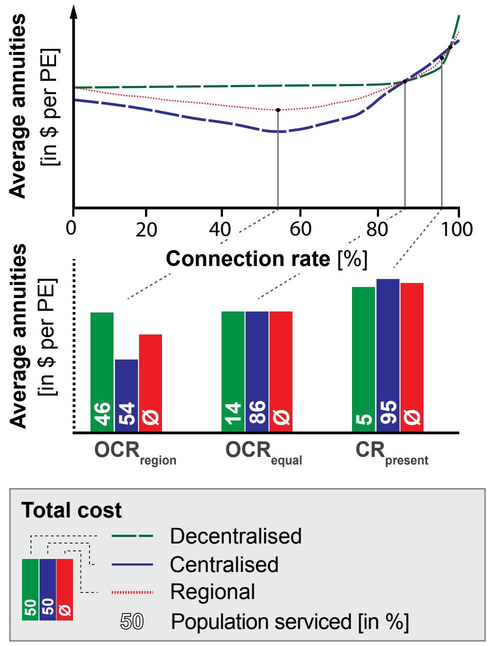


Figure 4.2: Calculated OCR, costs and distribution of central and on-site WMS from an exemplary cost curve configuration. A $CR_{present}$ of 95% is shown. *With help of the bar charts, the costs of the three total cost curves are visualised for all three CR. The numbers on the bar charts show the percentages of the population serviced with centralised or decentralised systems respectively for each CR*

et al. 2016). The resulting total average regional costs (C_{tot}^{region}) for hybrid WMS can now be derived from the total costs of both centralised and decentralised WMS (Eq. 4.2).

4.2.1.3 Optimal connection rate

The total regional cost curve and the respective centralised and on-site total cost curves exhibit some notable characteristics (Fig. 4.1g, Fig. 4.2). Firstly, we argue that the basic shapes of these curves are quite generically valid: both the ‘u-form’ shape of the centralised system and the ‘hockey stick’ of on-site systems have been identified in earlier literature (Adams et al. 1972, Eggimann et al. 2016). Secondly, by ignoring the trivial cases where one curve dominates the other, we expect two intersection points between these curves to exist almost independently of the specific cost characteristics. Thirdly, there will be a minimum on the regional total cost curve.

These points may be interpreted as different candidates for potential OCRs. The difference between these OCRs and the $CR_{present}$ can be interpreted as the regional cost improvement potential of WMS. The two relevant OCRs can be characterised as follows:

- The OCR_{region} is defined by the minimum on the regional total cost curve where the average aggregated costs for the entire region are minimal. However, the specific costs of the different systems are not equal at this point, in view of higher costs for on-site treatment.

- The lower intersection point $OCR_{equalcost}$ marks the CR where the specific costs of the two WMS options are equal. This point would be reached spontaneously if all households could opt for the cheapest system in their specific location.[4] The sanitation costs for each household are the same, irrespectively of the system choice.

4.2.2 Case study application

In this section, we apply the framework outlined here to an empirical case-study region in order to test whether and how the different OCR can be identified. We aim to derive general cost patterns in the form of a configuration typology from the various case-study catchments.

4.2.2.1 Canton of Glarus

We select the Canton of Glarus, a region with a population of ~40,000 in the north-east of Switzerland and covering an area of 685 km² (Fig. 4.3). We chose Glarus because it is a diverse region in terms of topography and settlement distribution which provides diverse contexts with respect to cost-curve configurations. This can be seen in the fact that we already find different CRs there (Fig. 4.3). The region underwent an organisational reform in 2011 - the 'Glarner Gemeindereform'

[4] The higher-level intersection would fulfill the criterion of equal costs equally well, but it represents a substantially higher level of total regional costs, so we do not elaborate further on its significance.

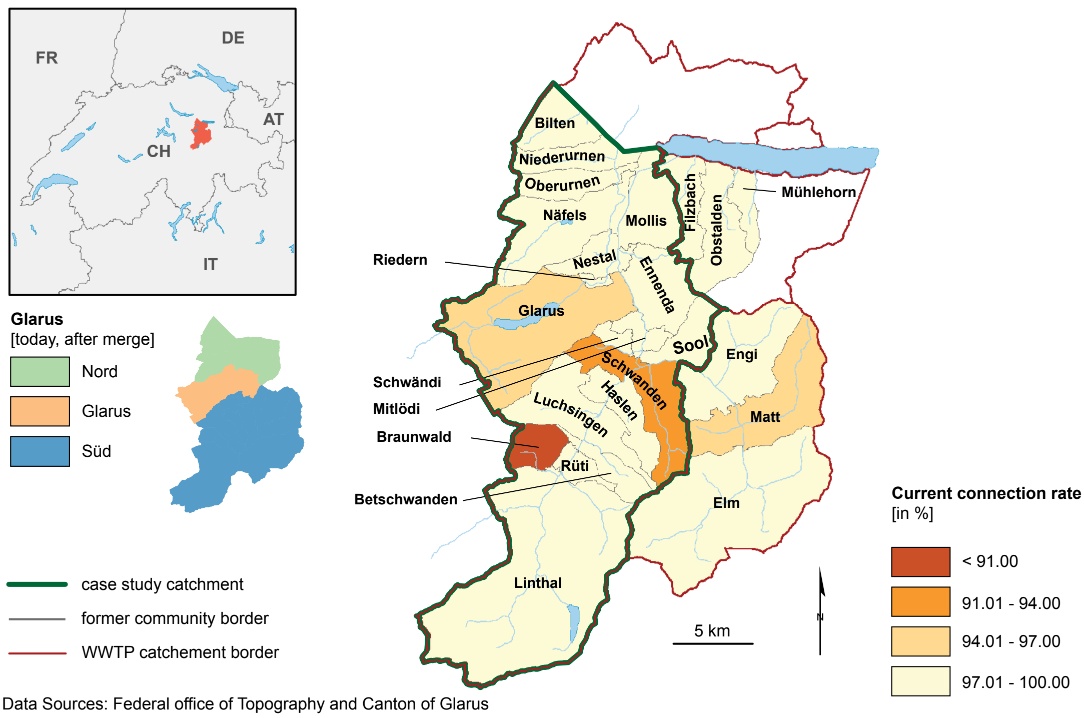


Figure 4.3: *Case study area showing the present and former community boundaries as well as different WWTP catchments. White areas belong to external catchment areas or are not part of the Canton of 'Glarus'.*

- where 25 communities were merged into three. We will consequently calculate cost curves for both sets of communities, before and after the merger.

In Switzerland, waste water catchments are not organised purely along administrative borders but depend on topographic settings. That is why we currently find three different waste water catchments, indicated with red borders in Fig. 4.3. In this paper, we only focus on the largest WWTP catchment, which we henceforth call the 'case-study catchment'.

4.2.2.2 Aggregation of urban structural units

For regional or medium-scale analysis, data aggregation is generally required to reduce computational complexity (Haggag and Ayad 2002). To run the heuristic sewer generation algorithm efficiently (Section 4.2.2.3), we choose an aggregation technique based on urban structural units (USU), as sanitation planning is closely linked to urban patterns (Spirandelli 2015, Bach et al. 2015). USU are defined as '*areas with a physignomically homogeneous character, which are marked in the built-up area by a characteristic formation of buildings and open spaces*' (Wickop 1998). With the emergence of geographical information systems, USU are increasingly used in different contexts (inter alia Osmond 2010, Wang et al. 2013, Behling et al. 2015), but have so far been rarely applied to the field of sanitation (for exceptions, see Schiller 2010, Eggimann 2013). Different approaches have been developed to classify the physiognomies of urban building which can be used to define USU (Steiniger et al. 2008, Meinel and Burgdorf 2008, Lüscher et al. 2009). To derive USU, we choose an approach based on the spatial intersection of linear urban features (street and railway networks) within the settlement area. This intersection is followed by a post-processing step in which USU containing no buildings are removed and smaller USU (<0.5 km^2) are merged with neighbouring ones. To estimate the population per USU, we disaggregate the community population data

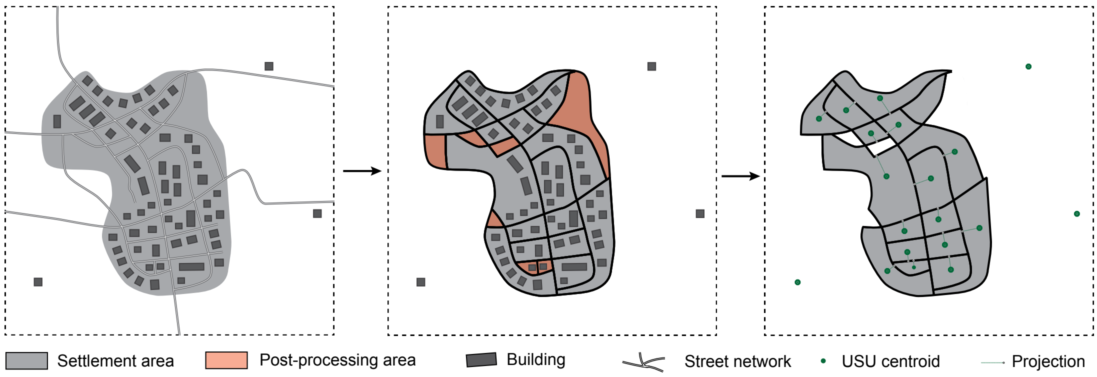


Figure 4.4: *Schematic representation of USU generation.*

according to a volumetric estimation by Lwin and Murayama (2009). The population data of the USU centroids is in a last step projected to the closest point on the street network (see Fig. 4.4).

4.2.2.3 Key cost components

For the cost calculation of the case study, we convert all local currencies to US$ using purchase power parities for the year 2014 (World Bank, 2015). All levelised costs are given in annuities (A) calculated from the net present value (NPV):

$$A = NPV \frac{q^n(q-1)}{q^n-1} \tag{4.3}$$

where q is the discount rate + 1 and n the number of years over which the infrastructure is depreciated (Crundwell 2008). We adjust on-site treatment costs to the year 2014 using conversion factors for the U.S. price index (U.S. Census Bureau 2015). We derive the various cost elements as follows (cf. Section 4.2.1.2):

Central treatment: we use typical Swiss replacement costs to estimate the large-scale WWTP costs (Fig. 4.5). As centralised costs are very unreliable for small treatment plants, we use on-site treatment costs for plants smaller than 20 population equivalents (PE).

Central transportation: in order to estimate the costs of the sewer network along the whole CR spectrum, we adapt and apply a heuristic sewer network generation algorithm developed by Eggimann et al. (2015) which is based on sewer-design principles from the real world. The adapted algorithm allows us to iteratively simulate a sewer network starting from a single connected household up to full catchment connection to a single WWTP. For in-depth explanations of the applied algorithm and the terminology used as outlined below, we refer to Eggimann et al. (2015). Compared to the original algorithm, we make three adaptations:

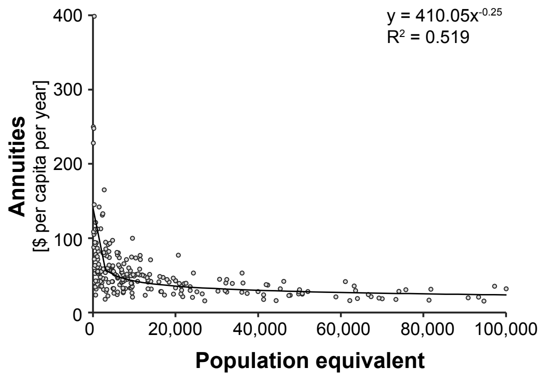


Figure 4.5: *Swiss capital and O&M expenditures for centralised treatment (VSA 2011), assuming an average lifespan of 30 years and a discount rate of 2%*

We do not consider semi-decentralised solutions but iteratively simulate an interconnected network. So there is no need to execute the merging module, as only a single WWTP exists at each iterative step. We therefore always force a sewer connection in each iteration, and thus reduce the system design module to two options.

