THESIS FOR THE DEGREE OF DOCTOR OF PHILOSOPHY

Fresh Perspectives on the Assessment of Sewage Sludge Management

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Storage of treated sewage sludge at the wastewater treatment works in Gothenburg.
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Abstract

Sewage sludge management and its role in closing nutrient cycles have received considerable attention in recent years. This thesis quantified phosphorus flows in Gothenburg, Sweden, under current and possible future waste management practices, and aimed to improve the assessment of wastewater and sludge management from an environmental and human health perspective through blending risk assessment (RA) and life cycle assessment (LCA). A review of previous environmental assessment case studies revealed inconsistent use of terminology regarding what is meant by integration, combination, hybridisation, or integrated use of RA and LCA. To facilitate a better understanding and more transparent communication of the nature of a given case study, this thesis proposed a design space that outlines choices to be made when blending RA and LCA. For the assessment of human health effects, this thesis suggests that a case study should only be referred to as a combination or integration of RA and LCA if it addresses two distinct perspectives: risks for specific members of a given human population (RA perspective) and overall impacts for a given human population (LCA perspective). RA and LCA can also be blended by transferring model elements from one framework to the other. This thesis explored the transfer of elements of quantitative microbial risk assessment (QMRA) to an LCA framework in order to account for adverse effects of pathogens on human health. Such practice was found to be adequate, but it is important to ensure that exposure pathways and parameters are chosen in accordance with the principles applied in the LCA study of which the assessment is a part. Also, in the context of sewage sludge management, the consideration of non-routine operation scenarios in LCA may be warranted. This thesis also explored different models to assess human health effects related to chemical contaminants in the context of land application of sewage sludge. The different model variants investigated provided different burden of disease estimates for individual chemical contaminants, but an aggregate burden of disease estimate of the same order of magnitude. Overall, this thesis emphasises the importance of explicitly contemplating which type of question relevant to sewage sludge management can be answered by quantitative assessment tools such as RA and LCA.

Keywords: life cycle assessment, risk assessment, life cycle impact assessment, wastewater, sewage sludge, biosolids, pathogen risk, human toxicity, chemical risks, land application

List of Publications

Appended Papers

This thesis is based on the work presented in the following peer-reviewed journal articles, referred to throughout the text using their Roman numerals.

I. Kalmykova Y, **Harder R**, Borgestedt H, Svanäng I (2012) Pathways and management of phosphorus in urban areas. *Journal of Industrial Ecology* 16(6):928-939. DOI: 10.1111/j.1530-9290.2012.00541.x

II. **Harder R**, Holmquist H, Molander S, Svanström M, Peters G (2015) Review of environmental assessment case studies featuring elements of risk assessment and life cycle assessment. *Environmental Science & Technology*. DOI: 10.1021/acs.est.5b03302

III. **Harder R**, Heimersson S, Svanström M, Peters G (2014) Including pathogen risk in life cycle assessment of wastewater management. 1. Estimating the burden of disease associated with pathogens. *Environmental Science & Technology* 48(16):9438-9445. DOI: 10.1021/es501480q

IV. **Harder R**, Peters G, Molander S, Ashbolt N, Svanström M (2015) Including pathogen risk in life cycle assessment: the effect of modelling choices in the context of sewage sludge management. *The International Journal of Life Cycle Assessment*. DOI: 10.1007/s11367-015-0996-2

V. **Harder R**, Peters G, Svanström M, Khan S, Molander S (2015) Accounting for heavy metals and organic contaminants in life cycle assessment of sewage sludge management: the effect of modelling choices on estimated human toxicity potential. *Manuscript submitted for consideration in The International Journal of Life Cycle Assessment*.

Related Papers

The following peer-reviewed and non-peer-reviewed° journal articles are related to the topic of this thesis but are not appended to this thesis. They are referred to throughout the text using their Roman characters.

A. Heimersson S, **Harder R**, Peters G, and Svanström M (2014) Including pathogen risk in life cycle assessment of wastewater management. 2. Quantitative comparison of pathogen risk to other impacts on human health. *Environmental Science & Technology* 48(16):9446-9453. DOI: 10.1021/es501481m

°**B.** **Harder R**, Schoen M, Peters G (2015) Including pathogen risk in life cycle assessment of wastewater management. Implications for selecting the functional unit. *Environmental Science & Technology* 49(1):14-15. DOI: 10.1021/es505828n

Contribution Report

The author of this thesis has made the following contributions to the appended papers.

I Co-author. Contributed to data collection, data analysis, and discussion of results. Main contributor to writing the paper.

II First author. Principal contributor to planning the research. Designed and carried out the analysis of the case studies. Wrote the paper for the most part and made adjustments after consultation with the co-authors.

III First author. Principal contributor to planning the research. Implemented the computational model and carried out the calculations. Wrote the paper and made adjustments after consultation with the co-authors.

IV First author. Principal contributor to planning the research. Implemented the computational model and carried out the calculations. Wrote the paper and made adjustments after consultation with the co-authors.

V First author. Principal contributor to planning the research. Implemented the computational model and carried out the calculations. Wrote the paper and made adjustments after consultation with the co-authors.

Additional Publications by the Author

The following peer-reviewed journal articles are not directly related to the topic of this thesis but were also authored or co-authored during the doctoral studies at Chalmers.

Harder R, Kalmykova Y, Morrison G, Feng F, Mangold M, Dahlén L (2014) Quantification of goods purchases and waste generation at the level of individual households. *Journal of Industrial Ecology* 18(2):227-241. DOI: 10.1111/jiec.12111

Ordoñez-Pizarro I, **Harder R**, Nikitas A, Rahe U (2015) Waste sorting in apartments: Integrating the perspective of the user. *Journal of Cleaner Production* 106:669-679. DOI: 10.1016/j.jclepro.2014.09.100

Mangold M, Morrison G, **Harder R**, Hagbert P, Rauch S (2014) The transformative effect of the introduction of water volumetric billing in a disadvantaged housing area in Sweden. *Water Policy* 16(5):973-990. DOI: 10.2166/wp.2014.105

Harder R, Dombi M, Peters G (2015) Boundary riding around household metabolism. *Manuscript submitted to Journal of Environmental Planning and Management. Major revisions recommended post peer-review.*

"You can’t find the right answer if you’re asking the wrong question."

– John C. Maxwell

Preface and Acknowledgements

The research journey leading to this doctoral thesis started in January 2011 at the Department of Civil and Environmental Engineering. I would like to express my thanks to Yuliya Kalmykova and Greg Morrison for enabling me to start my doctoral education at Chalmers, which brought forth my licentiate thesis entitled *Quantifying the Metabolism of Individual Households*. In July 2013, I was offered the opportunity to continue my research journey at the Department of Chemistry and Chemical Engineering. I am very grateful to Greg Peters and Magdalena Svanström for this chance to pursue my doctoral education at Chalmers, which led to the completion of this doctoral thesis.

The research presented in this thesis was funded by the Built Environment Area of Advance, Chalmers, the Swedish Research Council for Environment, Agricultural Sciences, and Spatial Planning (Formas), and the municipality of Gothenburg, which are gratefully acknowledged. Also the valuable contributions of Lena Blom (Göteborgs Stad) as well as David I'Ons and Ann Mattsson (Gryaab AB) are gratefully acknowledged.

Both internal and external collaborators were involved in the successful completion of the appended papers. Collaboration was particularly intense with my main supervisor and examiner Greg Peters, my two co-supervisors Magdalena Svanström and Sverker Molander, and two fellow doctoral students at Chemical Environmental Science: Sara Heimersson and Hanna Holmquist. I have very much enjoyed the exchange of ideas and thoughts, and appreciated the useful and relevant feedback on various manuscripts. It simply was a pleasure to be surrounded by people with sharp minds that are willing to continue asking questions and thinking further where others would stop. I was also fortunate to receive feedback quickly in times were a timely response was crucial. Apart from the co-authors of the appended papers, there have also been a number of anonymous reviewers, whom I don't know, but whose contributions were highly valuable and appreciated. I also enjoyed close collaboration with two fellow doctoral students at other divisions at Chalmers: Isabel Ordoñez and Mikael Mangold. I enjoyed the many inspiring discussions about the role and limitations of science in general, and the contributions of our research in particular. When it comes to the working environment, I am very thankful to Greg Peters for giving consideration to my individual working rhythm.

Life as doctoral student has ups and downs, but life in general probably isn't equally smooth at all times. I would like to thank all my colleagues at Chalmers for their support throughout the years, for sharing their own experiences of being a doctoral student, for offering their advice and perspectives, and for pleasant lunches and coffee breaks. Looking back at my time as doctoral student, I can definitely say that I learned a good deal about politics in academia, and about how I act in different situations. The doctoral education at Chalmers possibly was as much about personal development as about professional development. I would like to thank Francois Clemens for mentioning to me after the presentation of my MSc thesis in February 2010 that I actually should do a PhD. It took me a while until I figured it out myself, but he certainly was right. I am also thankful for his continued support throughout my doctoral studies by providing perspectives from a different academic institution.

Fortunately, in Sweden it is possible to have a life besides working on the doctoral thesis, which I was happy to share with family and friends. I am grateful to my parents for always being there for me and supporting me, for listening to how things go and talking about life in general, and for their generosity and unconditional love. A big thank you also goes to all my friends, I enjoyed spending time together, be it during lunches or dinners, during a hiking or bicycle tour, while dancing tango or practicing yoga, or during a visit to Gothenburg. Without your support and confidence this thesis might not have materialised.

Göteborg, November 2015

Robin Harder

Contents

Chapter 1

Introduction

1.1 Thesis Background*

The provision of clean and safe drinking water and reliable sanitation services is often taken for granted in industrialised countries, but is not a trivial task. Centralised systems consisting of sophisticated water and wastewater treatment plants as well as extensive water supply and sewer networks are currently the prevailing approach in industrialised countries. The origins of these systems can be traced back to the beginning of the 20th century, when water-borne sanitation through flush sewer systems was introduced following the increasing availability of household water supplies and the introduction of water closets.

The introduction of water-borne sanitation was successful in terms of its positive contribution to public health and, where flush sewers were designed to facilitate drainage of storm water along with domestic wastewater, also in terms of its contribution to reducing flooding in urban areas. But nutrient cycles were replaced by linear nutrient flows, as nutrients previously recycled to productive land were now dispersed to the environment. The introduction of water-borne sanitation hence also created new problems wherever raw wastewater was discharged to surface water bodies. The most evident problems were visual impairment, eutrophication, and oxygen depletion leading to fish kills.

* Parts of this section (pages 1-2) were adapted from section 1.1.2 in a report entitled "Source-separation in the urban water infrastructure" commissioned by the Dutch Foundation for Applied Water Research (STOWA) (Harder, 2012).

The invention of the activated sludge process in 1913 was the first step towards biological wastewater treatment. In its early days, wastewater treatment essentially aimed at removing organic material and nutrients from wastewater. Organic material and nutrients were concentrated into sewage sludge, the residual material left after collection and treatment of wastewater. Since the invention of the activated sludge process, wastewater treatment processes were continuously improved to meet ever more stringent limits to the emission of organic material and nutrients to receiving water bodies through the effluent.

In the course of time, other problems of industrialisation became increasingly apparent in the urban water infrastructure. The 20th century for instance saw the advent of cars and car culture, and a further expansion of the chemical and pharmaceutical industries. As a result, municipal wastewater now contains a wide range of pollutants released to stormwater (e.g. polychlorinated aromatic hydrocarbons, heavy metals, and nonylphenols in runoff from roofs and streets), industrial wastewater, or domestic wastewater (e.g. hormones and pharmaceutical residues). These pollutants, unless broken down in the treatment process, end up in either the treated effluent or the sewage sludge.