Due to this conceptual change, the introduction of further distance weighting factors of the Prim-based expansion module yields visually more realistic sewer networks. At each iteration step, we check whether there is a local elevation or depression ('hill' or 'valley') <25m between two nodes under consideration. If so, we multiply this distance by a weighting factor d_w ($d_w = 30$). Moreover, we always weight nodes which are topographically lower by d_w ($d_w = 10$), as pumping is necessary and is to be avoided.[5] The only exception is where the nodes under consideration form edges leading to the WWTP. This is because the network position of the WWTP can be switched with the considered node so that pumping can also be avoided.

We remove the a* algorithm in the case of missing connections to street networks to reduce the computational burden, and use straight-line distance approximations instead.

Decentralised treatment: it is challenging to determine the average on-site treatment costs because a wide variety of possible system alternatives exist (Maurer et al. 2012). But even more importantly, the functional equivalence of on-site WMS is hard to operationalise. We therefore opt for a fail-safe option and include disinfection costs derived from systems based on sodium hypochlorite and UV radiation (WERF 2010). We additionally assume that further costs arise due to the need to dispose of effluent on-site, ignoring possible synergies with storm-water management systems. We estimate the average non-spatially dependent costs of on-site WMS on the basis of a selection of international cost literature considering the costs of materials, planning and installation, sludge treatment and electricity (Fletcher et al. 2007, WERF 2010, JECES 2015). The assessed treatment systems are either of class C according to DIBt (2014), or where the provision of nitrification or denitrification was not specified we classify the systems with a range as class C-D.[6]

[5] The choice of these distance weighting factors is arbitrary and based only on visual quality inspection. We consider this a valid approximation given the intention and scope of this paper and the low sensitivity of this parameter.

[6] Nutrient recovery is especially promising for on-site WMS and affects the overall economic performance. However, we do not include this analysis in view of the scope of this paper.

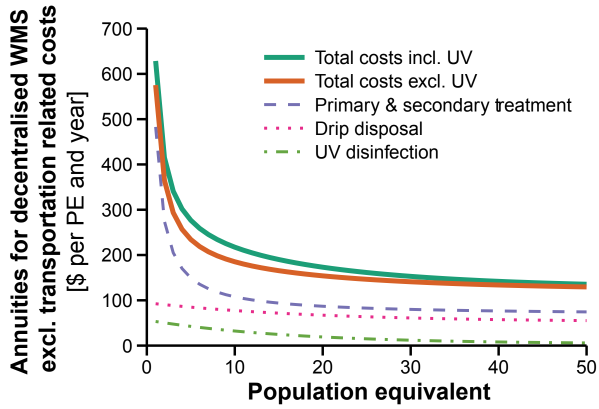


Figure 4.6: *Average treatment cost data for on-site WMS.*

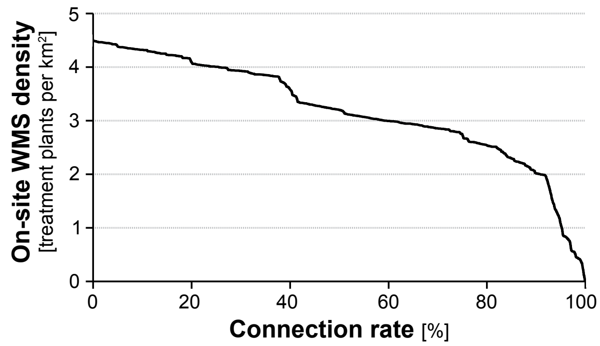


Figure 4.7: *Calculated relationship between the CR and the treatment plant density for the whole case study region as a basis for estimating local transportation-related costs.*

Fig. 4.6 shows the total cost function, including the costs of a drip disposal system and for a UV disinfection unit (WERF 2010) which fall in line with other cost estimations for Switzerland (cf. Abegglen 2008).

Decentralised transportation: To derive transportation-related costs for on-site WMS, we use model-based cost data from Eggimann et al. 2016, who provide a cost-density relationship (treatment plants per km^2) at regional level for a Swiss case study. The authors model a cost-density relationship in a two-dimensional geometrical space by means of a heuristic routing algorithm. To estimate the transportation costs in relation to

Table 4.1: *Overview of cost scenario assumptions for all key cost elements. Standard scenario values are given in bold.*

Cost element	Description	Unit	Scenario assumptions
$C^{cen}_{treatment}$	Assume idle capacity	%	**0**, 50, 100
$C^{cen}_{transport}$	Different minimum sewer slope ($f_{minslope}$) for running the sewer network generation algorithm	%	0.1, **1**, 1.5
$C^{dec}_{treatment}$	Assumed on-site WMS dimension	PE	5, **10**, 15
$C^{dec}_{transport}$	Systematic cost variatio	%	-20, **0**, +20

the treatment plant density on a local scale, we derive the relationship between the CR and the on-site WMS density over all CRs for the whole case-study region (Fig. 4.7).

4.2.2.4 Sensitivity analysis

To evaluate the model sensitivity, we calculate three different cost functions for each key cost element by systematically varying the underlying assumptions (Table 4.1). A systematic combination of all resulting cost functions yields 81 different scenarios. With this approach, we aim to produce diversity in order to indicate sensitivity rather than statistical representativeness.

4.3 Results

4.3.1 Cost curve configurations

Appendix A gives all standard parameter calculations of the former communities of Glarus. Fig. 4.8 shows the results over all cost scenarios with respect to the different OCR. We find very diverse OCR at the former communal level with broad sensitivity ranges resulting from the cost scenarios. For today's more dispersed southern community 'Süd', we find lower OCRs than for the more urbanised communities of 'Nord' or 'Glarus'.

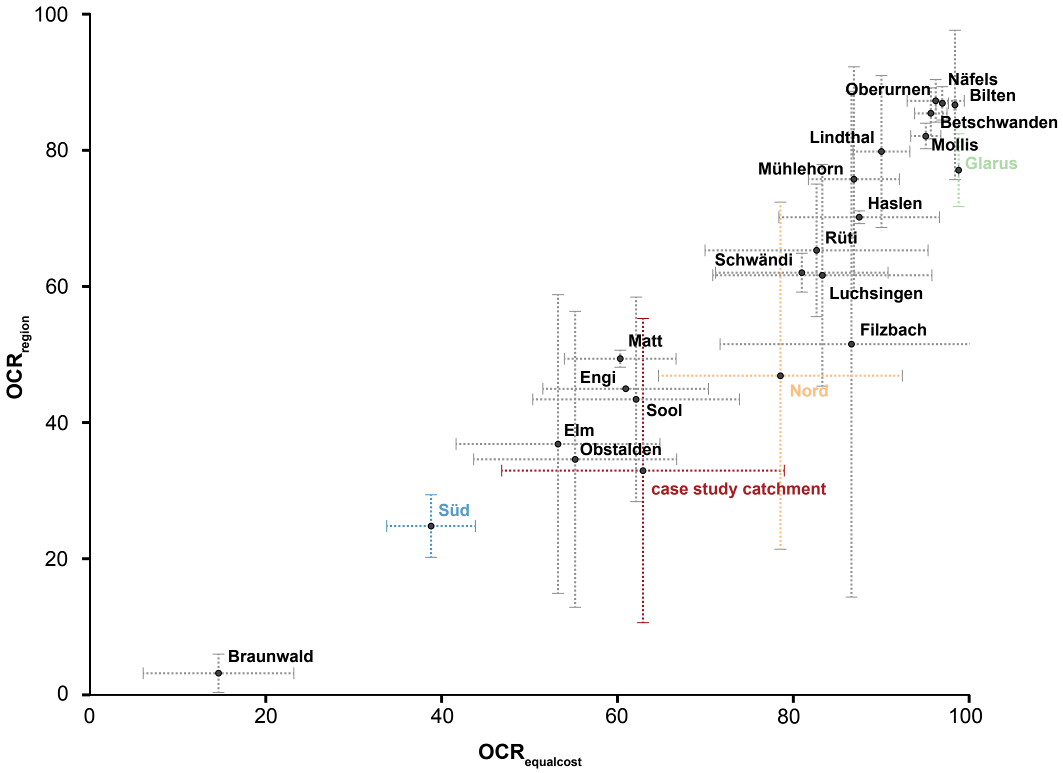


Figure 4.8: *Visualisation of the cost scenario calculations for all former (black) and current merged communities of the case study region 'Glarus' (coloured), including the case study catchment. The scenario sensitivity is indicated by error bars representing one standard deviation.*

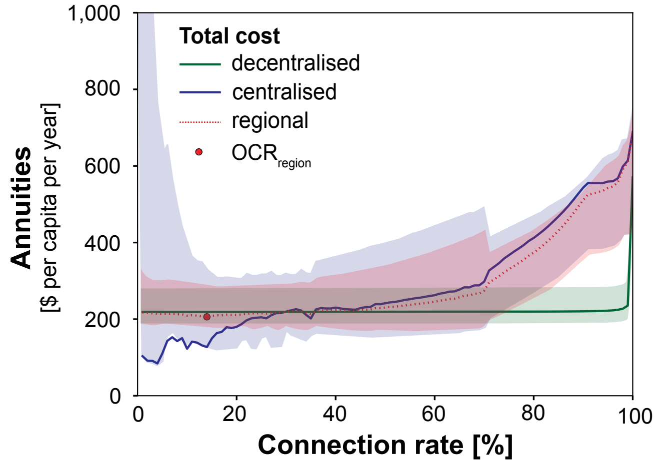


Figure 4.9: *Thick lines show standard parameter calculations, shaded areas indicate scenario uncertainties (maximum extent over all 81 scenarios).*

The detailed cost curve configuration of the case study catchment in Glarus is given in Fig. 4.9. We notice that the OCR$_{region}$ and the OCR$_{equalcost}$ are at very low CR of around 0.2 and 0.4 respectively. However, the total regional cost curve is more or less horizontal until a CR of around 0.6 - 0.7. The standard parameter calculation is thus very sensitive to small changes of any single cost component.

4.3.2 Typology

We can identify basic cost-shape behaviours on the basis of the configurations of all the communities. This enables us to build a typology that distinguishes between three major configurations (see Fig. 4.10 for typical examples):

- *Type A*: This cost curve configuration type has no OCR$_{equalcost}$, and centralised WMS costs are typically lower for all CR. The OCR$_{region}$ is typically very high.
- *Type B*: For this type, we do not find a distinct OCR$_{equalcost}$ because the intersection point is highly sensitive to cost-curve changes due to a more or less horizontal total regional cost curve (we may find multiple cost curve intersections). The OCR$_{region}$ is typically in the middle CR range. On-site WMS costs only become noticeably expensive at very high CR.
- *Type C*: For this type, we find distinctive OCR$_{equalcost}$ and OCR$_{region}$ with clear cost differences. Typically, we observe a distinctive exponential increase of the centralised costs at relatively low CR, leading to low OCRs.