Industrialisation has also brought about industrial livestock and biofuel production, which further contributed to disturbed nutrient cycles. The advent of industrial production of inexpensive mineral fertiliser in the first half of the 20th century meant that the nutrients dissipated through industrial production of food and fibre as well as the urban drainage infrastructure could be replenished by mineral fertiliser. As a result, contemporary industrialised agricultural production relies heavily on mineral fertiliser containing, amongst other ingredients, phosphorus mined from phosphate rock. Phosphorus is an element of paramount importance as a nutrient to safeguard food-supply.

The increasing awareness of the finite nature of phosphorus and the insight that food production cannot rely indefinitely on phosphate rock supplies (Cooper et al., 2011; Lougheed, 2011) has promoted several attempts to close nutrient cycles. In urban water management, the issue of broken nutrient cycles thus has reappeared on the agenda, though this time from a resource perspective rather than from an emission perspective.

Land application of sewage sludge is one option for recycling phosphorus and other nutrients to productive land, but has been heavily debated in Sweden and elsewhere for a long time (Bengtsson and Tillman, 2004). Land application is currently the dominant sewage sludge management option in the EU-27 countries with an average share of 44% in 2010 (Kelessidis and Stasinakis, 2012). However, this share varies strongly between EU-27 countries. In the Netherlands, land application is prohibited, whereas in Portugal close to 90% of the sewage sludge was used in agriculture in 2010 (Kelessidis and Stasinakis, 2012). Proponents and opponents of land application alike recognise the necessity of closing nutrient cycles and reducing the reliance on phosphate rock. Yet opponents highlight that pathogens and chemical pollutants present in sewage sludge may lead to undesired effects on the quality of agricultural soil and products as well as on human health.

Alternative nutrient recovery strategies advocated by opponents of land application include the extraction of nutrients from sludge or from solids remaining after partial or complete sludge oxidation. These strategies include, amongst others, the incineration of sewage sludge followed by thermochemical processes or leaching, supercritical water oxidation, and pyrolysis. The advantage of such strategies is the possibility to separate phosphorus from pathogens, organic pollutants, and possibly also from heavy metals. Nutrients could also be recovered at other places in the urban water infrastructure and from streams other than sewage sludge, for instance through source-separation approaches (e.g. urine diversion or dry toilets) or recovery from the sludge dewatering stream (i.e. leachate obtained during sludge dewatering). This thesis is rooted in efforts to recycle phosphorus from sewage sludge to productive land.

1.2 Thesis Context

Sewage sludge management and its role in closing nutrient cycles have received attention at the local, regional, national, and international levels, and the debate on how to best manage sewage sludge includes a variety of stakeholders. The work presented in this thesis is embedded in four research projects.

Phosphorus Flows in Gothenburg (P Flows)

In 2010, a new research group on Urban Metabolism was established at the Department of Civil and Environmental Engineering, Chalmers University of Technology (Chalmers). One of the first research projects of the newly established research group was the investigation of phosphorus flows and management options in urban areas. This project was funded through the Built Environment Area of Advance, Chalmers.

Novel Processing Routes for Sewage Sludge Management (EU Routes)

In 2010, a consortium of 18 partners from industry and academia received funding from the European Union's Seventh Framework Programme for Research and Technological Development under grant agreement 265156. The *Routes* project ran from May 2011 to April 2014 and was concerned with finding novel processing routes and innovative system solutions for effective wastewater treatment and sewage sludge management. The work described in this thesis relates in particular to work package 5 (integrated sustainability assessment), for which the research group Chemical Environmental Science, Chalmers, was the work package leader. The contribution of Chemical Environmental Science consisted of life cycle assessment (LCA) and life cycle costing (LCC) of different proposed wastewater and sludge treatment scenarios.

Hybrid Life Cycle Risk Assessment (Formas LiCRA)

In 2012, the research group Chemical Environmental Science received funding from the Swedish Research Council for Environment, Agricultural Sciences, and Spatial Planning (Formas) under grant agreement 2012-1122. Additional financial and in-kind support was received from the municipality of Gothenburg. The *LiCRA* research project aimed to provide a new and unifying perspective in the Swedish sludge debate by hybridising risk assessment (RA) and LCA, thus allowing for the consideration of both local and global impacts associated with different sewage sludge management options.

LCA of Sewage Sludge Management with Phosphorus Utilisation (SVU)

In 2014, the research group Chemical Environmental Science received funding from the Swedish Water and Wastewater Association (SVU) under grant agreement 14-128, and from six municipally-owned companies in the Swedish water sector: Stockholm Vatten (Stockholm region), Käppalaförbundet (Stockholm region), SYVAB (Stockholm region), Gryaab (Gothenburg region), VA SYD (Malmö region), and Uppsala Vatten (Uppsala region). This funding was earmarked for the evaluation of the environmental performance of different alternatives to recycle phosphorus from sewage sludge to productive land.

1.3 Research Objectives

The overall aim of the work presented in this thesis is to support the Swedish water sector in the strategic planning of future wastewater and sludge management systems that close nutrient cycles. Broadly speaking, this thesis is grounded in two overall research objectives.

- RO 1: To study phosphorus flows in urban areas and to investigate to what extent wastewater and sludge management practices can influence flows and sinks of phosphorus in urban areas.
- RO 2: To improve the assessment of wastewater and sludge management options from an environmental and human health perspective through blending RA and LCA.

1.4 Research Approach

RO 1 is a goal-driven research objective, as the choice of assessment method was not prescribed. The work on RO 1 formed the basis for Paper I.

> Paper I identified phosphorus flows for the current wastewater management system in Gothenburg by means of material flow analysis (MFA) and discussed alternative wastewater and sludge management options in terms of their effect on flows and sinks of phosphorus.

RO 2 is a method-driven research objective, as the choice of assessment method was specified. How exactly RA and LCA would be blended was to a large extent influenced by the aims related to the contributions made to the *LiCRA* and *Routes* projects. The main aim related to the contributions to the *LiCRA* project was to investigate hybridisation of RA and LCA in order to better assess human health effects related to chemical contaminants in sewage sludge. The main aim of the contributions made to the *Routes* project was to integrate human health effects related to pathogens in wastewater and sludge into LCA. Obviously, the choice of quantitative microbial risk assessment (QMRA) as starting point was influenced by the main aim of the *LiCRA* project. The research approach may be best described as experiential or explorative, with some of the research work focusing on case studies (Papers III to V), and other research work being of a more reflective character (Paper II).

> Paper II analyses previous environmental assessment case studies that claimed a combination or integration of RA and LCA, both in the field of wastewater and sludge management and in other fields.

> Paper III explores the use of quantitative microbial risk assessment (QMRA) with the intent to include the effects of pathogens on human health in LCA of wastewater and sludge management options.

> Paper IV explores whether the mathematical relationships and the model structure of QMRA can be simplified for use in an LCA framework.

> Paper V explores the application of a context-dependent LCIA model for human toxicity in the context of land application of sewage sludge.

Quite naturally, Papers II to V have influenced one another. For example, Paper II contains elements that represent a reflection on the approach underlying Paper III, and insights obtained during the work on Paper IV influenced the framing of Paper V.

1.5 Research Questions

In order to present the research work in a structured way, a set of research questions was formulated. The study of flows and sinks of phosphorus in urban areas (RO 1) corresponds to one research question (RQ 1). Blending elements of RA and LCA (RO 2) corresponds to three research questions (RQ 2 to RQ 4).

RQ 1: Which are the key flows and sinks of phosphorus in Gothenburg under the current and possible alternative wastewater and sludge management systems?

RQ 2: How can RA and LCA be meaningfully blended in order to better assess wastewater and sludge management options from an environmental and human health perspective?

RQ 3: How can the human health effects related to pathogens present in wastewater and sludge be included in LCA?

RQ 4: Does a context-dependent LCIA model to assess the human health effects related to chemical contaminants present in sludge applied to agricultural land provide significantly different results compared to a generic LCIA model?

1.6 Thesis Structure

The overall structure of this thesis in terms of the connections between the thesis background, the thesis context, the research objectives, the research questions, and the appended research papers is visualised in Figure 1.1. The remaining chapters of this thesis are arranged as follows. Chapter 2 provides the theoretical and methodological background required to place the thesis contributions in context. Chapter 3 summarises the background of the appended papers and discusses the main findings in relation to the research questions. Chapter 4 considers the implications of the research findings for strategic planning of wastewater and sludge management, and concludes this thesis by presenting suggestions for reflection and further research.

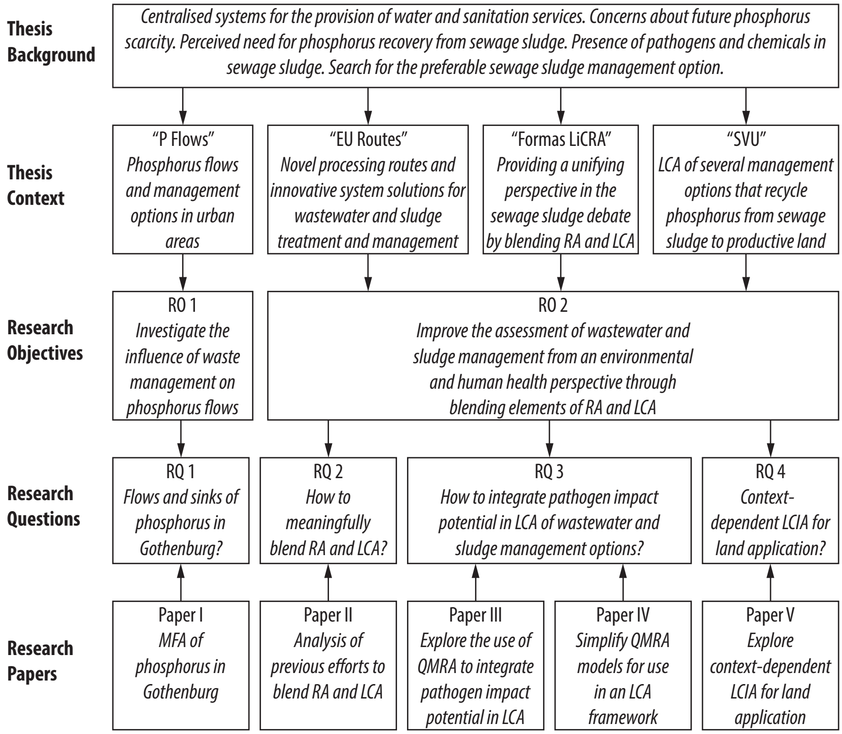

Figure 1.1: Overall structure of this thesis in terms of the connections between the thesis background, the thesis context, the research objectives, the research questions, and the appended research papers.

Chapter 2

Theory and Methods

The papers underpinning this thesis are based on MFA (Paper I) as well as on RA and LCA (Papers II to V). These tools are all attempts to describe certain properties of a real world system with the intent to support decision making. Ultimately, any such attempt is grounded in a particular view of the world. In addition to introducing the key characteristics of the three tools, and the relationships between them, this chapter therefore also touches upon philosophical perspectives on problem solving, the nature of real world systems, and the nature of modelling.

2.1 Approaches to Scientific Epistemology

Michel Foucault, a French philosopher and historian of science, claimed that all periods of history are subject to certain epistemological assumptions on how knowledge is ordered, and on what is acceptable as scientific discourse (Foucault, 1970). The foundations of thought that are taken for granted in a given period of history Foucault referred to as *episteme*. Foucault analysed three epistemes of relevance in the European context: the Renaissance, the Classical, and the Modern episteme. At first glance, Foucault's concept of epistemes may appear similar to the concept of paradigms put forward by Kuhn (1962). But epistemes apply more broadly than paradigms, that is, to the whole of society rather than a specific scientific discipline, and during longer periods of time.

2.1.1 Positivism

The Modern episteme corresponds well with the so-called *technocratic* or *positivist* worldview described by Tukker (2000). The Modern episteme currently is the dominant episteme in Western societies (Birkin and Polesie, 2012), possibly because training in science and scientific research is often firmly rooted in a positivist worldview (Ravetz, 1993). The Modern episteme and the positivist worldview in principle are characterised by a reductionist approach leading to mechanistic descriptions of the world in terms of its parts (Birkin and Polesie, 2012). Hereby, it is assumed that the real world is directly accessible for the observer, that observations and interpretations can be made without any ambiguity, and that value choices are identifiable (Tukker, 2000). Problem solving under a positivist worldview is characterised by defining a problem, selecting an appropriate assessment method, generating and assessing different alternatives, and selecting the best alternative (Tukker, 2000). Hereby, the positivist worldview takes for granted that a given problem can be solved based on objective and correct knowledge (Ravetz, 1993; Tukker, 2000).

2.1.2 Beyond Positivism

The positivist worldview and the corresponding approach to science have increasingly been challenged in recent years. Based on insights in the fields of neuroscience, psychology, and sociology, Ropeik (2012) suggested that the rationalist belief in the achievability of a perfectly rational, coldly objective, purely fact-based decision making is naïve. Birkin and Polesie (2012) extended Foucault's work and described the gradual emergence of a new episteme, which they refer to as the Primal episteme. The shift from the Modern to the Primal episteme is related to, amongst others, advancements in the understanding of living systems (von Bertalanffy, 1968; Maturana and Varela, 1973; Rosen, 1991; Varela et al., 1974), new insights into the relationship between models and the real world system under study (Rosen, 1978, 1985), the recognition of wicked problems (Rittel and Webber, 1973), and the development of the concept of post-normal science (Funtowicz and Ravetz, 1993, 1994; Ravetz, 1999).

2.1.3 Kuhnian Relativism

The term wicked problem was coined by Rittel and Webber to refer to a class of problems that are generally ill-defined and difficult to solve, where the information is confusing, where different stake-holders hold conflicting values, and where the proposed solutions often turn out to display symptoms worse than those attempted to be mitigated (West Churchman, 1967). The concept of post-normal science was proposed by Silvio Funtowicz and Jerome Ravetz (Funtowicz and Ravetz, 1993, 1994) as a problem-solving strategy when system uncertainties or decision stakes (or both) are high, and facts and values cannot be realistically separated. To address such problems, Funtowicz and Ravetz suggested an extended peer community, the consideration of different perspectives, and explicit handling of different types of uncertainties. The recognition that knowledge cannot be fully value-free, and that there may exist more than one, equally acceptable rationality or logic to analyse a situation corresponds to a worldview also referred to as *Kuhnian relativism* or *constrained relativism* (Tukker, 2000). Note that Kuhnian relativism does not mean that "anything goes". In fact, not *any* representation of the external world, however malconstructed, deserves a place (Tukker, 2000).

2.1.4 Robert Rosen's Modelling Relation

Robert Rosen, a theoretical biologist, instituted a rigorously mathematical treatise on the subject of anticipatory systems (Louie, 2010). A central part of Rosen's work is the so-called *modelling relation* (Figure 2.1).

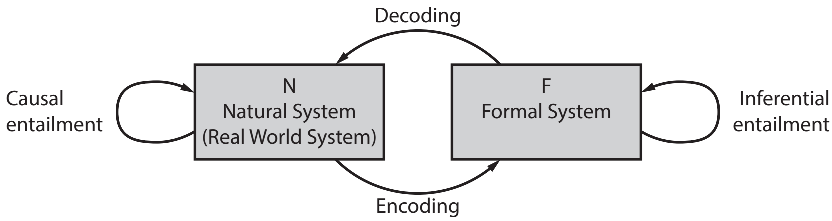

Figure 2.1: Robert Rosen's Modelling Relation (adapted from Keirstead (2014)).

Robert Rosen's modelling relation is based on the belief that the world has some sort of order associated with it (Mikulecky, 2000). The modelling relation relates two systems, a *natural system* and a *formal system* (Mikulecky, 2000). Hereby, the natural system is the world we are observing or trying to understand through other ways of knowing (Mikulecky, 2000, 2001). In order to clarify that the natural system also encompasses socio-technical systems, the term *real world system* will henceforward be used.

Events that lead to changes in the real world system are referred to as *causality* or *causal entailment* (Keirstead, 2014; Mikulecky, 2001). Our understanding of the world is then mapped into a formal system of our own making by a process referred to as *encoding* (Mikulecky, 2011). This formal system can then be manipulated in our minds in various ways in an attempt to mimic causal events observed or hypothesised in the real world system (Mikulecky, 2000, 2001). These manipulative changes in the formal system are referred to as *implication* or *inferential entailment* (Keirstead, 2014; Mikulecky, 2001). Once we have chosen a formal system and have found an implication in the formal system that corresponds to the causal event under consideration in the real world system, we can *decode* from the formal system in order to compare how well the changes in the formal system correspond with changes in the real world system (Mikulecky, 2001, 2011). The formal system is a *model* of the real world system if causal entailment yields the same result as the process of encoding, inferential entailment, and decoding (Keirstead, 2014).

Any attempt to make sense out of the world essentially boils down to an attempt to construct a successful modelling relation where the formal system is a model of the real world system (Mikulecky, 2001). In the Modern episteme, it is assumed that all of nature can be encoded into one exhaustive model (Mikulecky, 2000), which is based on a reductionist understanding of the world and is referred to as Newtonian Paradigm by Rosen (Mikulecky, 2001). In this world of simple mechanisms, the modelling relation may be forgotten and the formal system may be considered reality (Mikulecky, 2011). The epistemology (i.e. how we go about to produce knowledge about the real world system) spills over into an ontology (i.e. the specification of what we consider relevant to obtain knowledge about) (Mikulecky, 2001). In other words, ontology is reduced to man-made epistemology (i.e. the way how we produce knowledge about the real world system determines what we deem relevant in the real world system) (Birkin and Polesie, 2013).

When it was discovered that there are aspects of the real world system that the Newtonian Paradigm fails to capture, the study of complexity was born (Andersson et al., 2014; Mikulecky, 2001). In the view of Rosen (2000), complexity applies to things that cannot be modelled. Complexity thus is the property of a real world system manifest in the inability of any one formal system to adequately capture all its characteristics (Mikulecky, 2001).

Robert Rosen based his theory of modelling relation on the view that there is a fundamental duality between the external, objective world of phenomena, and the observer and his or her internal, subjective world (Mayumi and Giampietro, 2006). Any formal analysis of interactions in the external world of phenomena thus is one way of importing the external world of phenomena into the internal, subjective world of knowledge (Mayumi and Giampietro, 2006).

In a world where different observers and agents carry different beliefs, goals, and values, it is unavoidable that there can exist multiple, non-equivalent representations of the real world system (Giampietro et al., 2006). Every such representation reflects a different relevant reality, and is the result of two decisions: a decision about what are the relevant qualities and interactions of the real world system to address in a formal system, and a decision about how to formalise the representation of the selected qualities and interactions (Giampietro et al., 2006). Each relevant reality is thus described through a different simplification of an infinitely rich set of dynamics by means of a finite set of observable qualities and interactions (Giampietro et al., 2006).

2.1.5 Complexity and Wickedness

For Robert Rosen, complexity was not a matter of degree, as one can either model something or not (Allen and Giampietro, 2006). Robert Rosen suggested that complexity cannot be modelled because of uncertainty and contradiction (Allen and Giampietro, 2006). If uncertainty and contradiction are defined away, complexity is lost (Allen and Giampietro, 2006). This notion of complexity is at odds with what is called complexity in common parlance (Allen and Giampietro, 2006). In fact, a unified theory of what exactly complexity is seems to be lacking (Andersson et al., 2014; Mikulecky, 2001).

Zellmer et al. (2006) noted that the standard definitions of complexity in fact refer to the characteristics of a system after it has been simplified in order to be workable. A similar point is made by Andersson et al. (2014), who describe three classes of systems: complicated systems, complex systems, and wicked systems. Complicated systems exhibit low dynamical complexity but high structural complexity and are amenable to analysis using the toolbox of system based theories, whereas complex systems exhibit high dynamical complexity but low structural complexity and are amenable to analysis using the toolbox of mainstream complexity science (Andersson et al., 2014). In mainstream complexity science, complexity thus is synonymous with dynamical complexity (Andersson et al., 2014). The third class of systems, wicked systems, exhibits both high dynamical and high structural complexity, and includes ecosystems and societal systems (Andersson et al., 2014).

Complex systems as understood by Rosen and Zellmer et al. (2006) are in line with what Andersson et al. (2014) call wicked systems, but are at odds with complex systems in the parlance of mainstream complexity science. In this thesis, the parlance of complexity science is adopted, and complex systems as understood by Rosen and Zellmer et al. (2006) are referred to as wicked systems, as described by Andersson et al. (2014).

2.1.6 Dealing with Wickedness

Dealing with wicked systems requires that an infinitely rich set of dynamics in the real world system be simplified to a finite set of observable qualities and interactions (Giampietro et al., 2006). This simplification corresponds to the pre-analytical decision on what are the relevant qualities and interactions of the real world system to address in a formal system (Giampietro et al., 2006), and is also referred to as *narrative* (Allen and Giampietro, 2006; Andersson et al., 2014; Zellmer et al., 2006).

Narratives are devices for constructing mutual understanding between the storyteller and the listener, and make no pretence of being objective and unambiguous about the real world system (Allen and Giampietro, 2006). The analytical decision regarding how to formalise a real world system can be made only after a narrative has been chosen.

Wicked systems are particularly hard to address using formal approaches (Andersson et al., 2014), as there are several possible narratives and the choice of narrative may be contested. When attempting to deal with wicked systems through quantitative models based on a reductionist approach, there is a risk that the formalised system is an oversimplified and distorted picture of the real world system, particularly if the choice of narrative is influenced by the possibility of formalising the representation of the given qualities and interactions. It is in this light that the quality of computational models has been challenged (Bettencourt and Brelsford, 2015; Keirstead, 2014), and Giampietro et al. (2006) suggested two quality checks. First, the selected narrative must be relevant in relation to the beliefs and goals existing within a given knowledge system (i.e. the sum of knowledge held by different individuals and institutions). Second, within the chosen narrative, the selected formalisation must be scientifically accepted and the model results must be effective to guide action.

2.2 Material Flow Analysis (MFA)†

MFA is a systematic assessment of the stocks and flows of materials within a system defined in space and time, connecting the sources, pathways, intermediate sinks, and final sinks of a material. The term *material* comprises substances (e.g. phosphorus) as well as goods (e.g. wood). The analysis of flows of individual substances is also referred to as substance flow analysis (SFA). MFA can be used to analyse existing systems, but also to design new systems.

† This section is based on the "Practical Handbook of Material Flow Analysis" (Brunner and Rechberger, 2004). The interested reader is referred to this book for a more comprehensive treatment of MFA.

The mathematical core of MFA are mass balances (an application of the principle of mass conservation). Mass balances need to hold for every process and every material considered in an MFA (i.e. the mass of all inputs into a process must equal the mass of all outputs of this process plus a storage term that considers the accumulation or depletion of materials in the process). The results of an MFA are often visualised as a network of processes (nodes) and flows (edges) as exemplified in Figure 2.2.

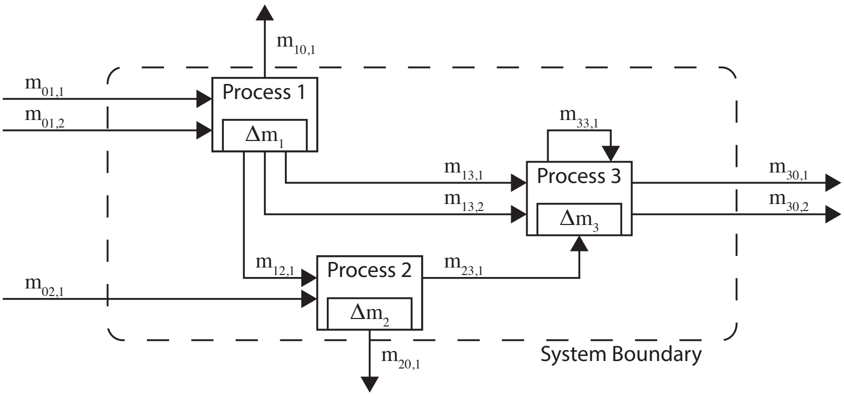

Figure 2.2: Connections between processes through flows as investigated in an MFA. The notation $m_{xy,z}$ indicates a mass flow: x indicates the source node (process x), y the target node (process y), and z the flow number (flow z from process x to process y). Note that process 0 stands for any process outside the system boundary.