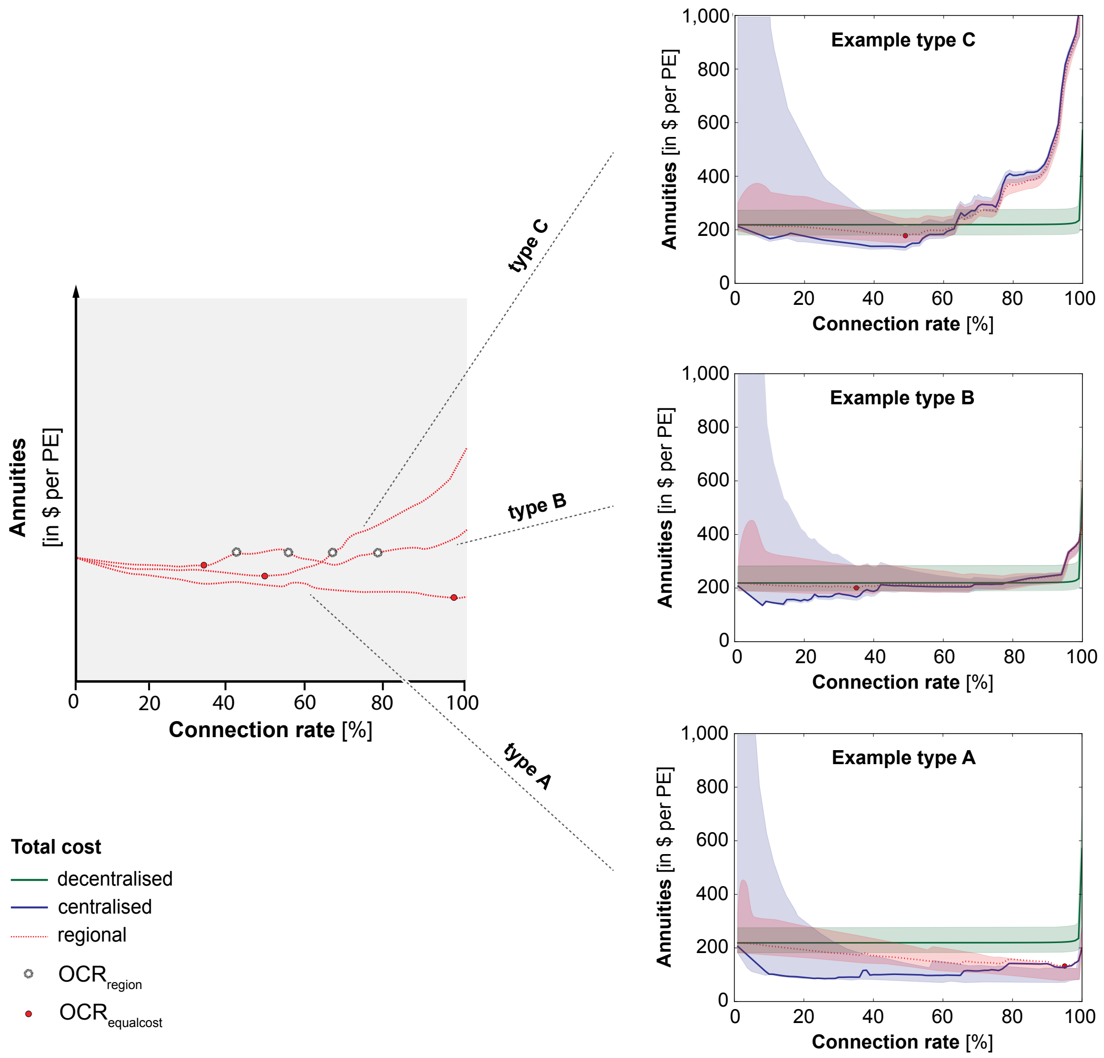


Figure 4.10: *Cost curve configuration typology with examples. For each example type, standard parameter runs (thick line) and scenario uncertainties are provided (cf. Figure 4.9).*

4.4 Discussion

We now reflect critically on the case study application and our framework in general. We then elaborate the institutional conditions under which the different OCRs could be realised. Finally, we identify potential research needs.

4.4.1 Case study application and OCR typology

The case study application confirms that we can indeed identify the conceptually outlined OCRs on a real example. We find very different OCR_{region} and $OCR_{equalcost}$ for the former and merged communities depending on the local geography, ranging from very low to very high CR. For example, we note that the WWTP catchment along the new community 'Süd' is unsuitable for a large centralised WMS. We see that the OCR depends on the chosen scale and catchment boundary, which is to be expected, as the topographic characteristics also depend on the chosen system boundaries. Specific characteristics of the various catchments enabled us to derive a typology of cost curves. However, the boundaries of this typology are fuzzy, so it only represents a broad classification.

An important finding for our case study catchment is that both the calculated OCRs are lower than the $CR_{present}$. We argue that this is because these sewers were not constructed primarily from a cost optimisation point of view and the regulators have often introduced a mandatory connection rule in order to force a higher number of households to connect to the sewers than a direct cost comparison would suggest. The reason for these regulations often lie in the argument that centralised systems are easier to control than a myriad of on-site WMS, or that the latter cannot cope in terms of treatment performance criteria (inter alia Moelants et al. 2008, Buchanan 2014). However, for this paper we assume that neither of these arguments will be valid if on-site WMS are properly designed and if appropriate business models are installed to run them. We thus presume that the role of institutions responsible for applying the mandatory connection rule or investment subsidies for centralised WMS explain why $CR_{present}$ is much higher than $OCR_{equalcost}$ or OCR_{region} in countries like Switzerland (cf. Eggimann et al. 2015).

High uncertainties result from the various cost scenarios. This is largely due to the cost curve configurations: for many communities (including the case study catchment estimated here), the total regional cost curve is rather flat and the cost values of OCR_{region} and $OCR_{equalcost}$ are thus very close. Consequently, only minor cost differences would lead to very different intersection points on the cost curve. This suggests that a focus on $OCR_{equalcost}$ might be a viable option if OCR_{region} is hard to implement.

Finally, territorial reforms are a challenge in water governance (OECD 2015b). The organisational centralisation in Glarus is in line with the general tendency to centralisation throughout Switzerland. We argue that this creates an opportunity to reach lower-cost CR because larger organisations are likely to develop higher professional competencies to run both centralised and on-site WMS (Maurer et al. 2012b). This is especially interesting in the case of on-site treatment, where larger contracts result in more standardised and professional operation and management.

4.4.2 The institutional and organisational setting of OCR

In order to decide which of the two candidate OCRs is more likely to be implemented, we have to take a closer look at the incentive structures and regulatory arrangements in the specific regions:

i.) To reach the OCR_{region}, a central decision maker would have to determine the total regional cost curve and identify the lowest cost point. In most empirical cases, this will coincide with the lowest cost point of the total centralised cost curve. It would therefore be sufficient to require central operators to connect new households as long as their total average cost curve decreases.[7] Beyond that point, households would have to seek services from companies that offer on-site alternatives. As a consequence, users connected to the centralised WMS would have lower costs than those serviced by on-site WMS. The rationale for this arrangement is that the total amount of money spent in the region would then be lowest. However, this solution would imply that users connected to the centralised WMS would pay substantially lower tariffs for their WMS services than those that have to rely on on-site solutions.[8] Such price differences could lead to political protests. This problem could be circumvented if the provision of WMS services for the whole region was delegated to a monopoly provider who would be obliged to charge households equal tariffs while minimizing the overall costs in the region. One way to implement such a solution would be for a public utility to build up equal professional competence in both centralised and decentralised WMS and be subjected to tight price regulation. Alternatively, the OCR_{region} could be reached by a private monopoly operator who bids for a long-term service contract through a public call for tenders (Demsetz 1968).

ii.) However, it may not be feasible to reach the OCR_{region} under specific conditions: there may be strong political preferences in the region for individual households to choose their service provider freely, and monopoly providers (public or private) may meet with resistance. Moreover, it may prove difficult to build up professional competencies in both centralised and decentralised WMS within a single organisation. In these situations, the $OCR_{equalcost}$ might be a second-best option, as the costs would be the same for all households while various organisations could compete to supply them. The $OCR_{equalcost}$ could be reached if a public or private organisation running the centralised system were required to offer its services at average cost and would be

[7] Following this logic, the central operator may not maximise his profit and consequently needs to be regulated, as he would otherwise connect too many households (the profit maximum lies somewhere between both OCR).

[8] In order not to complicate matters, we assume that utilities would be able to charge tariffs on a cost-plus basis.

prohibited from turning down customers. Households would be free to choose either to connect to a sewer or to accept services from one of the potentially many suppliers of on-site WMS. In this case, the centralised system would expand to the point where the average cost curves of the two systems intersect, i.e. the $OCR_{equalcost}$.

We can deduce from the general cost-curve characteristics of our case study that the following relationship holds for countries with very high $CR_{present}$: $OCR_{region} < OCR_{equalcost} < CR_{present}$. The first inequality is given by the shape of the cost curves and is generic. The second is very likely to hold in countries which have installed regulations such as mandatory connection rules. Otherwise, competition would likely lead to market shares that are close to or at around the $OCR_{equalcost}$.

We may summarise our framework for calculating the total regional cost for hybrid WMS systems as follows: the shape of the type C cost curve indicates two potential OCR that would be superior to the present CR. However, which of these OCR is actually reached depends on the role specification of households, the central system operator, the on-site suppliers and the regulator. Getting away from current mixing ratios will therefore depend on comprehensive reforms (including organisations and regulations) and cannot be considered purely as a matter of cost.

4.4.3 Critical reflection and research needs

The full cost assessment for regional WMS represents at least a first step towards determining more sustainable WMS services. However, it is not enough merely to assess the costs.

Most cost assumptions relating to the costs of on-site treatment were chosen on the conservative side in this paper (including for disinfection and on-site infiltration). However, these may be subject to considerable changes in the future, for instance if economies of scale could be reaped in manufacturing (Adler 2007, Dahlgren et al. 2013). On the other hand, assumptions about effluent disposal would require a more sophisticated analysis including storm water evacuation. As far as transportation costs are concerned, lacking economies of scale and the challenges involved in establishing fully functional O&M schemes are usually considered as the key disadvantages of on-site WMS (cf. Eggimann et al. 2016). However, off-grid infrastructure systems also possess specific advantages, although these are hard to express in monetary terms: the independence from a sewer network increases the flexibility to respond to socio-economic or technological boundary conditions (Panebianco and Pahl-Wostl 2006, Hug et al. 2010). It also reduces interdependence-related disruptions (Rinaldi et al.

2001) and lessens the potential environmental impact in case of failure of a single plant, whether due to malfunctions, earthquakes (Hamada 2014) or terrorism (Panebianco and Pahl-Wostl 2006). Centralised and on-site WMS thus offer unique strengths and weaknesses which are often intangible and difficult express in to monetary terms (cf. Gikas and Tchnobanoglous 2009, Libralato et al. 2012, Larsen et al. 2013, Vousvouras 2013). However, the quantification of non-monetary advantages or disadvantages goes far beyond this study as it would require a research approach of much greater scope (cf. Morera et al. 2015, Arora et al. 2015, Naik and Stenstrom 2016).

With the aid of the framework presented here, we can address the question of the degree to which on-site WMS can be considered as substitutes from an economical point of view. However, we refer to the literature (inter alia Larsen et al. 2013, Libralato et al. 2012) concerning the key assumption as to whether on-site WMS can be considered as functionally equivalent from a technological point of view.

In this study, we assume stable context conditions even though many exogenous factors affect infrastructure planning, such as changing public goals or environmental concerns (Hansman et al. 2006). Furthermore, WMS are exposed to different long-term dynamics (e.g. population, role of industry, water consumption trends), making it very challenging to plan optimal systems (Dominguez and Gujer 2006). However, this study provides valuable insights into changing population and settlement dynamics related to sanitation costs: we showed that different catchments result in diverse characteristic cost configurations, which gives an indication of what cost configurations may look like and evolve for future projected catchments. For instance, let us assume an anticipated increase in settlement area together with sprawling tendencies of the settlement distribution for an urban catchment classified as type A. For such a case, we might expect a cost configuration shift from type A towards type C. On the other hand, for catchments classified as type A, urban infill or settlement shrinking in rural areas (Siedentop and Fina 2010) shifts the cost curve from type A towards type C. A final assumption of the framework outlined here is that the basic choice for households in a particular region is either between a fully centralised or a small-scale on-site WMS. We believe that in reality the choice is indeed often limited to these basic two options, namely either to connect to a large centralised WWTP or to select small-scale package treatment plants.

4.5 Conclusion

In this paper, we have prepared the way towards achieving a cost-based identification of lowest cost connection rates (CR) in a given region. We thus contribute to the broader debate about sustainable CR with a cost analysis over the whole CR continuum. In particular, we identify a potentially optimal CR (OCR_{region}) from a total regional cost point of view, and a second-best CR ($OCR_{equalcost}$) which may be easier to implement under specific institutional and organisational conditions.