The spatial system boundary of MFA is usually fixed by the geographical area in which the processes considered are located. The investigated system can be a company, a city, a region, a country, or the whole planet. Also more abstract systems can be investigated, such as an average private household or the waste management system of a given municipality. In theory, MFA delivers a complete and consistent set of information about all flows and stocks of a given material within the analysed system. In practice, completeness and consistency can often not be achieved due to system complicatedness and complexity as well as a lack of data.

2.3 Risk Assessment (RA)

RA is a broad assessment method applicable in many different contexts. Here, the discussion is limited to RA aiming to evaluate the risks posed by chemicals and pathogens to humans and other organisms in the environment. In this context, risk is commonly defined as a combination of the probability and the severity (nature and magnitude) of the effects resulting from a given hazard (stressor) and a given actual or proposed situation or action (Suter, 2006; Van Leeuwen and Vermeire, 2007).

Depending on its purpose, RA can cover one or several stressors, and one or several sites. RA conducted in the permitting process for a specific industrial activity could for instance encompass several stressors at one site but disregard these stressors elsewhere. RA could also encompass emissions of a chemical from different sources and at different locations throughout the life cycle of the chemical. It is also possible to adopt a product/service-based life cycle perspective in RA (Aissani et al., 2012; Dhingra et al., 2010; Kuczenski et al., 2011; Milazzo and Spina, 2015; Shih and Ma, 2011).

2.3.1 Chemical Risk Assessment (CRA)[‡]

The objective of CRA is to assess the risks associated with the emission of chemicals in terms of damage to human health or ecosystems. For the hazards identified (i.e. the chemicals of concern), CRA traditionally encompasses a comparison of exposure (i.e. the concentration or dose of the substance to which receptors are exposed) with effects (i.e. the highest concentration or dose at which no effects on the receptors are expected, or at which effects are considered acceptable). The remainder of this section focuses on human health effects, as damage to ecosystems was not considered in this thesis.

[‡] This section is largely based on the book "Risk Assessment of Chemicals: An Introduction" (Van Leeuwen and Vermeire, 2007). For a more comprehensive treatment of CRA, the reader is referred to this book.

Exposure Assessment

Exposure assessment is concerned with quantifying the concentrations of a chemical in the environment or the amount of a chemical (dose) a given receptor takes up in a given period of time. If exposure assessment encompasses chemicals in a given actual situation, the concentrations of the given chemical in different environmental compartments (e.g. soil) can be measured at different locations and points in time. If exposure assessment encompasses new chemicals or future situations, environmental concentrations cannot be based on measurements but can only be predicted. Such prediction requires knowledge about emissions (i.e. location of sources, magnitude of emissions), receptors (i.e. nature and size of the populations or compartments exposed, magnitude and duration of exposure), and the pathways the chemicals of concern can take from the sources to the receptors. This knowledge is often formalised in fate and exposure models. Many of the models used in CRA are compartment models, where the environment is assumed to be made up of homogeneous, well-mixed compartments. Various exposure routes can be considered individually or can be combined in order to determine a total intake for a given receptor.

Effect Assessment

Effect assessment, also referred to as dose-response assessment, provides the link between an exposure dose, and the incidence and severity of an effect. If a substance can produce different toxic effects, different dose-response relationships are required to represent the relationship between the degree of exposure to a substance and the extent of different toxic effects. Also, a distinction is usually made between threshold effects and non-threshold effects.

For threshold effects, adverse effects are assumed not to occur below a certain dose threshold. The standard approach for evaluating dose-response data for threshold effects has been the derivation of a no observed adverse effect level (NOAEL). NOAELs are then converted into predicted no effect levels (PNELs) by applying assessment factors reflecting the degree of uncertainty associated with the extrapolation from model organisms to humans and ecosystems.

For non-threshold effects, it is assumed that every single molecule of a substance has a very small probability of causing an adverse effect. The relationship between carcinogens and cancer is an example of this. The standard approach for evaluating dose-response data for non-threshold effects such as cancer effects has been to fit a dose-response model to the observations of toxicity studies.

Risk Characterisation

For threshold effects, risk characterisation consists of comparing the concentration or dose of an individual chemical, such as the predicted environmental concentration (PEC), measured environmental concentration (MEC), or predicted daily intake (PDI) with a threshold, such as the predicted no effect concentration (PNEC), acceptable daily intake (ADI), or tolerable daily intake (TDI) (see Figure 2.3). The rationale underlying this approach is that the manifestation of an observable effect is not acceptable. The ratio between exposure level and threshold is referred to as risk quotient (RQ) or hazard quotient (HQ). This RQ or HQ can then be compared with a maximum acceptable RQ or HQ, usually equal to 1, but possibly smaller than 1.

Due to the lack of a dose-threshold, risk characterisation of non-threshold effects can only try to determine a dose where the risk is acceptably small. The maximum permissible risk (MPR) is usually defined as the daily dose, taken during an entire lifetime, that is associated with a specified probability of incidence (e.g. cancer) (P_{inc}), for example 10^{-6}. Dose-response data usually originate from animal studies. As dose groups normally consist of about 50 to 100 animals per dose, only doses associated with risks on the order of 10^{-1} can be observed. This problem can be dealt with through the specification of a reference point (RP), for example a benchmark dose (BMD), such as the estimated dose at a 10% probability of effect (BMD10) or the lower confidence bound thereof (BMDL10). The predicted daily intake (PDI) can then be compared with this reference point (RP). The resulting ratio, referred to as margin of exposure (MOE), reflects the interval between the exposure dose and the dose with a known risk level. Another possibility is linear extrapolation from the reference point (RP). This linear extrapolation yields a probability of incidence (P_{inc}). For human receptors, at this point the concept of Disability Adjusted Life Years (DALY) (Murray, 1994) becomes useful as it enables the conversion from a measure of incidence to a measure of severity. The probability of incidence (P_{inc}) can be multiplied with the burden of disease per incident (BoD_{inc}) to yield a burden of disease (BoD). This BoD can then in turn be compared with a maximum acceptable BoD.

The different approaches to risk characterisation resulting from various approaches to exposure assessment and effect assessment are visualised in Figure 2.3 for both non-threshold effects and threshold-effects.

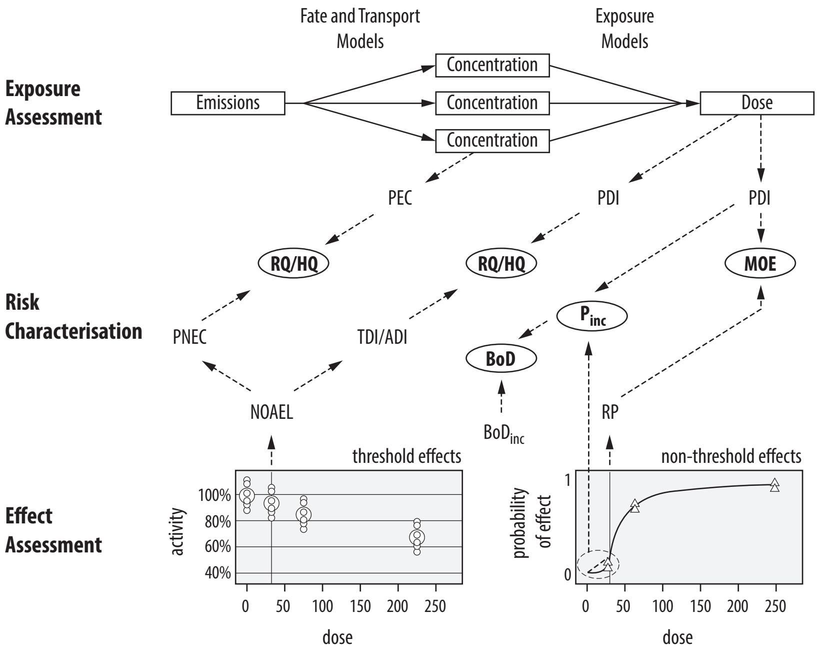

Figure 2.3: Visual summary of the relations between different approaches to exposure assessment and effect assessment.

2.3.2 Quantitative Microbial Risk Assessment (QMRA)[§]

The objective of QMRA is to evaluate the risks to human health resulting from the exposure to pathogens. For the hazards identified (i.e. the pathogens of concern that are to be assessed), QMRA traditionally encompasses the estimation of a probability of incidence or a health burden.

Exposure Assessment

Exposure assessment is concerned with quantifying the exposure of a receptor to a given pathogen. If exposure assessment encompasses a given actual situation, pathogen concentrations in the environment can be measured. If exposure assessment encompasses future situations, pathogen concentrations cannot be based on measurements but need to be predicted. Such prediction requires knowledge about sources (i.e. location and occurrence of pathogens), receptors (i.e. nature and size of the populations exposed, magnitude and duration of exposure), and the pathways pathogens take from the sources to the receptors. Just as for CRA, this knowledge is often formalised in transport, fate, and exposure models. Various exposure routes can be combined to determine a total dose for a given receptor. In summary, exposure assessment aims at estimating the pathogen dose a given receptor is exposed to in a given period of time, usually one day.

Effect Assessment and Risk Characterisation

Effect assessment, usually referred to as dose-response assessment, traditionally provides the link between the exposure dose and the associated probability of infection. The probability of infection corresponding with the exposure dose can then be compared with a maximum acceptable probability of infection, for example 10^{-6}.

[§] This section is largely based on the book "Quantitative Microbial Risk Assessment" (Haas et al., 2014) and a report on the use of QMRA in the water sector (Medema and Ashbolt, 2006). The interested reader is referred to these documents for a more comprehensive treatment of QMRA.

The diseases caused by different pathogens differ in nature, severity, and duration. Ideally, the relation between exposure dose and different health outcomes would be known. While there often is a clear relationship between exposure dose and probability of infection, the relationship between exposure dose and other health outcomes (specific symptoms or illnesses) is often not available or less clear. Therefore, the usual approach is to combine the dose-response relationships for infection with data on the fraction of the exposed population falling ill after infection as well as the occurrence of different health outcomes. Here, the DALY concept (Murray, 1994) is useful and enables the conversion of a probability of infection into a health burden, which can in turn be compared with a maximum acceptable health burden.

The overall structure of QMRA and the relation between exposure assessment and effect assessment are summarised visually in Figure 2.4.

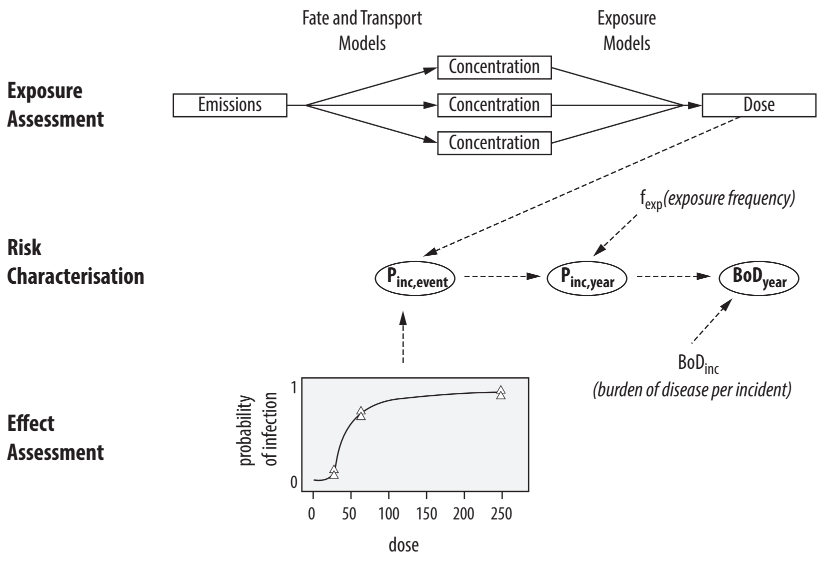

Figure 2.4: Visual summary of the structure of QMRA and the relations between the different steps of exposure assessment and effect assessment.

2.4 Life Cycle Assessment (LCA)[¶]

LCA is a tool for the assessment of products or services and attempts to cover the entire life cycle of a product or service. Emissions and resource consumption throughout this life cycle are quantified and associated with potential impacts on a number of safeguard subjects (e.g. human health, natural environment, and natural resources). This type of LCA is also referred to as environmental LCA in order to distinguish it from social LCA and life cycle costing (LCC). Initially, LCA was primarily concerned with energy resource management and impacts such as global warming and acidification (Oberbacher et al., 1996). Since its emergence in the late 1970s, LCA methodology has developed considerably, and several life cycle inventory (LCI) databases and LCIA methods have been made available. LCIA methods cover a continuously expanding number of impact categories and corresponding characterisation models for the conversion of emissions from, and resources used in, the life cycle of a product/service into impacts (Hauschild et al., 2013).

2.4.1 Four Stages of LCA

The procedure of performing an LCA is described in the ISO standards 14040 and 14044 and consists of four stages, which interact with one another in an iterative manner.

> *Goal and scope definition* is concerned with stating the intended application of the study, the reason for carrying it out, to whom and how the results are to be communicated, as well as a number of important modelling specifications including the functional unit (i.e. the reference flow to which all other modelled flows are related, for example the amount of water treated for wastewater treatment systems), system boundaries (spatial, temporal), cut-off criteria, allocation principles (in case of multi-input, multi-output or open recycling), and which options to model.

[¶] This section is, where not indicated otherwise, largely based on the book "The Hitch Hiker's Guide to LCA" (Baumann and Tillman, 2004) and the "ILCD Handbook: Framework and requirements for LCIA models and indicators" (JRC, 2010).

> *Inventory analysis* is concerned with quantifying environmentally relevant flows of matter and energy in relation to the selected functional unit. The result of inventory analysis, the life cycle inventory (LCI), consists of a number of flows (i.e. resource use and emissions to the environment) passing the system boundaries.

> *Impact assessment* (LCIA) is concerned with translating the resource use and emissions estimated in the inventory phase into environmental impacts.

> *Interpretation* is concerned with assessing results in order to draw conclusions.

2.4.2 Life Cycle Impact Assessment (LCIA)

This thesis is mainly concerned with LCIA. LCIA may consist of up to four steps: selection of impact categories and classification, characterisation, normalisation, and weighting.

> *Selection of impact categories and classification* means that emissions and resource consumption included in the LCI are assigned to one or several of the impact categories considered in the study.

> *Characterisation* refers to the actual quantification of the environmental impact of each emission or resource consumption.

> *Normalisation* refers to the association of characterisation results with a common reference. The environmental impact in a given impact category caused by the system under study can, for example, be divided by the total impacts of the given type in a given region, or the impacts of the given type caused by one person during one year in a given region. Normalisation facilitates comparisons across impact categories.

> *Weighting* is the process of determining the relative importance of different impact categories with respect to each other. Through weighting, individual impact categories can be merged into aggregated impact categories.

The case studies exploring the blending of elements of RA and LCA appended to this thesis relate particularly to the characterisation step of LCIA.

2.5 Relationships and Interactions between the Tools

2.5.1 Relationships between MFA, RA, and LCA

Each of the tools employed in this thesis involves the analysis of material flows. In MFA, the stocks and flows of materials are the end goal of the analysis. In RA and LCA, the flows of material form the basis of the subsequent quantitative assessment of effects associated with these flows.

Of course, the object of study has traditionally been different for each of the three tools. LCA is the only tool with a strict definition of the object of study, namely the life cycle of a product or service. In MFA and RA, the choice of the object of study is more flexible. The object of study can be the life cycle of a product of service, but usually is different.

2.5.2 Interactions between RA and LCA

RA and LCA are two tools used to support decision making and were initially developed and used by separate groups of specialists (Cowell et al., 2002). The similarities, differences, and synergies between RA and LCA regarding the assessment of the effects of chemical pollutants on humans and other organisms have been thoroughly discussed in a special issue of *Risk Analysis* in 2002 (Cowell et al., 2002; Hofstetter and Hammitt, 2002; Hofstetter et al., 2002; Matthews et al., 2002; Tukker, 2002), a special section in *Human and Ecological Risk Assessment* in 2006 (Bare, 2006; Udo de Haes et al., 2006; Pennington et al., 2006; Russell, 2006; Socolof and Geibig, 2006; Sweet and Strohm, 2006), and in regular issues of various other journals and reports (Boize et al., 2008; Klöpffer, 2010; Olsen et al., 2001; Owens, 1997; Sleeswijk et al., 2003) as well as in technical reports (Flemström et al., 2004).

A particularly useful analysis was conducted by Udo de Haes et al. (2006), who discussed similarities, differences and synergies between RA and LCA at three levels: basic equations, overall model structure, and applications. These three levels form the basis for the summary of the relationships and interactions between RA and LCA presented in the remainder of this section.

Level 1: Basic Equations

The first level of analysis suggested by Udo de Haes et al. (2006) concerns the basic ingredients of the two analytical tools: the environmental processes and phenomena that are to be incorporated, the mathematical relationships postulated for each of these phenomena, and the chemical and environmental data needed in these relationships.

Early attempts to integrate human toxicity and ecotoxicity into LCA (Guinée and Heijungs, 1993; Guinée et al., 1996; Heijungs et al., 1992; Hertwich et al., 1996; Keller et al., 1998) were to a large extent inspired by expert knowledge and mathematical relationships borrowed from CRA. The transfer of ideas and concepts encompassed fate and exposure assessment (Schulze et al., 2001), as well as effect assessment for human toxicity and ecotoxicity (Larsen and Hauschild, 2007; Pennington et al., 2004, 2006). The inclusion of fate and exposure modelling for chemicals in LCA led to the further development of multimedia fate and exposure models for use in both RA and LCA (Birkved and Heijungs, 2011; Huijbregts et al., 2005; Humbert et al., 2009; Pizzol et al., 2012; Strandesen et al., 2007). Furthermore, the results obtained from LCIA models were compared with those obtained from existing RA models, or expert judgment by RA practitioners, for validation purposes (Demou et al., 2011; Dijkman et al., 2012; Lim et al., 2011; Mattila et al., 2011; Pant et al., 2004; Tolle et al., 2001). As a result, a number of LCIA models for the integration of human toxicity and ecotoxicity into LCA are now available via commercial LCA software packages. Early attempts to integrate the effect of pathogens on human health into LCA (Aramaki et al., 2006; Larsen and Hauschild, 2007) were based on QMRA. These early attempts inspired further efforts to integrate results obtained through QMRA in an LCA framework (Harder et al., 2014; Heimersson et al., 2014; Kobayashi et al., 2015).

Consequently, at the level of the basic equations, few differences are found between RA and LCA, as RA and LCA essentially use the same fate, transport, exposure, and effect models (Udo de Haes et al., 2006). Differences mainly concern the selection of exposure pathways and the choice of parameter values. These differences belong to the level of the overall model structure.

Level 2: Overall Model Structure

The second level of analysis suggested by Udo de Haes et al. (2006) concerns the overall model structure, where a range of differences can be found that distinguish RA and LCA. RA usually seeks to provide a realistic worst-case estimation for toxicity effects related to critical exposure pathways, receptors, and effects. LCA is usually oriented towards providing a median toxicity indicator related to a broad range of exposure pathways, receptors, and effects. RA usually considers the actual emission characteristics. LCA usually treats emissions as pulses scaled to the functional unit. Also, the degree of spatial and temporal differentiation used to be a difference in the overall structure of RA and LCA. The absence of spatial and temporal differentiation in many LCA studies is mostly for pragmatic reasons (Cowell et al., 2002; Matthews et al., 2002). Simplifications are often required in order to make LCA more tractable, as increased model detail usually would need more extensive data gathering (Matthews et al., 2002; Peters et al., 2014). Spatial and temporal differentiation is considered one way of deepening LCA (Assies, 1998; Jeswani et al., 2010).

The recognition of the potential importance of spatial and temporal differentiation has led to the development of LCIA models that take specific regional characteristics into consideration (Potting and Hauschild, 2006), such as spatially differentiated fate and exposure models for human toxicity and ecotoxicity (Kounina et al., 2014; Wegener Sleeswijk and Heijungs, 2010). Also, there have been efforts towards considering time in LCA through a dynamic LCA approach, where time-dependent emissions and characterisation factors are applied (Kounina et al., 2014; Levasseur et al., 2010). Spatial and temporal differentiation in LCIA models has been accompanied by efforts towards spatial and temporal differentiation in LCI (Collet et al., 2014; Manneh et al., 2012; Pinsonnault et al., 2014; Steinberger et al., 2009). It should be borne in mind, however, that the extent to which spatially and temporally disaggregated data is needed in a particular LCA study is a consequence of the role and goal of the given study (JRC, 2010; Sandin et al., 2014), and that some impact categories may require a more disaggregate approach to modelling of cause-effect chains than others (Matthews et al., 2002).

In the field of contaminated site remediation, for example, it was recognised that the estimation of the environmental impacts of contaminant leaching would require site-specific fate, transport and exposure models (Lemming et al., 2012; Sparrevik et al., 2011). The importance of including site-specific fate, transport, and exposure models for chemical toxicity was also acknowledged in other contexts, including water and wastewater treatment (Muñoz et al., 2009; Ribera et al., 2014), and land application of sewage sludge (Sablayrolles et al., 2010).

Level 3: Applications

The third level of analysis suggested by Udo de Haes et al. (2006) concerns the application of RA and LCA in environmental assessment case studies, where RA and LCA can complement each other.

One reason behind the efforts to integrate RA and LCA in environmental assessment case studies is to overcome the respective blind spots of the individual analytical tools. The focus of RA on comparing the predicted exposure of certain receptors to certain stressors with corresponding thresholds makes RA blind to the overall impacts related to the life cycle of a product or service. Similarly, the focus of LCA on comparing different product or service systems (or different life cycle phases of a product or service system) makes LCA blind to the time-space distribution of stressors, the existence of toxicity thresholds, and varying acceptability of impacts.

In the field of contaminated site remediation, which traditionally has been more concerned with the mitigation of local risks and impacts, it has been recognised that the consideration of life cycle impacts of the risk mitigation measures may avoid problem shifting (Lemming et al., 2010; Sanscartier et al., 2010; Volkwein et al., 1999). Problem shifting may occur also in the context of water and wastewater treatment, where the removal of chemicals and pathogens through treatment operations may reduce local health risks at the expense of environmental impacts caused elsewhere as a result of for instance energy production required for treatment operations (Jones et al., 2007; Papa et al., 2013; Ribera et al., 2014; Wenzel et al., 2008).

Volkwein et al. (1999) used LCA to find the best among a series of remediation options that had been shown to be financially, legally, and technically possible. That is, LCA was performed for the options considered acceptable based on RA. Other studies (Godin et al., 2004; Socolof and Geibig, 2006) suggested the reverse, namely that LCA can be a screening tool to identify whether an RA of selected processes or scenarios should be performed. The combination of the perspectives provided by RA and LCA, however, is limited neither to sequential use of RA and LCA, nor to the contexts of site remediation and water treatment.