The framework presented here suggests that the OCR_{region} may be achieved if the operator of the centralised WMS is required to expand his service as long as his average costs decrease. Alternatively, the OCR_{region} is reached if a single operator runs both WMS alternatives and tariffs are regulated either explicitly or through a call for tenders relating to service contracts. The second-best $OCR_{equalcost}$ could be reached if individual households can choose freely between central and on-site WMS. A potentially intermediary form would be for households beyond the OCR_{region} to be charged tariffs proportional to the costs for a sewer connection on the basis of actual household connection costs, thus increasing their incentive to choose on-site WMS. In our case study, we find relatively small cost differences between the two OCR, which suggests that opting directly for the $OCR_{equalcost}$ is an advisable option. We argue that neither OCR can be reached without regulating the centralised WMS and introducing adequate policy measures.

We optimise CR by building on long-term average costs, thus assuming that the context conditions remain static in the long run. In further elaborations of the framework, it would make sense to include dynamic considerations (e.g. changing settlement patterns or population dynamics).

Finally, we believe that a holistic consideration is needed in view of the complexity of the question of cost-efficient CR for sustainable urban water management. We conclude that this discussion cannot be separated from analyses of the respective organisational, institutional and regulatory arrangements in a region and argue for a co-evolution of technological advances in on-site WMS with the prevailing institutional and organisational arrangements.

Acknowledgements

Our special thanks go to Ivana Logar, Roy Brouwer and Matteo Mattmann for their suggestions to improve the manuscript. We thank Rachel Barrett for her help with the data preparation and Richard Michell for the proofreading. We also thank the Canton of Glarus for their data provision. ■

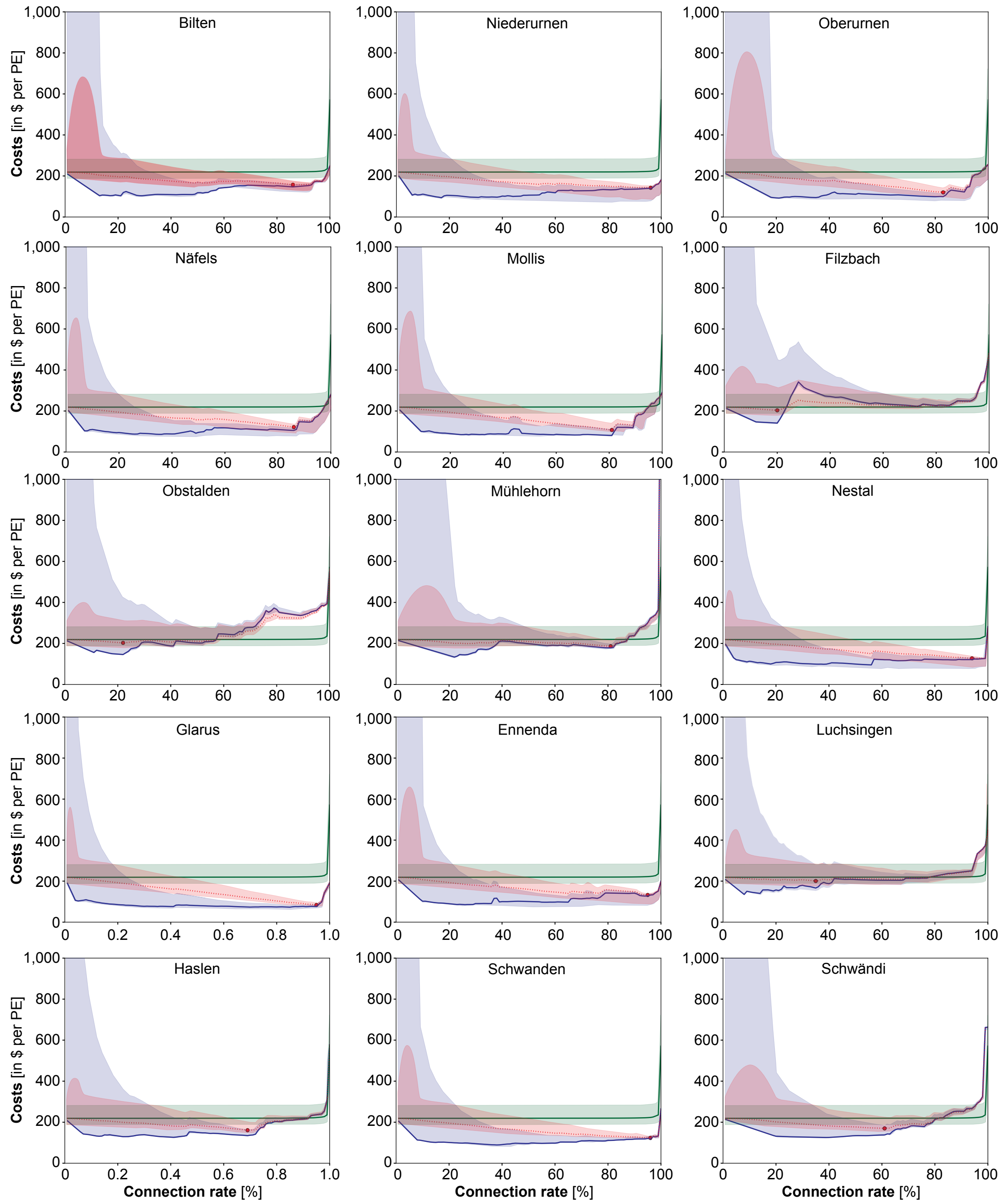


Figure A.4.1: *Communal cost curve configurations. The results show standard parameter calculations, including scenario uncertainties indicated by shaded areas.*

4.6 References

Abd El Gawad, H., Butter, J., **1995**. Clustering of towns and villages for centralized wastewater treatment. *Water Science and Technology*, 32(11), 85–95.

Abegglen, C.K., **2008**. Membrane bioreactor technology for decentralized wastewater treatment and reuse. Dissertation, ETH Zürich, Switzerland.

Adams, B.J., Dajani, J.S., Gemmell, R.S., **1972**. On the centralization of wastewater treatment facilities. *Water Resources Bulletin,* 8(4), 669–678.

Adler, C., **2007**. Market potential of a membrane based wastewater treatment plant for decentralised application in China. Master Thesis. Eawag, Dübendorf, Switzerland.

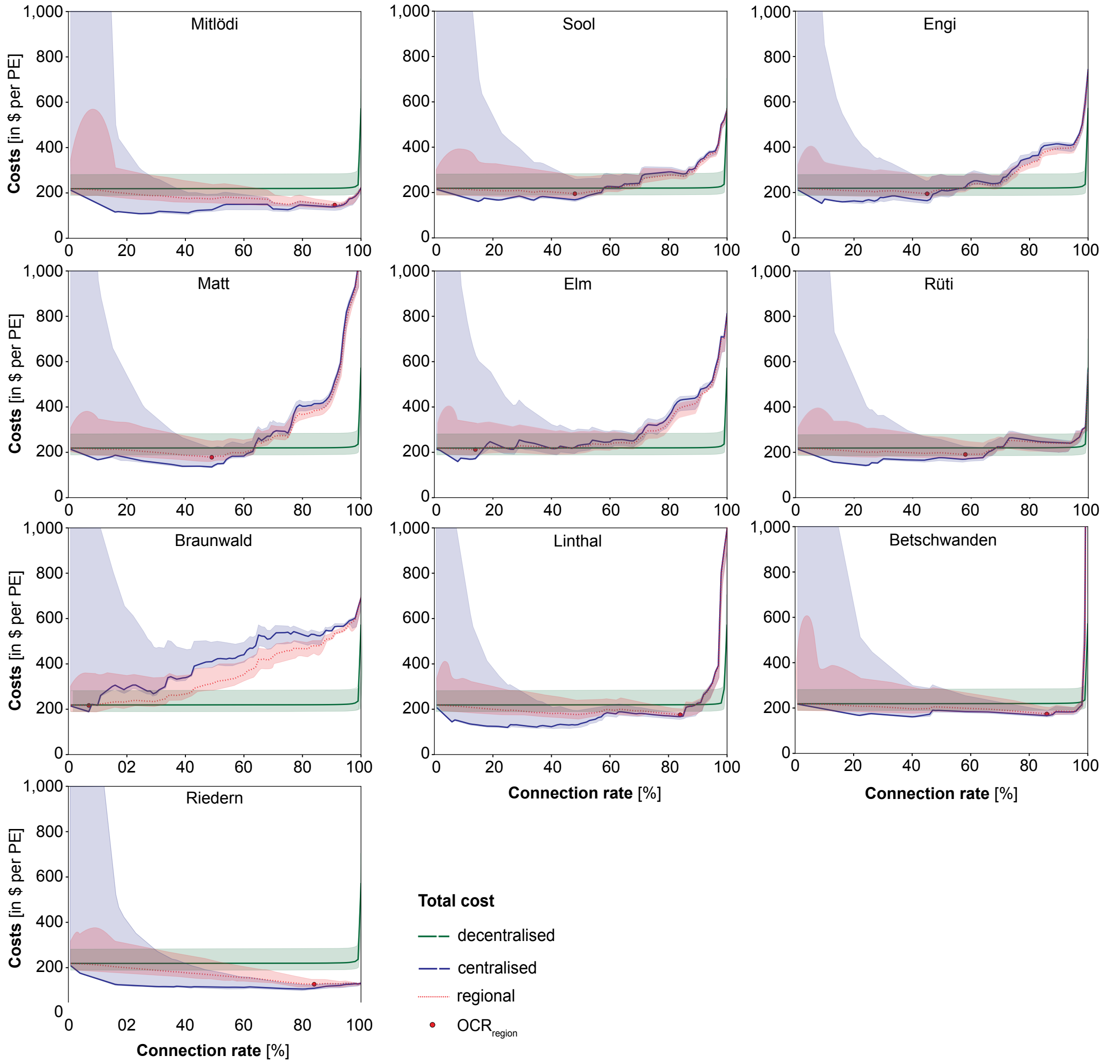


Figure A.4.1: *(continued).*

Arora, M., Malano, H., Davidson, B., Nelson, R., Geroge, B., **2015**. Interactions between centralized and decentralized water systems in urban context: A review. *WIREs Water 2015(2)*, 623–634.

Arthur, W.B., **1989**. Competing technologies, increasing returns, and lock-in by historical events. *The Economic Journal*, 99(394) 116–131.

Bach, P.M., Staalesen, S., McCarthy, D.T., Deletic, A., **2015**. Revisiting land use classification and spatial aggregation for modelling integrated urban water systems. *Landscape and Urban Planning*, 143, 43–55.

Behling, R., Bochow, M., Foerster, S., Roessner, S., Kaufmann, H., **2015**. Automated GIS-based derivation of urban ecological indicators using hyperspectral remote sensing and height information. *Ecological Indicators*, 48, 218–234.

Buchanan, J.R., **2014**. Decentralized wastewater treatment. Book chapter, volume three, 'Remediation of Polluted Water', in the Series 'Comprehensive Water Quality and Purification. Elsevier, Oxford, UK.

Crundwell, F.K., **2008**. Finance for engineers. Springer, London, UK.

Dahlgren, E., Göçmen, C., Lackner, K., Vvan Ryzin, G., **2013**. Small modular infrastructure. *The Engineering Economist*, 58(4), 231–264.

Daigger, G.T., **2007**. Wastewater Management in the 21st Century. *Journal of environmental engineering*, 133(7), 671–680.

Demsetz, H., **1968**. Why Regulate Utilities? *Journal of Law and Economics*, 11(1), 55–65.

DIBt, **2014**. Approval guidelines for small wastewater treatment plants (Zulassungsgrundsätze Kleinkläranlagen). Berlin, Germany.

Dominguez, D., Gujer, W., **2006**. Evolution of a wastewater treatment plant challenges traditional design concepts. *Water Research*, 40(7), 1389–1396.

Dominguez, D., **2008**. Handling future uncertainty: Strategic planning for the infrastructure sector. Dissertation, ETH Zürich, Switzerland.

Downing, P.B., **1969**. The economics of urban sewage disposal. Praeger, New York, NY, USA.