An early example of the parallel use of RA and LCA is a retrospective comparison of the environmental profile of laundry detergents at different points in time (Saouter et al., 2001, 2002), where RA yielded risk quotients (regarding the aquatic compartment) and LCA yielded overall environmental impact scores for the respective granular laundry detergents (covering seven impact categories). Other contexts where RA and LCA were used in a parallel manner include bleaching of mechanical pulp (Scheringer et al., 1999), building insulation products (Schmidt et al., 2004), a surface-active adhesion promoter used in hot mix asphalt pavement (Manuilova et al., 2005), contaminated site remediation (Cappuyns and Kessen, 2014; Inoue and Katayama, 2011; Lemming et al., 2010), solid waste management (Benetto et al., 2007; Carpenter et al., 2007; Liu et al., 2012; Song et al., 2015; Venturini et al., 2014), metal degreasing (Kikuchi et al., 2011; Kikuchi and Hirao, 2008, 2010), toluene emissions along the life cycle of printed matter (Walser et al., 2014), crop protection systems (Mouron et al., 2012), the development of a two-component adhesive (Askham et al., 2013), the production of an alumina nanofluid (Barberio et al., 2014), or ecodesign of a beverage bottle (Herva et al., 2012).

Most studies reported the results obtained through the use of the two tools separately with the intent to broaden decision support. Some studies used multicriteria analysis methods in order to weight and aggregate the results obtained through RA and LCA. These methods range from simple ranking (Schmidt et al., 2004) or the use of weighting factors (Cappuyns and Kessen, 2014) to more complicated approaches such as fuzzy logic (Herva et al., 2012;

Liu et al., 2012), the creation of new evaluation indices (Inoue and Katayama, 2011) or impact categories (Benetto et al., 2007), and hierarchichal attribute trees (Mouron et al., 2012).

Apart from the combination of the perspectives provided by RA and LCA in environmental assessment case studies, the importance of the combination of multiple perspectives (including perspectives other than RA and LCA, such as MFA, LCC, system dynamics, or cost benefit analysis) has also been highlighted from a more theoretical standpoint for sustainability assessment in general (Ness et al., 2007) as well as for a number of specific contexts, such as remediation of contaminated sites (Jordan and Abdaal, 2013; Morais and Delerue-Matos, 2010), waste management (Tiruta-Barna et al., 2007), industrial production and product development (Herrchen and Klein, 2000; Herva and Roca, 2013; Herva et al., 2011), wastewater management (Chen et al., 2012), nanomaterials (Breedveld, 2013; Grieger et al., 2012; Meyer and Upadhyayula, 2014; Som et al., 2010), or chemical regulation (Askham, 2012).

Other studies have discussed potential contributions of RA and LCA (as individual tools) to environmental impact assessment (Liu et al., 2013), strategic environmental assessment (Finnveden et al., 2003; Loiseau et al., 2012), comprehensive environmental assessment (Davis and Thomas, 2006; Powers et al., 2012), or multicriteria decision analysis (Linkov and Seager, 2011; Palme et al., 2005). Yet other studies have investigated how an appropriate data structure can facilitate integration of RA and LCA (Ingwersen et al., 2015; Kikuchi and Hirao, 2009a,b; Kikuchi et al., 2012; Kuczenski et al., 2011).

Chapter 3

Summary of Thesis Contributions

This chapter provides the background to the five appended papers and discusses their main findings in relation to the research questions formulated in chapter 1. Recommendations will be provided in chapter 4.

3.1 Flows and Sinks of Phosphorus in Gothenburg

> *Which are the key flows and sinks of phosphorus in Gothenburg under the current and possible alternative wastewater and sludge management systems?*

Phosphorus is an element of paramount importance as a nutrient to safeguard food-supply (Cordell et al., 2009). Contemporary industrialised agricultural production relies heavily on mineral fertilisers that, amongst other ingredients, contain phosphorus mined from mineral phosphorus deposits (Neset and Cordell, 2012). Many current agricultural, waste management, and wastewater management practices lead to a dispersion of phosphorus to different sinks. Ott and Rechberger (2012), for instance, mapped flows and sinks of phosphorus in the European Union (EU15) and found that the main sinks of phosphorus are agricultural soils, followed by landfills and the hydrosphere. The issue of phosphorus could be said to embrace both scarcity and overabundance of a key nutrient (Lougheed, 2011).

As mineral phosphorus reserves are finite, the international community is investigating how to more effectively manage phosphorus in the future. Paper I focuses on phosphorus flows in urban areas. The overarching objective of the underlying case study was to map the flows and sinks of phosphorus in Gothenburg for the current and possible future wastewater and sludge management systems. Gothenburg was chosen due to ongoing discussions among stakeholders regarding potential changes in the municipal waste management infrastructure towards more favourable pathways for nutrients.

The first part of the study consisted of a quantification of phosphorus flows within and across the geographical border of the municipality of Gothenburg for the reference year 2009 through MFA. The choice of MFA was self-evident given the nature of the question at hand. The collection areas for the main municipal wastewater treatment plant and municipal solid waste treatment plant located within the geographical borders of the municipality of Gothenburg extended well beyond the municipality's geographical borders. Five adjacent municipalities were partly connected to the wastewater treatment plant, and solid waste was collected from ten neighbouring Swedish municipalities and sometimes even imported from Norway. The data obtained from the treatment works hence required disaggregation into a local component (municipality of Gothenburg) and an imported component (inputs from adjacent municipalities and Norway). Although MFA, in theory, delivers a complete and consistent set of information about all flows and stocks of a particular material within a given system (Brunner and Rechberger, 2004), completeness and consistency can often not be achieved in practice. In particular, the estimation of the amount of phosphorus contained in goods had to be restricted to a limited number of goods, as for some types of goods, neither supply nor demand estimates were available. Furthermore, data on the origin and composition of solid waste were hard to interpret due to a partial lack of information on solid waste pathways and inconsistent usage of the term household waste. The MFA showed that the amount of phosphorus found in wastewater and solid waste, respectively, was of the same order of magnitude. A substantial fraction (60%) of the phosphorus entering the municipality of Gothenburg was contained in food commodities.

The phosphorus contained in the incineration ashes of the solid waste treatment plant stemmed predominantly (55%) from food commodities that were wasted rather than consumed, while for a significant fraction of the phosphorus (38%), no source could be identified. Phosphorus recycling to productive land was marginal at the time, as sewage sludge was mostly used as backfilling soil at construction sites and for other civil engineering restoration activities, and the incineration ashes from the solid waste treatment plant were put on landfill.

The second part of the study consisted of an evaluation of five alternative waste management strategies with regard to their impact on flows and sinks of phosphorus: (1) all food waste is incinerated at the solid waste treatment plant (i.e. no separate collection of food waste), (2) separate collection of 70% of the food waste from households and businesses for separate treatment, (3) installation of kitchen grinders (i.e. food waste is diverted to the wastewater treatment plant), (4) urine diversion, (5) separation of blackwater and food waste for separate treatment. The different scenarios differ significantly with regard to the type of fertilising product for use in agriculture that can be obtained, while the amount of phosphorus that can be recovered is of the same order of magnitude for all scenarios. In any case, Paper I indicated that research and policy initiatives for comprehensive recycling of phosphorus should preferably address both wastewater management and solid waste management, as food waste is a significant source of phosphorus in urban areas, and can be discharged through either of the two waste management infrastructures.

3.2 Blending RA and LCA in Case Studies

> *How can RA and LCA be meaningfully blended in order to better assess wastewater and sludge management options from an environmental and human health perspective?*

The expression *hybridisation of RA and LCA* was initially used in the LiCRA project group to refer to the combination and integration of RA and LCA. Following a discussion about what the term *hybridisation* actually means in this context, the idea for Paper II was born.

The Scopus database was searched in March 2014 for case studies mentioning both RA and LCA in the title, abstract, or keywords. The literature search was reiterated in October 2014 and June 2015 to also include more recent case studies. The literature search yielded 30 case studies that claimed to combine or integrate RA and LCA in one way or another. These case studies were subjected to a critical analysis based on three overarching questions: what methodological characteristics distinguish environmental assessment case studies claiming to combine or integrate RA and LCA, has a unified terminology emerged, and what can be learnt for future environmental case studies attempting to combine or integrate RA and LCA? The focus of the analysis hereby was on how the respective case studies evaluated emissions of chemical pollutants and pathogens.

The analysis revealed three clusters of similar case studies: case studies aiming at a site-dependent assessment of human health risk, studies applying life cycle thinking to RA, and studies aiming to explicitly address the trade-off between local and global effects. The analysis also revealed inconsistent use of terminology, particularly regarding what is meant by *integration*, *combination*, *hybridisation*, or *integrated use* of RA and LCA. The expression *hybridisation of RA and LCA* was therefore abandoned in favour of the expression *blending elements of RA and LCA*, or simply *blending RA and LCA*.

Several case studies blended RA and LCA by combining the perspective traditionally offered by RA (i.e. risks for specific members of a given human population) and the perspective traditionally offered by LCA (i.e. overall impacts for a given human population). These case studies often aimed at addressing trade-offs between global impacts and local risks or impacts. Paper II argues that combining the perspectives provided by RA and LCA is the only way of blending elements of RA and LCA that actually warrants to be referred to as combination or integration of RA and LCA. Hereby, the results of the RA and LCA perspective can be reported separately or used as input for multicriteria analysis methods. Other case studies blended RA and LCA by transferring elements (i.e. the environmental processes and phenomena to be incorporated, the mathematical relationships postulated for these processes and phenomena, and the chemical and environmental data needed in these relationships) from one perspective to the other. For instance, elements of QMRA, which usually provides results representing the RA perspective, can be used to provide results representing the LCA perspective on the effect of pathogens on human health. Or, site-specific fate and transport models usually applied when addressing the RA perspective can be adapted to improve LCIA for certain emissions when addressing the LCA perspective. It is these latter ways to blend RA and LCA that give rise to a number of issues.

If a given study, within the LCA perspective, applies site-specific fate and transport models for certain emissions but site-generic fate and transport models for other emissions of the same type, this model asymmetry may introduce bias and run counter to the initial idea of LCA to treat impacts at different locations in a similar way. Moreover, assumptions and parameter choices may be inadvertently transferred from one perspective to the other. If elements of RA, for instance, are adopted in the LCA perspective, it may happen that the choice of exposure pathways and exposure conditions (at least for some exposure pathways) within a given impact category reflects periods of non-routine operations under realistic worst-case assumptions, while the remaining category indicators reflect routine operations. These two issues at the same time provide further opportunities, namely the consideration of non-standard operation scenarios and disparate subpopulation effects in LCA.

In order to facilitate a better understanding and more transparent communication of the nature of a given case study, Paper II proposed a common design space, similar to the evaluative framework for the evaluation of conceptual and analytical approaches used in environmental management proposed by Baumann and Cowell (1999) and the identification key for selecting methods for sustainability assessments proposed by Zijp et al. (2015).

3.3 Accounting for Pathogens in LCA

How can the human health effects related to pathogens present in wastewater and sludge be included in LCA?

The risks posed by pathogens present in wastewater and sludge are commonly quantified through QMRA. There are currently no standardised LCIA models available to account for pathogens in LCA.

The study reported in Paper III aimed to estimate the potential impact of pathogens on human health for subsequent comparison with other potential impacts on human health associated with a given wastewater and sludge management option and commonly accounted for in LCA. The calculations represented a model wastewater treatment system serving 28600 individuals, treating 12500 m^3 wastewater per day on average, and consisting of primary and secondary treatment (activated sludge process with biological nitrification-denitrification and chemical phosphorus removal), tertiary treatment (constructed wetland), anaerobic sludge digestion, and land application of sewage sludge. In order to maximise the number of different combinations of pathogen and exposure pathway included in the estimation, the study combined a QMRA model starting from pathogen concentrations in raw wastewater, treatment plant effluent, raw sewage sludge, and treated sewage sludge with the results from previously published QMRA studies on similar wastewater treatment systems. The total pathogen-related burden of disease for the model wastewater and sludge treatment system considered was estimated to be of the order of 0.2-9 DALY per year of operation for the population served. This estimate was compared to other impacts on human health considered in LCA for a similar wastewater and sludge treatment system (Paper A).

Paper III is one of the pioneering papers exploring the integration of QMRA into an LCA framework. In hindsight, it became apparent that the assumptions regarding exposure conditions were directly transferred from an RA framework into an LCA framework. As discussed in Paper II, this may be problematic. Also, shortly after the publication of Paper III, we were approached by a researcher who expressed concerns about the loss of relevant information when QMRA results are aggregated and presented in an LCA framework. In particular, the question arose whether QMRA results can be scaled to a functional unit, and if so, whether some functional units are preferable to others. The ensuing discussions resulted in Paper B.

Paper IV is a continuation of the work presented in Paper III and explored whether the mathematical equations and model structure used in QMRA can be simplified if the results are to be used in an LCA framework. The modifications investigated were the linearisation of dose-response relationships, the linearisation of severity assessment relationships, and the adaptation of the overall model structure.

The work on including human health effects related to pathogens in LCA (Papers III and IV) led to a number of important insights, which have been discussed in detail in Paper II and Paper B. Human health effects related to pathogens can be addressed in a meaningful way in LCA, and the mathematical relationships and data used in QMRA can be of use. But it is of crucial importance to keep in mind that the rationale traditionally followed by QMRA is to identify the exposure pathways and parameter choices with the highest effect per individual exposed. In an LCA framework, this approach may be inappropriate, as a potentially larger number of people may be exposed through a less critical exposure pathway, leading to a potentially higher overall impact. If the LCA aims to reflect routine operations, it might be inconsistent to consider exposure pathways reflecting periods of non-routine operations, or to parameterise exposure pathways using realistic worst-case assumptions common in QMRA. When accounting for pathogens in LCA, it therefore is important to make sure that exposure pathways and parameters are chosen in accordance with the principles applied in the LCA study of which the assessment of human health effects related to pathogens is a part. Particular attention is warranted regarding exposure conditions such as pathogen decay time, the number of people exposed, and the frequency of exposure. Furthermore, upon aggregation and scaling to a functional unit, details regarding individual exposure pathways and subpopulation effects are masked. In fact, the human health effects related to pathogens may still be higher than acceptable for certain individuals in the population, even in case the human health effects related to pathogens turn out to be small compared to other LCA derived health effects (where the burden of disease may be distributed to a larger number of individuals).

3.4 Context-dependent LCIA

> *Does a context-dependent LCIA model to assess the human health effects related to chemical contaminants present in sludge applied to agricultural land provide significantly different results compared to a generic LCIA model?*

From the LCA perspective, an interesting and relevant question is whether the overall impact on human health related to chemical contaminants for one sewage sludge management option (e.g. application of sewage sludge to agricultural land) is greater or lesser than for another sewage sludge management option (e.g. incineration of sewage sludge with subsequent phosphorus recovery). LCA studies addressing the human toxicity potential related to chemical contaminants in sewage sludge often rely on existing LCIA models such as USES-LCA (Heimersson et al., 2014; Hospido et al., 2010; Lane et al., 2015) and USEtox (Yoshida et al., 2014), or models tailored to a given specific context (Nakakubo et al., 2012; Sablayrolles et al., 2010).

Paper V investigated the generic LCIA models USEtox 1.01 and USEtox 2.0 as well as a context-dependent LCIA model tailored to the context of land application of sewage sludge. The context-dependent model was largely based on the mathematical relationships described in the European Chemicals Bureau Technical Guidance Document on Risk Assessment (European Chemicals Bureau, 2003). The calculations were performed for 15 metals and 106 organic contaminants.

The LCIA models investigated in Paper V (i.e. USEtox 1.01, USEtox 2.0, and a context-dependent LCIA model) provided different burden of disease estimates for individual chemical contaminants, but an aggregate burden of disease estimate of the same order of magnitude. In light of these findings, context-dependent LCIA models for human toxicity potential appeared of lesser importance in the context of land application of sewage sludge, at least as long only routine exposure through agricultural produce, air, and water is considered. If treatment operations often do not proceed according to design specifications, accidental ingestion of sewage sludge may become relevant.

Chapter 4

Discussion and Outlook

In the previous chapter, the thesis contributions were discussed in relation to the four research questions underlying this thesis. This final chapter aims to provide useful recommendations regarding the quantitative assessment of wastewater and sludge management options. In this regard, three issues need to be considered before any recommendations can be given.

The first issue concerns the nature of sewage sludge management, more specifically whether or not sewage sludge management belongs to the domain of wicked problems. The second issue concerns the choice of assessment methods, which can be challenging and might be guided by the expertise of the analyst rather than the requirements of the decision-making context (Zijp et al., 2015). The third issue concerns how one knows when to make what recommendations.

The chapter is concluded by recommendations and a set of ideas for further reflection and research.

4.1 Preparing the Ground for Giving Recommendations

4.1.1 Sewage Sludge Management - Wicked or Not?

As previously outlined in section 2.1.5, systems can be complicated (a structural quality) or complex (a dynamical quality), or both (Andersson et al., 2014). Wicked systems are characterised by both complicatedness and complexity (Andersson et al., 2014), and correspond with Rosen's notion of complexity.

Andersson et al. (2014) suggest that societal systems and ecosystems are prime examples of wicked systems, as complexity and complicatedness can be seen as mutually reinforcing in societal systems and ecosystems.

Sewage sludge management is intimately connected with carbon and nutrient cycles as well as agricultural production, and therefore has a significant impact on ecosystems. Moreover, there are many interconnections between sewage sludge management and other features of modern socio-technical systems. For example, society's use of chemicals and pharmaceuticals influences the quality of sewage sludge and the solution space when it comes to the recycling of nutrients to productive land. Also, sewage sludge management can aggravate or mitigate phosphorus scarcity. Given the global nature of nutrient cycles and the long lifetime of urban water infrastructures, system uncertainties and decision stakes are high, which make sewage sludge management belong to the domain of post-normal science. For all these reasons, sewage sludge management is considered to be a wicked problem belonging to the domain of wicked systems.

4.1.2 LCA - Suitable or Not?

As previously outlined in section 2.1.4, the adequacy and usefulness of quantitative assessment tools to deal with wicked systems has been questioned. One important reason for this reservation towards formal approaches to wicked systems is that the simplifications required for formalisation fundamentally imply an assumption of non-wickedness (Andersson et al., 2014). That is, formal approaches can only provide a limited snapshot of the system in question, providing one out of a larger set of multiple, non-equivalent representations of the real world system under consideration.

LCA is a tool that essentially casts wicked systems as complicated systems. This means that one may get spurious results where the assumptions needed to simplify the system mean poor realism of the formalisation (Andersson et al., 2014). Therefore, it comes as no suprise that uncertainty and wickedness are among the main challenges discussed in the literature, and authors like Lundie et al. (2008) recommend employing LCA in decision-making processes that elicit stakeholder narratives. To these challenges, this thesis adds the choice of functional unit in LCA.

Uncertainty

There are several different kinds of uncertainty, and all uncertainty cannot be captured quantitatively (Bradley and Drechsler, 2014). For the following discussion, the distinction between quantifiable (measurable) uncertainty and unquantifiable (unmeasurable) uncertainty suggested by Knight (1921) is particularly useful.

In RA, it is common to distinguish between variability (i.e. quantifiable uncertainty that cannot be reduced by further research and measurements) and uncertainty (i.e. quantifiable uncertainty that can be reduced by further research and measurements) (Van Leeuwen and Vermeire, 2007). Also in LCA, the necessity to deal with uncertainty has long been recognised (Cowell et al., 2002; Hertwich et al., 2000; Hofstetter et al., 2000; Huijbregts, 1998a,b; Ross et al., 2002; Steen, 1997; Tukker, 2000). To address the role of value choices, Hofstetter et al. (2000), for instance, suggested to conduct several structurally identical types of LCA, each based on a coherent but different set of values. Other efforts include the quantification of parameter uncertainty (Ciroth and Srocka, 2008; Ciroth et al., 2013). Hofstetter et al. (2000) even attempted to include a proxy indicator for unknown damage into LCA, which in principle is an attempt to estimate the uncertainty associated with something that is not quantifiable at the time of the analysis, but would in principle be quantifiable provided enough information were available.

Quantifiable uncertainty can be accounted for in RA and LCA, at least to a certain extent. The data gaps discussed in Paper V, for instance, all are quantifiable in principle. This means that the data gaps could be closed in theory, even if the research efforts needed to close these gaps in practice probably are prohibitive, leading to quantifiable but unquantified uncertainty. The main concern with RA and LCA, and any other quantitative assessment tool, is with quantifiable but unquantified uncertainty as well as with unquantifiable uncertainty. Funtowicz and Strand (2011), for instance, highlighted that the results of state-of-the-art risk assessment and risk management procedures may be invalidated within a few years upon the discovery of novel, unforeseen and surprising consequences of a given action. The same applies also to LCA and other quantitative assessment tools.

Wickedness

The issue of unquantified and unquantifiable uncertainty is closely related to the nature of wickedness and may hamper the usefulness of quantitative assessment tools such as RA and LCA in the face of wicked systems. Bettencourt and Brelsford (2015) noted that simple engineering approaches aiming at optimal solutions might not work in all contexts, as some problems are intractable given that the required information is distributed across many different entities. Andersson et al. (2014) argued that tools adequate for purely complicated systems (i.e. systems displaying structural but not dynamical complexity) or complex systems (i.e. systems displaying dynamical but not structural complexity) are inadequate for wicked systems (i.e. systems displaying both structural and dynamical complexity), as these tools rely on simplifications of either the structure or the dynamics of the system, which frequently means that an acceptable levels of realism cannot be maintained.

But what if approaches that successfully deal with complicated systems were combined with approaches that successfully deal with complex systems? Andersson et al. (2014) argue that also such a combination of tools would be of no help, because each respective tool derives its power not from the presence of either complicatedness or complexity, but rather from the lack of either complicatedness or complexity (Andersson et al., 2014). Combined approaches may thus actually combine the weaknesses rather than the strengths of the constituent approaches (Andersson et al., 2014).

At the same time, formal approaches are absolutely necessary, because the ability of humans to make strict inferences and abstractions without the use of mathematics and formal logic is poor (Andersson et al., 2014). Simulation models are particularly useful due to the flexibility they provide to address different scenarios (Andersson et al., 2014). However, it is important that narrative and formal approaches are used in conjunction. For some problems, sophisticated computational models may not always be the right choice, and a simplified model may be more useful (Keirstead, 2014).

Functional Unit

Let us turn our attention to a third important issue, the choice of functional unit in LCA. For sewage sludge management options aiming to supply nutrients to agriculture, a possible functional unit is the production or recycling of a certain amount of phosphorus (Kalmykova et al., 2015), or the supply of a certain amount of phosphorus to productive land (Linderholm et al., 2012). These functional units do not distinguish between the supply of phosphorus in the form of mineral fertiliser and the supply in the form of organic fertiliser. Certainly, if soil is seen as merely a substrate wherein to grow agricultural produce, this distinction is indeed of lesser relevance. But what if soil were not seen as merely a substrate? What if the importance of soil health was recognised (Cardoso et al., 2013), and soil health and soil degradation were consequently taken into account? It would go beyond the scope of this thesis to discuss the issue of soil health and soil degradation. The LCA community is aware of the need for soil quality to be considered more comprehensively in the future, and dedicated a one-day workshop at the 2015 Life Cycle Management conference to discuss soil quality indicators in LCA. More generally, the issue of soil health is meant to illustrate that the choice of functional unit is neither trivial nor arbitrary.

Say we perform an LCA study to evaluate different sewage sludge management options that return phosphorus to productive land. If our starting point were that nutrient supply through mineral fertiliser and organic fertiliser are equivalent, we might choose the functional unit as supplying a certain amount of phosphorus. With this choice of functional unit, it would be possible to compare sewage sludge management options such as land application of sewage sludge and the production of mineral fertiliser based on phosphorus recovered from incineration ash. Such a comparison may become invalidated, however, if our starting point were that nutrient supply through mineral fertiliser and organic fertiliser are not equivalent. Certainly, the impact of different types of fertiliser on soil quality could be taken into account through one or several impact categories. Yet, it may be preferable to choose a different functional unit in order to give the issue of soil quality more prominence.