Eggimann, S., **2013**. Potential for a sustainability transition of decentralised wastewater systems for Switzerland on the level of urban structural units (Potenzial für eine Nachhaltigkeits-Transition dezentraler Abwassersysteme auf Strukturtypebene in der Schweiz). Master Thesis, University of Zürich, Switzerland (in German).

Eggimann, S., Truffer, B., Maurer, M., **2015**. To connect or not to connect? Modelling the optimal degree of centralisation for wastewater infrastructures. *Water Research,* 84, 218–231.

Eggimann, S., Truffer, B., Maurer, M., **2016**. Economies of density for on-site waste water treatment. *Water Research*, 101, 476–489.

Eetnier, C., Nelson, V., Pinkham, R., **2000**. Economics of decentralised wastewater treatment systems: Direct and indirect costs and benefits. In: *National research needs conference proceedings: Risk-based decision making for onsite wastewater treatment.* St. Louis, MO, USA.

Fane, A.G., Fane, S.A., **2005**. The role of membrane technology in sustainable decentralized wastewater systems. *Water Science and Technology*, 51(10), 317–325.

Fletcher, H., Mackley, T., Judd, S., **2007**. The cost of a package plant membrane bioreactor. *Water Research*, 41(12), 2627–2635.

Fuenfschilling, L., Truffer, B., **2014**. The structuration of socio-technical regimes—Conceptual foundations from institutional theory. *Research Policy*, 43(4), 772–791.

Gikas, P., Tchobanoglous, G., **2009**. The role of satellite and decentralized strategies in water resources management. *Journal of Environmental Management*, 90(1), 144–152.

Guo, T., Englehardt, J.D., **2015**. Principles for scaling of distributed direct potable water reuse systems: A modeling study. *Water Research*, 75, 146–163.

Haggag, M.A, Ayad, H.M., **2002**. The urban structural units method: a basis for evaluating environmental prospects for sustainable development. *Urban Design International*, 7(2), 97–108.

Hamada, M. (eds), **2014**. Critical urban infratructure handbook. CRC Press, Boca Raton, FL, USA.

Hamilton, B.A., Pinkham, R.D., Hurley, E., Watkins, K., Lovins, A.B., Magliaro, J., Etnier, C., Nelson, V., **2004**. Valuing decentralised wastewater technologies: A catalog of benefits costs and economic analysis techniques. Rocky Mountain Institute, Snowmass, CO, USA.

Hansman, R.J., Magee, C., Neufville, R. DE., Robins, R., Roos, D., **2006**. Research agenda for an integrated approach to infrastructure planning, design and management. *International Journal of Critical Infrastructures*, 2(2/3), 146–159.

Hug, T., Dominguez, D., Maurer, M., **2010**. The cost of uncertainty and the value of flexibility in water and wastewater infrastructure planning. Proceedings of the Water Environment Federation, Cities of the Future/ Urban River Restoration, 487–500.

Jeces, **2015**. Personal communication with Satoru Takahashi, Japan Education Center of Environmental Sanitation, 22 October 2015.

Kaundinya, D.P., Balachandra, P., Ravindranath, N.H., **2009**. Grid-connected versus stand-alone energy systems for decentralized power—A review of literature. *Renewable and Sustainable Energy Reviews*, 13(8), 2041–2050.

Kiparsky, M., Ssedlak, D.L., Thompson, B.H., Truffer, B., **2013**. The innovation deficit in urban water: The need for an integrated perspective on institutions, organisations, and technology. *Environmental Engineering Science*, 30(8), 395–408.

Knops, H., **2008**. The impact of technical characteristics of network industries upon the governance of infrastructure adequacy. In: Arts, G., Dicke, W., Hancher, L. (eds). *New perspectives on investment in infrastructures*. Amsterdam University Press, Amsterdam, The Netherlands.

Larsen, T.A., Udert, K.M., Lienert, J. (eds), **2013**. Source separation and decentralization for wastewater management. IWA Publishing, London, UK.

Lee, E.J., Criddle, C.S., Bobel, P., Freyberg, D.L., **2013**. Assessing the scale of resource recovery for centralized and satellite wastewater treatment. *Environmental Science and Technology*, 47(19), 10762–10770.

Libralato, G., Ghirardini, A.V., Avezzù, F., **2012**. To centralise or to decentralise: An overview of the most recent trends in wastewater treatment management. *Journal of Environmental Management,* 94(1), 61–68.

Llieberherr, E., Truffer, B., **2015**. The impact of privatization on sustainability transitions: A comparative analysis of dynamic capabilities in three water utilities. *Environmental Innovation and Societal Transitions*, 15, 101–122.

Llieberherr, E., Fuenfschilling, L., **2016**. Neoliberalism and sustainable urban water sectors: A critical reflection of sector characteristics and empirical evidence. *Environment and Planning C: Government and Policy*, 1–16.

Lüscher, P., Weibel, R., Burghardt, D., **2009**. Integrating ontological modelling and Bayesian inference for pattern classification in topographic vector data. *Computers, Environment and Urban Systems*, 33, 363–374.

Lwin, K., Murayama, Y., **2009**. A GIS approach to estimation of building population for micro-spatial analysis. *Transactions in GIS*, 13(4), 401–414.

Marlow, D.R., Moglia, M., Cook, S., Beale, D.J., **2013**. Towards sustainable urban water management: A critical reassessment. *Water Research*, 47(20), 7150–61.

Maurer, M., Rothenberger, D., Larsen, T.A., **2006**. Decentralised wastewater treatment technologies from a national perspective: at what cost are they competitive? *Water Science and Technology: Water Supply*, 15(6), 145–154.

Maurer, M., Wolfram, M., Herlyn, A., **2010**. Factors affecting economies of scale in combined sewer systems. *Water Science and Technology*, 62(1), 36–41.

Maurer, M., Bufardi, A., Tilley, E., Zurbrügg, C., Truffer, B., **2012**. A compatibility-based procedure designed to generate potential sanitation system alternatives. *Journal of Environmental Management*, 104, 51–61.

Maurer, M., Chawla, F., Horn, J., Stauffer, P., **2012**B. Water Sanitation 2025 (Abwasserentsorgung 2025 in der Schweiz). Schriftreihe der eawag, Dübendorf, Switzerland (in German).

Meinel, G., Burgdorf, M., **2008**. Automatic derivation of urban land use pattern and integration in a Geographical Information System. Forschungen, Bundesministerium für Verkehr, Bau und Stadtentwicklung, Bundesamt für Bauwesen und Raumordnung, Bonn, Germany (in German).

Moelants, N., Janssen, G., Smets, I., Van Impe, J., **2008**. Field performance assessment of onsite individual wastewater treatment systems. *Water Science and Technology*, 58(1), 1–6.

Morera, S., Comas, J., Poch, M., Corominas, L., **2015**. Connection of neighboring wastewater treatment plants: economic and environmental assessment. *Journal of Cleaner Production*, 90, 34–42.

Naik, K.S., Stenstrom, M.K., **2016**. A feasibility analysis methodology for decentralized wastewater systems – Energy-efficiency and cost. *Water environment research*, 88(3), 201–209.

Panebianco, S., Pahl-Wostl, C., **2006**. Modelling socio-technical transformations in wastewater treatment—A methodological proposal. *Technovation*, 26(9), 1090–1100.

Poustie, M.S., Deletic, A., Brown, R.R., Wong, T., De Haan, J., Skinner, R., **2014**. Sustainable urban water futures in developing countries: the centralised, decentralised or hybrid dilemma. *Urban Water Journal*, 1–16.

Rinaldi, S.M., Peerenboom, J.P., Kelly, T.K., **2001**. Identifying, understanding, and analyzing critical infrastructure interdependencies. *IEEE Control Systems Magazine*, 21(6), 11–25.

Schiller, G., **2010**. Cost evaluation of the adaptation of wastewater treatment systems under shrinkage (Kostenbewertung der Anpassung zentraler Abwasserentsorgungssysteme bei Bevölkerungsrückgang). IÖR Schriften, 51, Rhombos (in German).

Sedlak, D., **2014**. Water 4.0: The past, present, and future of the world's most vital resource. Yale University Press, New Haven & London.

Shahabi, M.P., McHugh, A., Anda, M., Ho, G., **2015**. Comparative economic and environmental assessments of centralised and decentralised seawater desalination options. *Desalination*, 376, 25–34.

Siedentop, S., Fina, S., **2010**. Urban sprawl beyond growth: the effect of demographic change on infrastructure costs. *Flux (n° 79-80)*, 90–100.

Singh, N.K., Kazmi, A.A., Starkl, M., **2015**. A review on full-scale decentralized wastewater treatment systems: techno-economical approach. *Water Science and Technology*, 71(4), 468–478.

Spirandelli, D., **2015**. Patterns of wastewater infrastructure along a gradient of coastal urbanization: A study of the Puget Sound Region. *Land*, 4(4), 1090–1109.

Steiniger, S., Burghardt, D., Weibel, R., **2008**. An approach for the classification of urban building structures based on discriminant analysis techniques, *Transactions in GIS,* 12(1), 31–59.

Tchobanoglous, G., Leverenz, H., **2013**. The rationale for decentralization of wastewater infrastructure. In: Larsen, T. A., Udert, K. M., Lienert, J. (eds). *Source separation and decentralization for wastewater management.* IWA Publishing, London, UK.

Townend, C.B., **1959**. The economics of the disposal of sewage and trade effluents. *Bulletin of the World Health Organization*, 20, 535–562.

Truffer, B., Störmer, E., Maurer, M., Ruef, A., **2010**. Local strategic planning processes and sustainability transitions in infrastructure sectors. *Environmental Policy and Governance*, 20, 258–269.

UN, **2015**, Environmental Indicators. Inland Water Resources, Wastewater. Archived by WebCite® http://www.webcitation.org/6gQ6z23bk

U.S. Census Bureau, **2015**. Construction Price Indexes. Archived by WebCite® http://www.webcitation.org/6d6sN7StR

OECD, **2010**. Pricing water resources and water and sanitation Services. OECD Publishing, London, UK.

OECD, **2015**. Environment at a Glance 2015: OECD Indicators. OECD Publishing, Paris, France.

OECD, **2015**B. Water and cities. Ensuring sustainable futures. OECD Studies on Water. OECD Publishing, Paris, France.

O'Flaherty, B., **2005**. City economics. Harvard University Press, USA.

Osmond, P., **2010**. The urban structural unit: towards a descriptive framework to support urban analysis and planning. *Urban Morphology*, 14(1), 5–20.

Vousvouras, C.A., **2013**. Large-scale, small-scale, and hybrid water utilities in cities of the developing world: The impact of scale beyond "economies of scale". Dissertation, University of St. Gallen, Switzerland.

VSA, **2011**. Costs and performances of sewage disposal (Kosten und Leistungen der Abwasserentsorgung). Verband Schweizer Abwasser- und Gewässerschutzfachleute, Glattbrugg, Switzerland (in German).

Wang, H.F., Qiu, J.X., Breuste, J., Friedman, R.C., Zhou, W.Q., Wang, X.K., **2013**. Variations of urban greenness across urban structural units in Beijing, China. *Urban Forestry and Urban Greening*, 12(4), 554–561.

WEF, **2010**. Global Risks 2010: A Global Risk Network Report. A Global Risk Network Report. Geneva, Switzerland.

WERF, **2010**. Performance and cost of decentralised unit processes. Archived by WebCite® http://www.webcitation.org/6d6qEmdU6.

Wickhop, E., **1998**. Environmental quality targets for urban structural units in Leipzig with a view to sustainable urban development. In: Breuste, J., Feldmann, H., Uhlmann, O. (eds). *Urban Ecology*. Springer, Berlin, Germany.

World Bank, **2015**. PPP conversion factor. International comparison program database. Available at: http://data.worldbank.org/

Zvoleff, A., Kocaman, A.S., Huh, W.T., Modi, V., **2009**. The impact of geography on energy infrastructure costs. *Energy Policy*, 37(10), 4066–4078.

Chapter 5

General discussion

In this chapter, the most important limitations and potential research needs are discussed (Section 5.1), followed by recommendations for policy and practice based on the findings of this dissertation (Section 5.2).

5.1 Limitations and research needs

The question of central or decentral wastewater treatment can be addressed with varying degrees of complexity and with very different thematic foci. Given the broadness of the overarching research question, it is inevitable that various aspects are neglected or simplified in this thesis. A non-exhaustive selection of key research areas and shortcomings related most closely to this thesis is provided in the following sections. These shortcomings constitute recommended fruitful strands of further research in these areas in order to refine our understanding and assessment of more sustainable connection rates. The order of presentation does not reflect any order of priority.

5.1.1 Integrated planning and multi-objective optimisation

Planning regional sanitation systems has long been formulated as a classic multi-objective optimisation problem (inter alia de Melo and Câmara 1994). Multi-objective approaches are necessary to determine the ODC in an encompassing way as various criteria are relevant for planning WMS (inter alia Ho 2005, Weber et al. 2007, Arora et al. 2015). Different criteria can be used to assess and compare system adequacy: these may be *economic* (e.g. total costs), *environmental* (e.g. effluent quality with respect to micro-pollutants, dissolved oxygen, phosphorus or ammonia), *social* (e.g. protection of public health) or *technical* (e.g. system reliability) (see Table 5.1). To find appropriate sanitation systems among the manifold alternatives appropriate decision support methodologies must be developed to weight these different criteria, such as multi-criteria decision analysis (inter alia Maurer et al. 2012, McConville et al. 2014, Poustie et al. 2014). The focus within this thesis was placed on economic assessments. As regards the cost model, however, different degrees of complexity can be applied when comparing centralised and decentralised WMS. From an economic point of view, additional elements such as resource recovery, wastewater recycling or reuse are increasingly expressed in monetary terms and included in the (cost) analysis and optimisation with respect to scale (inter alia Liang et al. 2010, Suriyachan et al. 2012, Naik and Stenstrom 2016, Cornejo et al. 2016). Clearly, an almost unlimited number of possible further cost elements could be considered (e.g. cost of traffic diversion whilst renewing sewers etc.). However, to minimise the overall economic costs is only one (however important) criterion among the many required for successful wastewater infrastructures. Integrating more than purely economic criteria with integrated modelling procedures is a highly

Table 5.1: *Collection of possible categories and criteria which can be used for multi-objective evaluations. The table is partly based on Zeferino et al. (2010), Bradley et al. (2002), Scholten (2013), Ho (2005).*

Economic	Environmental	Social	Technical
Total cost	Effluent quality	Public health	Reliability
Fixed cost	Resource use	Acceptance	Flexibility/adaptability
Variable cost	Resource recovery/recycling	Accessibility, equality	Durability
Externalities	Emissions	Aesthetics	Performance
Affordability	Transport requirement	Risks	Interdependence
…	Ground and surface water protection	Social inclusion	Resource efficiency
		Intergenerational equity	Space requirement
	…	…	…

relevant current research area (inter alia Bach et al. 2014): thus Morera et al (2015) present a more integrated approach which combines economic and environmental aspects by considering minimum river flow criteria when treatment plants are merged. Lienhoop et al. (2014) use cost-benefit analyses and include non-monetary benefits (especially environmental criteria) in their case study overview, and Haghighi and Bakhshipour (2015) take reliability criteria (e.g. clogging performance) into consideration. To achieve more sustainable infrastructures in our society, more systematic attention should also be given to cross-sectoral approaches (Hall et al. 2016), i.e. integrating different sectors such as energy, agriculture or solid waste management (inter alia Starkl et al. 2012). A closely related issue with respect to different sectors therefore relates to infrastructure dependencies (inter alia Hall et al. 2016), because critical infrastructures such as WMS, '*are not isolated, but instead are tightly coupled, creating a complex system of interconnected infrastructures. Dependencies between critical infrastructures can cause a failure to propagate from one critical infrastructure to other critical infrastructures*' (Laugé et al. 2015). Such considerations were completely left out of this thesis but would offer a fruitful strand of research. Overall, however, there is still a lot of room for improvement and an ambitious integrated research agenda is needed which combines technical, social, ecological, economic and institutional aspects (Hansman et al. 2006). The close link between wastewater and water in particular calls for an integrated perspective (Daigger 2007, Arora et al. 2015) and decentralised approaches need to be integrated into the urban environment (Urich and Rauch 2014). However, one critical thought on this matter may be allowed from a modeller's perspective: the question of choosing the right level of model complexity remains. The models presented in this thesis involve some strong assumptions and limitations which do, however, makes it easy to put the derived conclusions into context.

These models and tools should not be seen as detailed engineering solutions to be followed uncritically. More complex and integrated models will certainly make the interpretation of their results more difficult and there will be a stronger temptation to trust them blindly (cf. Cook 2016).

5.1.2 Including uncertainty and flexibility

Besides the obvious single-objective optimisation, one of the most fundamental modelling limitations and shortcoming of this thesis is static optimisation, even though the tools developed for this purpose can provide insight into possible transition pathways (cf. Section 2.4.4, Section 4.2.2.1 for more specific model limitations). No elaborate scenario analysis is included in this thesis and new approaches need to capture the temporal and spatial dynamics of the urban environment more effectively (inter alia Urich and Rauch 2014). More research is necessary to assess optimal investment strategies and quantify the benefits of flexibility (inter alia Hug et al. 2010). Robust strategies need to be found which consider not only the optimal distribution of wastewater systems dimensioning on 'green field sites' but include currently built infrastructures which cannot be neglected in serious decision-making. This would allow improved mapping of the transition towards new system configurations and allow dynamic transformations to be better captured. The methods presented here only allow for very crude mapping with the focus on screening for potential niches. In accordance with Morera et al. (2015), it remains a challenge to quantify the benefits of hybrid systems whilst considering the associated risks, flexibility and uncertainty.

5.1.3 Storm water and effluent removal

The transition towards decentralised WMS and a modification of existing centralised systems affects not only the remaining centralised infrastructure (e.g. Sitzenfrei et al. 2013a) but also poses new challenges. Thus in the case of the decentralised provision of drinking water, new solutions need to be found for fighting fires (Global Water Intelligence 2016). For decentralised wastewater treatment on a larger scale, the most important and open questions are the disposal of the treated effluent and the evacuation of storm water (inter alia Sedlak 2014, Manning et al. 2016). The approaches developed in this thesis do not consider storm water and effluent removal in depth and do not restrict the geographical placement of treatment units with respect to effluent criteria. However, the receiving water bodies and alternative ways of draining effluent would need to be considered for situating facilities. Even in the case of economically competitive treatments, decentralised systems cannot break free from drainage or sewer networks unless the excess water can be either safely (re-)used (Harris-Lovett et al. 2015) or released to the environment, e.g. via local effluent percolation or transportation to near receiving waters. Effluent and storm water disposal is especially important for urbanised areas, where limited space and the limited availability of receiving waters is generally used to argue against on-site solutions (cf. Rendings and Tranow 2006 who critically evaluate this argument) and the limited availability of receiving waters. Wastewater effluent and storm water would need to be integrated into urban-sensitive designs (Grant et al. 2016). However, a shift towards dispersed storm water management in urban situations can already be observed today and cost-effective systems are increasingly becoming possible with a smart combination of green infrastructures (e.g. vegetated swales, detention ponds, constructed wetlands, green roofs…), even though more research effort is clearly needed (cf. Raja Segaran et al. 2014). Bach et al. (2013a) show that on-site infiltration systems may often offer feasible opportunities for lot-scale storm water management. However, they stress the relationship between infiltration capacity and soil and that the percentage of the available lot size for percolation is important. Finally, to truly assess the effluent percolation potential and decide on decentralised treatment, extensive geographically explicit data are needed with respect to factor such as slope, soil characteristics, geological characteristics, groundwater depth, land use etc. (cf. Makropoulos et al. 2007, Deepa and Krishnaveni 2012), thus highlighting the importance of GIS (inter alia Ellis et al. 2013, Raja Segaran et al. 2014).

Fully or highly decentralised UWM may still depend on networked infrastructures, but only for draining waters (instead of combined sewers transporting different types of water to a central WWTP). Such networks would be much smaller, fractured and oriented with respect to the availability of the receiving waters. The challenge of disposing of storm- and effluent water is a promising research area, where approaches such as fully integrated modelling could be fruitfully applied to explore and find new ways of dealing with these waters (cf. Bach et al. 2013b/2014). The challenge of environmental (water, groundwater…) quality is naturally closely related to distributed effluent, but is discussed elsewhere (e.g. Morrissey et al. 2015).

5.1.4 Spatial price differentiation & incentivisation

Policymakers can influence the uptake of distributed urban water systems by creating cost incentives and disincentives (inter alia Arora et al. 2015). Just as the centralised building of sewer networks was heavily controlled and subsidised, financial incentives may be necessary to achieve more sustainable overall connection rates. There has been little spatial price differentiation for consumers so far (e.g. Moss 2001). With respect to the challenges of storm water and effluent removal as well as cost incentivisation (see previous section), one suitable research area would be the setting up of tax regulations, rather like charging storm water fees on the basis of the available impervious surfaces, which might be one way to incentivise behaviour in order to increase the percolation potential (e.g. Parikh et al. 2005, OECD 2015). Overall, more analysis is needed to promote water sensitive urban design (Iftekhar et al. 2016). Generally, municipalities may be forced to introduce incentives because of the reluctance of homeowners (for whatever reasons) to go off-grid (Wood et al. 2015). O'Flaherty (2005) argues that if households don't have to pay the actual connection costs, this affects where people live and how densely regions are populated. Network infrastructures tend to distribute costs evenly among space as costs are usually averaged among users connected to a network if no additional financial instruments are put in place (e.g. users must pay to connect to the sewer networks depending on their location). However, it remains a challenge to determine the actual network connection costs (and not average costs) of central sewers for a single household. There has so far been a lack of spatial price differentiation, and more research is needed on how to set prices in networks according to the principle that the polluter pays. Haug (2004) writes that '*if households choose their residence on the outskirts of town or in sparsely populated, remote regions it ought not to be debated in a market economy that they have to pay the adequate market price of housing and local public goods.*' Which financial mechanisms or institutional forms should be chosen may differ from case to case, and this leaves room for further research. Instead of distributing the high costs of a central connection over the whole network, it is worthwhile to start thinking about charging households according to their real connection costs, which would incentivise decentralised solutions (Reese et al. 2015). Eggimann et al. (2016a) touch on the issue of applying different tariffs within the same region for centralised or decentralised systems. However, the political implications of this step could be studied in more detail.

5.1.5 Decentral operation and management

Many different research needs and open questions remain, especially relating to the successful operation and management of decentralised systems (cf. Etnier et al. 2000). One recurring theme in infrastructure sectors is whether public or private ownership and/or operation automatically implies that a more cost effective service will be achieved. Both public and private modes come in many different guises (Marques et al. 2010, Lieberherr and Truffer 2015), and research efforts could look in more detail into this issue with respect to hybrid systems in particular. Furthermore, the implementation of cost-effective logistics and systems for decentralised operation was touched upon in Chapter 2. The relevant questions for widespread decentralised installation naturally go beyond road-based faecal sludge emptying and transportation and its cost effectiveness. Thus much more in-depth analysis is necessary with respect to safe operation via remote control thanks to new developments in sensors (cf. Eggimann et al. 2016b). But here too, the technological innovations designed to improve the O&M of new WMS configurations represents only one dimension: questions remain with respect to institutions, regulations and organisational forms (cf. Eggimann et al. 2016a), what kind of business model adaptations would be necessary (cf. Bolton and Hannon 2016, Hannon 2012) or with respect to how best to set up the required controlling institutions (inter alia Geyler and Holländer 2005). Learning from other systems such as the heating sector looks potentially promising, so scientific approaches must be developed that focus increasingly on decentralised solutions in diverse academic disciplines as a fruitful basis for further research.