For example, the functional unit could be chosen as to supply a given amount of phosphorus in a manner that does not compromise soil life, or in a manner that contributes to the regeneration rather than the degeneration of soil. Such a functional unit would correspond with a narrative that has a strong focus on soil quality and the insight that all forms of fertiliser may not have the same impact on soil quality and soil life. Admittedly, such functional units may be hard to operationalise.

The main point with above example is that the choice of narrative and the choice of functional unit are related. The functional unit is not an inherent property of the real world system but something that is decided upon when encoding the real world system into a formal system. In other words, the concept of functional unit in LCA applies to the internal, subjective world of the observer, that is, a given representation of a given relevant reality. But the specification of a functional unit does not automatically imply functional equivalence in the external, objective world of phenomena. The concept of functional unit in LCA thus simply means that the systems are functionally equivalent in terms of the aspects considered relevant according to the chosen narrative. This is why the choice of functional unit is related to the choice of narrative, and the choice of functional unit is neither trivial nor arbitrary.

4.1.3 On Making Recommendations

Before making recommendations based on quantitative assessment tools such as RA and LCA, it is important to distinguish two types of recommendations. This distinction is illustrated for the decision-making context of sewage sludge management.

Say an LCA study is performed to evaluate different options to return phosphorus to productive land. The pre-analytical decision of what qualities and interactions are relevant may point towards a range of indicators (e.g. global warming potential, acidification potential, human toxicity potential, pathogen impact potential) that are assumed to be relevant and should be considered from a life cycle perspective. The analytical decision then is concerned with how to formalise this narrative in a concrete LCA study.

Recommendations that directly target the decision-making context span both the choice of narrative and the formalisation of the real world system. Their validity hence crucially depends on the choice of narrative. This type of recommendation will henceforward be referred to as a *narrative-sensitive* recommendation. Say a given sewage sludge management option is recommended based on a given LCA study. If a different narrative were chosen to simplify the wicked system under consideration, this recommendation may simply break down. For instance, if it became evident that certain qualities and interactions not deemed relevant by the chosen narrative actually were relevant nonetheless, the recommendation might be invalidated. Given the existence of multiple, non-equivalent representations of the real world system, narrative-sensitive recommendations are particularly susceptible to contestation.

Recommendations relating to how to best perform a given LCA study, however, are contained within the boundaries of the formalisation of the chosen narrative, and do not span the narrative itself. That is, they target the formalisation rather than the decision-making context. They are independent from the choice of narrative insofar as they are still valid if a different narrative were chosen to simplify a wicked system, though they would become irrelevant if a different narrative were chosen. A recommendation of this type will henceforward be referred to as a *narrative-contained* recommendation.

The distinction between narrative-contained and narrative-sensitive recommendations is related to the quality check suggested by Giampietro et al. (2006) and described in section 2.1.6. For both types of recommendations, within the chosen narrative, the selected formalisation must be scientifically accepted and the model results must be effective to guide action in the specific context. For narrative-sensitive recommendations, however, also the selected narrative must be relevant in relation to the beliefs and goals existing within a given knowledge system (i.e. the sum of knowledge held by different individuals and institutions). If the selection of narrative is contested, one should be cautious about making narrative-sensitive recommendations, as they by definition might be invalidated upon a change of narrative.

Within the narrative of LCA, to a certain extent it may be possible to account for different sub-narratives. For example, one could perform a set of different LCA studies, each representing a certain sub-narrative (e.g. one study where mineral and organic fertiliser are considered equivalent, and one where they are not considered equivalent). This may increase the robustness of LCA results, while they still would be susceptible to invalidation if indicators not considered in the overall LCA narrative turned out to be of relevance.

4.2 Recommendations

The thesis contributions indicated that: (1) it is possible and meaningful to integrate adverse health effects related to pathogens in LCA of wastewater and sludge management; (2) generic LCIA models and LCIA models tailored to the context of land application of sewage sludge provide similar results regarding the exposure through agricultural produce, drinking water, and air; and (3) from an LCA perspective, the aggregate disease burden related to organic chemicals present in sewage sludge applied to agricultural land seemed to be significantly smaller than the disease burden related to metals. The latter finding might be invalidated if monitoring data were available for more contaminants, more human health EFs were available, or the predicted iFs for metals were in better alignment with field observations.

4.2.1 Design of LCA Studies

Based on above findings, it is possible to make a range of narrative-contained recommendations regarding the design of LCA studies in the context of sewage sludge management.

Quantifying Pathogen Impact Potential

LCA traditionally addresses average operating conditions where the system works according to design specifications. For adverse effects of pathogens on human health, however, the operating conditions with significant impacts may be events where processes do not work as they are supposed to. In the context of wastewater and sewage sludge management, and possibly also in other contexts, the frequency of the occurrence of operating conditions where the system does not work according to the design specifications may warrant consideration of these operating conditions also in LCA. When accounting for pathogens in LCA, the mathematical equations and model structure underlying QMRA can be useful. Yet, particular attention is warranted regarding the choice of exposure pathways and exposure conditions such as pathogen decay time, the number of people exposed, and the frequency of exposure.

It should also be kept in mind that, upon aggregation and scaling to a functional unit, details regarding individual exposure pathways and subpopulation effects are masked. That is, the inclusion of pathogen impact potential in LCA does not replace the information provided by QMRA. Rather, it provides a different perspective on the effect of pathogens on human health.

Quantifying Human Toxicity Potential

When estimating the human toxicity potential associated with metals and organic chemicals present in sewage sludge applied to agricultural land, Paper V indicated that the results obtained through the context-specific LCIA model and the generic LCIA models USEtox 1.01 and 2.0 were not significantly different from one another, at least as far as the aggregate burden of disease rather than the individual burdens of disease of the 15 metals and 106 organic chemicals considered are concerned, and as long as the exposure pathways of concern are routine exposure through agricultural produce, drinking water, and air.

However, in case treatment operations do frequently not operate according to design specifications, it may be advisable to assess also accidental ingestion of sewage sludge. Moreover, in light of the data gaps (i.e. incomplete monitoring data, lack of human health EFs), more monitoring data and human health EFs would be useful. Given above two caveats, it would be premature to conclude that context-specific LCIA models could not be useful in the context of land application of sewage sludge, and even more premature to make such a conclusion for sewage sludge management in general.

Choice of Functional Unit

Explicitly choosing the narrative prior to choosing the functional unit, rather than choosing a functional unit without explicit consideration of the associated narrative, can lead to a better alignment between the choice of functional unit and the desired functionality of the system under consideration. Recommending such practice in the goal and scope stage of LCA may increase awareness for the interrelation between the choice of narrative and functional unit.

By considering different functional units, each aligned with a certain narrative, we may realise that some of the considered sewage sludge management options actually are not functionally equivalent under some of the narratives considered, and that the comparison of different sewage sludge management options with the intent to identify the preferable one is quite intricate, as it is currently unclear how to meaningfully quantify soil health and impacts on soil life. If the choice of narrative is made implicitly through the choice of functional unit, rather than explicitly based on a profound understanding of the system at hand (in the context of this thesis the understanding of natural and agricultural production systems as well as soil health), we may attempt to compare things that cannot be compared.

4.2.2 Application of LCA Studies

When it comes to the application of LCA studies in the context of wicked systems, it is important to realise that making narrative-sensitive recommendations requires a good deal of caution. The relevant question here is under which circumstances LCA is most useful in the context of wicked systems.

Weinberg (1972) proposed that there are questions which can be stated in the language of science, but which are unanswerable by science. With regard to LCA, and any other quantitative assessment tool, one may just have to accept that it might not always be possible to get a clear answer to support decision making (Huijbregts et al., 2004).

In light of the inherent unpredictability of wicked systems, it appears reasonable to acknowledge that it is difficult for quantitative tools such as LCA to reliably predict a priori which of a set of sewage sludge management options is preferable, whichever set of criteria is applied. Whichever sewage sludge management option is chosen, unintended consequences and unforeseen effects are close to inevitable. These effects and consequences are precisely the effects and consequences that were not accounted for by the narrative applied to tame the wickedness, either because they were unknown unknowns or because they were known but deemed more irrelevant than they turned out to be. Moreover, as one can only get to know the unintended consequences and unforeseen effects of the management option that was implemented, but not those of the management options that were rejected, it is also difficult to evaluate a posteriori whether the implemented option indeed was preferable. This does not mean, however, that LCA is of no value in the context of sewage sludge management and other wicked systems.

Perhaps LCA simply is more useful as a pedagogic tool rather than a pure calculation tool (Huijbregts et al., 2004; Lundie et al., 2008). Encouraging dialogue between stakeholders in diverse parts of the system, helping them understand each others' perspectives through discussing the definition of system boundaries and the elements of the system, may facilitate a process of discovery that enlarges the understanding of what qualities, relationships, and values are important (Lundie et al., 2008). In this spirit of providing a structure for dialogue, matching the choice of functional unit and the choice of narrative in LCA can make an important contribution. LCA as a calculation tool could still be very useful, for example to identify optimisation potential and trade-off relationships for multiple variants of a given sewage sludge management option.

4.3 Ways Forward

4.3.1 Beyond LCA

Funtowicz and Strand (2011) questioned whether it is responsible to attempt to resolve wicked governance issues by state-of-the-art risk assessment and risk management procedures, when suspecting that the results of these procedures will be invalidated in a few years by the discovery of novel, unforeseen and surprising consequences of action. West Churchman (1967) insinuated that it may be morally wrong to deceive people into thinking that a wicked problem has been completely tamed, and suggests that we may rather make an attempt to better understand the untamed aspects of the problem. The bottom line of these arguments is that for wicked problems, different approaches to decision making may be required.

So, what if we embarked on an exciting journey transcending the risk management approach, starting to seriously contemplate whether there are alternative approaches to govern wicked problems such as the sewage sludge management challenge? What if we, in addition to trying to refine quantitative assessment tools such as RA and LCA, seriously contemplated under which circumstances such tools are fit for purpose (Bettencourt and Brelsford, 2015; Keirstead, 2014)? What if we, much in line with a more holistic approach to science described by Miller (1993), started to ask broader questions than which of a set of different, possibly non-equivalent sewage sludge management options is the preferable one? Let us conclude this thesis with setting a research agenda consisting of research questions that could not be addressed by this thesis, but which may open many new opportunities.

4.3.2 A Research Agenda

The research agenda proposed here is grounded in a narrative based on the idea that nutrient cycling, including sewage sludge management, in the long run should contribute to feeding the soil rather than feeding the crops. Such an approach, in the long run, may be inevitable to mitigate declining soil quality and soil degradation. When it comes to sewage sludge management in particular, five research questions are key:

(1) Which agricultural practices maintain or improve soil health?

(2) In which form are nutrients to be supplied?

(3) How does technology need to be designed in order to support soil health?

(4) How does the urban water infrastructure need to be designed in order to support soil health?

(5) Which other aspects of contemporary society interacting with the urban water infrastructure might require attention (e.g. the use of chemicals and pharmaceuticals)?

Addressing broader questions emerging from this new perspective on sewage sludge management may open many new opportunities towards infrastructural systems that are rooted in a profound understanding of natural as well as agricultural production systems and the importance of soil health, rather than a preference for a certain technological system of nutrient recovery from sewage sludge. The solution space considered in light of the sewage sludge management challenge would become much broader, and ideally stimulate creative solutions that combine aspects of public hygiene and soil health.

Certainly, the proposed research agenda would not prevent us from experiencing unintended consequences, for also with above research agenda it is impossible to know a priori all relevant qualities and interrelations. However, a design approach based on understanding principles and patterns in real world systems may be more adequate than an approach based on system optimisation through comparison of non-equivalent alternatives. Tools like RA and LCA can be helpful in this decision to a certain extent, but not to the extent one may possibly wish they were. Let us start asking the right questions and looking for new tools that can complement the range of tools that are already available.

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