5.2 Recommendations for policy and practice

One basic underlying assumption of the research focus chosen for this dissertation is that wastewater management systems undergo transformations, i.e. technologies, practices, costs and many more boundary conditions change over time. This thesis argues that such a perspective is highly recommended as a means of getting away from the view that infrastructure systems are set in stone. The findings of this dissertation allow different recommendations for policy and practice to be deduced (see also concluding remarks of all individual publications for more detailed information in Chapters 2, 3 and 4). All the following recommendations are made in order to achieve more optimal degrees of centralisation with the goal of minimising overall costs. However, these general recommendations need to be considered against the background of the limitations outlined in the previous section (see Section 5.1). The recommendations for policy and practice can be summarised as follows:

- Increasing the spatial price differentiation is necessary to facilitate the introduction of decentralised WMS and eliminate price distortions resulting from current practices in network infrastructure construction. Central networks should be regulated so that they connect households only for as long as the average central costs of all users connected to the network decrease (cf. Eggimann et al. 2016a).
- The strict enforcement of regulations such as the mandatory connection rule should be reconsidered. Adaptations to the regulatory framework for countries such as Switzerland are needed so that decentralised systems are not excluded *a priori* (cf. Truffer et al. 2013). Instead of continuously asking the question to whether a household should be allowed '*to go off-grid*', it should also be asked whether it should be '*allowed to connect*', even though this might be counter-intuitive to existing practices. Such alternative thinking should be applied especially when infrastructures are being constructed or renovated.
- Optimal CR are best achieved with hybrid WMS in many empirical cases. Countries with barely existing sewer infrastructures would be well advised to leapfrog the age of fully-centralised sewer systems, such as are the norm for Switzerland (cf. Poustie et al. 2016). This is especially true where today's local configurations clearly do not support the introduction of centralised infrastructures. Furthermore, the extension or construction of large sewer networks should especially be critically evaluated, especially in areas with complex topographies and low population densities.

- Rural areas with complex topographies offer particularly good niches for introducing the transition towards more decentralised WMS. Today's sewer-based systems such as the Swiss one could be transformed gradually if decentralised solutions are made to fit and complement the existing system. A shift from a centralised to a decentralised system could take place '*through a stepwise process of reconfiguration*' (Geels 2002). This thesis demonstrates that the build-up of decentralised infrastructures may become market-driven in some parts of the world as the installation of decentralised systems often makes economic sense. There is a need to integrate a long-term planning perspective where existing and still incompletely depreciated infrastructure networks constitute an economic challenge to the introduction of alternative WMS.

- Policy makers should be aware of 'threshold-effects' and changing on-site treatment costs. Even though a transition might be incremental due to of path-dependent investments (cf. '*hybridisation process*' by Marlow et al. 2013), this process may achieve radically new dynamics when certain connection rates are achieved so that the cost differences between centralised and decentralised systems could become negligible. Furthermore, the cost of decentralised systems should not be considered as static and it makes good sense to envision future scenarios with much lower on-site treatment costs.

- The spatial influence on the operation and maintenance (O&M) costs of decentralised systems is minor. However, smart O&M schemes need to be put in place in order to prevent costly idle capacities.

- Economic geospatial modelling tools are recommended for use in infrastructure planning, especially for screening potential spaces for niche experiments and to identify where centralised approaches prove to be economically less efficient. Investing in the development of off-the-shelf and easy-to use software might be one way forward to achieve the more widespread application of similar and adequate decision-making tools. ■

5.3 References

Arora, M., Malano, H., Davidson, B., Nelson, R., George, B., **2015**. Interactions between centralized and decentralized water systems in urban context: A review. *WIREs Water 2015(2)*, 623–634.

Bach, P.M., Deletic, A., Urich, C., Sitzenfrei, R., Kleidorfer, M., Rauch, W., McCarthy, D.T., **2013**a. Modelling interactions between lot-scale decentralised water infrastructure and urban form - a case study on infiltration systems. *Water Resources Management*, 27(14), 4845–4863.

Bach, P.M., McCarthy, D.T., Urich, C., Sitzenfrei, R., Kleidorfer, M., Rauch, W., Deletic, A., **2013**b. A planning algorithm for quantifying decentralised water management opportunities in urban environments. *Water Science and Technology*, 68(8), 1857–1865.

Bach, P.M., Rauch, W., Mikkelsen, P.S., McCarthy, D.T., Deletic, A., **2014**. A critical review of integrated urban water modelling – Urban drainage and beyond. *Environmental Modelling & Software*, 54, 88–107.

Bolton, R., Hannon, M., **2016**. Governing sustainability transitions through business model innovation: Towards a systems understanding. *Research Policy,* 45(9), 1731–1742.

Bradley, B.R., Daigger, G.T, Rubin, R., Tchobanoglous, G., **2002**. Evaluation of onsite wastewater treatment technologies using sustainable development criteria. *Clean Technologies and Environmental Policy*, 4, 87–99.

Cook, J.D., **2016**. Advantages of crude models. Link: http://www.johndcook.com/blog/2011/05/25/crude-models/ (Accessed 10.08.2016)

Cornejo, P.K., Zhang, Q., Mihelcic, J.R., **2016**. How does scale of implementation impact the environmental sustainability of wastewater treatment integrated with resource recovery? Environmental Science & Technology, 50(13), 6680–6689.

Daigger, G.T., **2007**. Wastewater Management in the 21st Century. *Journal of Environmental Engineering*, 133(7), 671–680.

Deepa, K., Krishnaveni, M., **2012**. Suitable site selection of decentralised treatment plants using multicriteria approach in GIS. *Journal of Geographic Information System*, 4, 254–260.

de Melo, J.J., Câmara, A.S., **1994**. Models for the optimization of regional wastewater treatment systems. *European Journal of Operational Research*, 73, 1–16.

Dominguez, D., Gujer, W., **2006**. Evolution of a wastewater treatment plant challenges traditional design concepts. *Water Research*, 40(7), 1389–1396.

Eggimann, S., Truffer, B., Maurer, M., **2016**a. The cost of hybrid waste water systems: A systematic framework for specifying minimum cost-connection rates. *Water Research*, 103, 472–484.

Eggimann, S., Mutzner, L., Wani, O., Schneider, M., Moy de Vitry, M., Beutler, P., Maurer, M., **2016**b. The potential of knowing more – A review of data-driven urban water management. Submitted to *Environmental Science & Technology*.

Ellis, J. B., **2013**. Sustainable surface water management and green infrastructure in UK urban catchment planning. *Journal of Environmental Planning and Management*, 56(1), 24–41.

Etnier, C., Nelson, V., Pinkham, R., **2000**. Economics of Decentralized Wastewater Treatment Systems: Direct and Indirect Costs and Benefits. A white paper for the National Decentralized Water Resources Capacity Development Project. In: *National Research Needs Conference Proceedings: Risk-Based Decision Making for Onsite Wastewater Treatment*, St. Louis, MO, USA.

Geels, F.W., **2002**. Technological transitions as evolutionary reconfiguration processes: a multi-level perspective and a case-study. *Research Policy*, 31(8-9), 1257–1274.

Geyler, S., Holländer, R., **2005**. Ein Vergleich von zentralen und dezentralen Lösungen zur Abwasserentsorgung im ländlichen Raum. Institutional Change in Agriculture and Natural Resources. Discussion Paper 8/2006.

Global Water Intelligence, **2016**. Water technology futures: a global blueprint for innovation. Report Singapore International Water Week.

Gils, H.C., Cofala, J., Wagner, F., Schöpp, W., **2013**. GIS-based assessment of the district heating potential in the USA. *Energy*, 58, 318–329.

Grant, G., **2016**. The Water Sensitive City. John Wiley & Sons, Chichester, UK.

Greengard, S., **2015**. The internet of things. MIT Press, London, England, UK.

Haghighi, A., Bakhshipour, A.E., **2015**. Reliability-based layout design of sewage collection systems in flat areas. *Urban Water Journal*, 1–13.

Hall, J.W., Tran, M., Hickford, A.J., Nicholls, R.J., **2016**. The Future of National Infrastructure. A System-of-Systems Approach. Cambridge University Press, Cambridge, UK.

Hannon, M., **2012**. Co-evolution of Innovative Business Models and Sustainability Transitions: The Case of the Energy Service Company (ESCo) Model and the UK Energy System. PhD Thesis. School of Earth and Environment, University of Leeds, UK.

Hansman, R.J., Magee, C., Neufville, R. de., Robins, R., Roos, D., **2006**. Research agenda for an integrated approach to infrastructure planning, design and management. *International Journal of Critical Infrastructures*, 2(2/3), 146–159.

Harris-Lovett, S.R., Binz, C., Sedlak, D.L., Kiparsky, M., Truffer, B., **2015**. Beyond User Acceptance: A Legitimacy Framework for Potable Water Reuse in California. *Environmental Science & Technology*, 49(13), 7552–7561.

Haug, P., **2004**. Decreasing Population and Rising Costs of Providing Water and Sewage Treatment for Urban Areas: A Case Study, Halle (Saale), Germany.

Ho, G., **2005**. Technology for sustainability: the role of onsite, small and community scale technology. *Water Science and Technology*, 51, 15–20.

Hug, T., Dominguez, D., Maurer, M., **2010**. The Cost of Uncertainty and the Value of Flexibility in Water and Wastewater Infrastructure Planning. Proceedings of the Water Environment Federation, Cities of the Future/Urban River Restoration 2010, 487–500.

Iftekhar, M.S., Urich, C., Schilizzi, S., Deletic, A., **2016**. Effectiveness of incentives to promote adoption of water sensitive urban design: A case study on rain water harvesting tanks. *8th International Congress on Environmental Modelling and Software*, Paper 64, 10–14. July, Toulouse, France.

Inamdar, P.M., Cook, S., Sharma, A.K., Corby, N., O'Connor, J., Perera, B.J. C., **2013**. A GIS based screening tool for locating and ranking of suitable stormwater harvesting sites in urban areas. *Journal of Environmental Management*, 128, 363–370.

Johnson, N., Ogden, J., **2012**. A spatially-explicit optimization model for long-term hydrogen pipeline planning. *International Journal of Hydrogen Energy*, 37(6), 5421–5433.

Laugé, A., Hernantes, J., Sarriegi, J. M. **2015**. Critical infrastructure dependencies: A holistic, dynamic and quantitative approach. *International Journal of Critical Infrastructure Protection*, 8, 16–23.

Liang, X., van Dijk, M.P., **2010**. Financial and economic feasibility of decentralized wastewater reuse systems in Beijing. *Water Science and Technology*, 61(8), 1965–1973.

Lieberherr, E., Truffer, B., **2015**. The impact of privatization on sustainability transitions: a comparative analysis of dynamic capabilities in three water utilities. *Environmental Innovation and Societal Transitions,* 15, 101–122.

Lienhoop, N., Al-Karablieh, E.K., Salman, A. Z., Cardona, J.A., **2014**. Environmental cost-benefit analysis of decentralised wastewater treatment and re-use: a case study of rural Jordan. *Water Policy*, 16(2), 323–339.

Makropoulos, C.K., Argyrou, E., Memon, F.A., Butler, D., **2007**. A suitability evaluation tool for siting wastewater treatment facilities in new urban developments. *Urban Water Journal*, 4(2), 61–78.

Manning, L.J., Graham, D.W., Hall, J.W., **2016**. Wastewater system assessments. In: Hall, J.W., Tran, M., Hickford, A.J., Nicholls, R.J., *The Future of National Infrastructure. A System-of-Systems Approach*. Chapter 7, Cambridge University Press, Cambridge, UK.

Marlow, D.R., Moglia, M., Cook, S., Beale, D.J., **2013**. Towards sustainable urban water management: A critical reassessment. *Water Research*, 47(20), 7150–7161.

Marques, R., Simões, P., Pires, J., **2010**. Regulation of water and wastewater services: an international comparison. Management International. IWA Publishing, London, UK.

Maurer, M., Bufardi, A., Tilley, E., Zurbrügg, C.,Truffer, B., **2012**. A compatibility-based procedure designed to generate potential sanitation system alternatives. *Journal of Environmental Management*, 104, 51–61

McConville, J.R., Künzle, R., Messmer, U., Udert, K.M., Larsen, T.A., **2014**. Decision support for redesigning wastewater treatment technologies. *Environmental Science & Technology*, 48(20), 12238–12246.

Morera, S., Comas, J., Poch, M., Corominas, L., **2015**. Connection of neighboring wastewater treatment plants: economic and environmental assessment. *Journal of Cleaner Production*, 90, 34–42.

Möller, B., Lund, H., **2010**. Conversion of individual natural gas to district heating: Geographical studies of supply costs and consequences for the Danish energy system. *Applied Energy*, 87(6), 1846–1857.

Morrissey, P.J., Johnston, P.M., Gill, L.W., **2015**. The impact of on-site wastewater from high density cluster developments on groundwater quality. *Journal of Contaminant Hydrology*, 182, 36–50.

Moss, T., **2001**. Flow management in urban regions: introducing a concept. In: Guy, S., Marvin, S., Moss, T., 2001. Urban infrastructure in transition: networks, buildings and plans. Earthscan, Abingdon, UK.

Naik, K.S., Stenstrom, M.K., **2016**. A feasibility analysis methodology for decentralized wastewater systems

– Energy-efficiency and cost. *Water environment research*, 88(3), 201-209.

OECD, **2015**. Water and cities. Ensuring sustainable futures. OECD Studies on Water. OECD Publishing, Paris, France.

O'Flaherty, B., **2005**. City economics. Harvard University Press.

Parikh, P., Taylor, M.A., Hoagland, T., Thurston, H., Shuster, W., **2005**. Application of market mechanisms and incentives to reduce stormwater runoff. An integrated hydrologic, economic and legal approach. *Environmental Science and Policy*, 8(2), 133–144.

Poustie, M.S., Deletic, A., Brown, R.R., Wong, T., de Haan, F.J., Skinner, R., **2014**. Sustainable urban water futures in developing countries: the centralised, decentralised or hybrid dilemma. *Urban Water Journal*, 1–16.

Poustie, M.S., Frantzeskaki, N., Brown, R.R., **2016**. A transition scenario for leapfrogging to a sustainable urban water future in Port Vila, Vanuatu. *Technological Forecasting and Social Change*, 105, 129–139.

Rendings, S., Tranow, I., **2006**. Dezentrale Abwasserentsorgung im urbanen Raum Ermittlung und Analyse von Eignungsgebieten in Hagen und Magdeburg. Diplomarbeit. Universität Dortmund. Germany.

Resse, M., Gawel, E., Geyler, S., **2015**. Die Nachhaltigkeitsgebote der Siedlungswasserwirtschaft – Kernziele, Grundvoraussetzungen und institutionelle Ansatzpunkte nachhaltiger Wasserver- und Abwasserentsorgung. In: Gawel, E. (eds), *Die Governance der Wasserinfrastruktur (Band I). Rahmenbedingungen, Herausforderungen und Optionen*. Duncker & Humblot, Berlin, Germany.

Raja Segaran, R., Lewis, M., Ostendorf, B., **2014**. Stormwater quality improvement potential of an urbanised catchment using water sensitive retrofits into public parks. *Urban Forestry and Urban Greening*, 13(2), 315–324.

Sanoh, A., Kocaman, A.S., Kocal, S., Sherpa, S., Modi, V., **2014**. The economics of clean energy resource development and grid interconnection in Africa. *Renewable Energy*, 62, 598–609.

Scholten, L., **2013**. Multi-criteria decision analysis for water supply infrastructure planning under uncertainty. Dissertation, ETH Zürich, Switzerland.

Sedlak, D., **2014**. Water 4.0: The past, present, and future of the world's most vital resource. Yale University Press, New Haven & London.

Sitzenfrei, R., Möderl, M., Rauch, W., **2013**a. Assessing the impact of transitions from centralised to decentralised water solutions on existing infrastructures--integrated city-scale analysis with VIBe. *Water Research*, 47(20), 7251–7563.

Starkl, M., Parkinson, J., Narayana, D., Flamand, P., **2012**. Small is beautiful but is large more economical? Fresh views on decentralised versus centralised wastewater management. *Water 21 (June)*, 45–47.

Suriyachan, C., Nitivattananon, V., Amin, A.T.M.N., **2012**. Potential of decentralized wastewater management for urban development: Case of Bangkok. *Habitat International*, 36(1), 85–92.

Truffer, B., Binz, C., Gebauer, H., Störmer, E., **2013**. Market success of on-site treatment: a systematic innovation problem. In: Larsen, T. A., Udert, K.M., Linert, J. (eds), *Source separation and decentralization for wastewater management*. IWA Publishing, London, UK.

Urich, C., Rauch, W., **2014**. Modelling the urban water cycle as an integrated part of the city: A review. *Water Science and Technology*, 70(11), 1857–1872.

Weber, B., Cornel, P., Wagner, M., **2007**. Semi-centralised supply and treatment systems for (fast growing) urban areas. *Water Science and Technology*, 55(1-2), 349–356.

Wood, A., Blackhurst, M., Lawler, D., **2015**. To Sewer or Not to Sewer: Incentivizing Decentralized Wastewater Treatment. World Environmental and Water Resources Congress 2015, 1362–1367.

Zeferino, J.A., Antunes, A.P., Cunha, M.C., **2010**. Multi-objective model for regional wastewater systems planning. *Civil Engineering and Environmental Systems*, 27(2), 95–106.

Chapter 6

General conclusion and contribution

This dissertation has addressed hitherto neglected economic aspects of assessing the optimal degree of centralisation for wastewater infrastructures within a geospatial modelling framework. By applying a more comprehensive full-cost framework, it is increasingly becoming possible to go beyond vague and unsubstantiated economic speculation about central versus decentral service provision in urban water management. Despite the limitations of the framework presented here, which calls for further analysis, this socio-technical approach to infrastructure transformation represents an innovative and much-needed research effort for UWM infrastructure planning. An exploration of how socio-technical innovations in wastewater treatment affect the transformation of existing infrastructure systems on the basis of a spatial and economic focus has so far been lacking. The economic geospatial modelling tools developed here contribute to the discussion about the ODC not on a purely theoretical level but allow us to explore optimal UWM system layouts and detect potentially interesting case studies for further in-depth analysis. This dissertation makes it very clear that the seemingly straightforward research question about the ODC becomes truly challenging if additional complexities are included in the analysis. By including the socio-technical dimension in particular, it becomes evident that it is not only important to promote technological advances but also to suggest appropriate institutional and organisational arrangements. The main contributions of this thesis are summarised concisely below:

- This thesis clearly shows that geography impacts the costs and planning of wastewater management infrastructures in different ways. Spatial influences therefore need to be considered when seeking economically more sustainable connection rates. Space has a significant influence on the costs of centralised, decentralised and hybrid WMS, and any assessments of optimal infrastructure planning that ignore spatial aspects must consequently be critically evaluated in this respect.
- So far very few in-depth and (economic) geospatially explicit tools for determining optimal system layouts have been presented in UWM. However, they offer a meaningful approach for considering spatial influences in the cost calculation. In this thesis, cost-optimised hybrid WMS layouts are found within a heuristic geospatial optimisation framework consisting of centralised and decentralised WMS. In the process, this thesis argues in favour of embracing a more hybrid systems thinking in the field of UWM. In several Swiss case studies, considerable potential was shown to exist for more decentralised system configurations from an economic point of view. Regulations such as the mandatory central connection rules which make the introduction

of decentralised technologies more difficult should consequently be reconsidered (see Section 5.2 for more specific policy advice to be derived from the findings of this dissertation).

- This thesis provides a first cost assessment of the economies of density for decentralised WMS with the aid of a geospatial model based on a heuristic routing algorithm. Our theoretical understanding of the operation and maintenance costs relating to spatial aspects was improved by this modelling exercise, indicating that these are generally overestimated and may not constitute a major stumbling block to decentralised WMS.

- Different optimal connection rates can be determined depending on the institutional and organisational setting. Cost-efficient infrastructure planning is therefore inherently linked to non-technical aspects and cannot be analysed separately. Even though technological innovations are important, the same applies to innovations relating to the institutional framework of the UWM. The co-evolution of institutional, organisational and technological innovations in the field of UWM is necessary and once more underlines that the successful introduction of decentralised approaches would require highly focused action which '*would involve engineers as well as social scientists, industry representatives and policymakers*' (Truffer et al. 2013).

- Today's way of thinking about the service provision of households is often dominated by the question as to whether households should be allowed to go off-grid. This dissertation encourages a complementary approach to current ways of thinking by asking the question of whether households should be allowed to be connect to the network.

- The research has so far not seriously considered the implications of the costs for decentralised WMS becoming truly competitive to centralised WMS. This thesis provides a first indication of '*threshold*' effects: As the costs of decentralised systems continue to fall, the connection rate will first start to decrease slowly, especially as marginal population living in peripheral areas are disconnected. However, if the central connection rate falls low enough, the cost differences between central and decentral systems may become minimal because of the interplay of the various cost elements involved (see Chapter 4). This would result in questioning the central approach even in regions where it has so far been clearly considered as being more cost-effective.

- Improved (modelling) efforts are needed to assess the ODC in a more holistic way. Further promising strands of research which would improve this analysis are suggested in Section 5.1. The development of multi-objective modelling approaches as well as research on how to deal with effluents are particularly necessary to improve our understanding of the ODC. Furthermore, future research needs to include more sophisticated uncertainty analyses, especially with the focus on flexibility and the question about 'when' and 'where' to invest.

The fundamental question of whether to connect as many households as possible to a centralised WMS or whether it would make more sense to invest in decentralised systems remains challenging and urgent. It is exciting to see how (future) changes in the socio-technical configuration in UWM will influence the transformation of today's WMS with respect to the optimal degree of centralisation. I hope that the findings of this thesis will encourage to envision and reflect more thoroughly on radical approaches for future UWM infrastructure provision, not least from an economic perspective, with the aim of providing UWM services that are more economically sustainable. ■

Curriculum Vitae

2013 - 2016	Dissertation at eawag *Dübendorf, Switzerland*
2013	**Master of Advanced Studies in Secondary and Higher Eduction** *University of Zürich, Switzerland*
2010 - 2013	**Master of Science in Geography** Major in Geographical Information Systems Minors in Environmental Sciences, Agricultural Economics, Spatial Planning and Regional Development *University of Zürich, Switzerland*
2010 - 2013	**Assistantship, ETH Department of Architecture and ETH Studio Basel** *Basel and Zürich, Switzerland*
2011 (Jun - Oct)	**Internship at the Environment and Energy Department** *Lucern, Switzerland.*
2007 - 2010	**Bachelor of Science in Geography** *University of Zürich*
17.03.1988	Born in Unterseen, Switzerland.
Languages	(Swiss) German mother tongue Full professional Proficiency in English (CPE) Fluency in French (8 years) Basic working knowledge in Italian (3 years)

Font Copyright

In many-high income countries, enormous investments have been made in the last century to build a centralised urban water management system with vast infrastructure networks and centralised treatment facilities. This historically evolved centralised infrastructure paradigm is increasingly being questioned with respect to sustainability challenges, and a trend towards decentralisation can be observed. Today, large differences exist with respect to the population percentages connected to the centralised systems of different countries and it is clearly unclear to what degree centralised or decentralised wastewater systems should be installed in a region from an economic point of view. This thesis develops spatio-economic modelling tools to assess the optimal degree of centralisation and presents a cost framework to develop the argument for a transition towards more sustainable wastewater management systems.

Eggimann Sven (1988) is specializing in Geographic Information Systems, sustainable infrastructure planning and economic aspects of sustainability transitions.

From 2013 to 2016 he wrote his doctoral thesis at eawag, the Swiss Federal Institute of Aquatic Science, and at ETH Zurich.