Understanding flows for integrated catchment management: water quality, quantity and ecosystem services
Dirk Vrebos

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Proefschrift voorgelegd tot het behalen van
de graad van doctor in de wetenschappen aan
de Universiteit Antwerpen te verdedigen door

Dirk Vrebos

Promotor: prof. dr. Patrick Meire
Copromotor: dr. Jan Staes

Faculteit Wetenschappen - Departement Biologie
Onderzoeksgroep Ecosysteembeheer

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Understanding flows for integrated catchment management: water quality, quantity and ecosystem services.

Inzichten in stromen voor integraal waterbeheer: waterkwaliteit, -kwantiteit en ecosysteemdiensten.

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Dankwoord

Iets wat vier jaar zou duren… werden er tien. Maar na al die tijd, waar er met tussenpozen hard aan werd gewerkt, is het dan toch gelukt. Eindelijk gedaan met nietszeggende antwoorden te geven op eeuwig weerkerende vragen.

Hoe gaat het met je doctoraat?
        Goed. Er wordt aan gewerkt.
Schiet het wat op?
        Toch wel.
Heb je nog veel werk?
        Wel wat.
Wanneer ga je verdedigen?
        Waarschijnlijk volgend jaar…

Sommige vrienden, en misschien ook familie, hadden er ondertussen toch wel serieuze twijfels bij of het ooit zou lukken. Maar dan is het altijd leuk om hun ongelijk aan te tonen. Blijkbaar hielpen ook twee paar kinderogen, die mee op mijn laptop kijken, om wat vaart achter de laatste pagina's te zetten.

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Summary

Human developments, such as increasing urbanization, have a wide range of effects on catchments and river systems. This results in profound changes in the hydrological regime and a deterioration of the water quality, which threaten the sustainable use of these systems and bring considerable costs. To manage these changing catchment characteristics sustainably, both natural and anthropogenic processes need to be understood and evaluated in an integrated approach. Over the past decades, related management concepts, such as integrated catchment management, have been translated into policy and legislation. But their actual implementation remains a challenge. Integrating the concept of ecosystem services might improve the effectiveness of these management frameworks.

The overall objective of this work is the development of methodologies that allow for the implementation of the ecosystem services concept in integrated catchment and natural resources management. But to develop these, first a good understanding of the catchments processes is required. Therefore, in the first part of this work, the impact of human development and land use patterns on flow pathways, water quantity and quality is investigated in the Nete catchment, Belgium. Different modelling and statistical analysis are used to assess spatial and temporal relationships over different scales. Special attention goes to the impact of wastewater treatment infrastructure on catchment functioning. The results signify the fundamental changes that have taken place in river dynamics and reveal a number of specific challenges a complex catchment system poses. In the second part this system knowledge is used to develop methodologies which aim to integrate the ecosystem service concept in ICM and INRM in both data rich and data scarce catchments. Indicators are developed to evaluate the supply and demand of several ecosystem services in an upstream-downstream analysis. The developed methodologies illustrate the opportunities of such an integration, but also the remaining challenges and serious limitations of such an approach are discussed.

This thesis provides evidence of the human impact on the hydrological regime of rivers, with an emphasis on the importance of the sewer system. The temporal and spatial scales at which these changes have taken place, makes it difficult to investigate the relevant processes and flow paths. This hampers the actual implementation of such integrated management concepts. Overall this thesis illustrates how human development has affected natural systems to a point where management from an integrated, system perspective has become almost impossible.

Samenvatting

Ontwikkelingen, zoals toenemende verstedelijking, hebben een breed scala aan effecten op rivierbekkens en -systemen. Dit resulteert in ingrijpende veranderingen in het hydrologische regime en een algemene verslechtering van de waterkwaliteit, die een bedreiging vormen voor het duurzame gebruik en aanzienlijke kosten met zich meebrengen. Om deze veranderende stroomgebieden op een duurzame manier te beheren, moeten zowel natuurlijke als antropogene processen worden begrepen en geëvalueerd in een geïntegreerde benadering. In de afgelopen decennia zijn gerelateerde managementconcepten, zoals integraal bekkenbeheer, vertaald in beleid en wetgeving. Maar hun effectieve implementatie blijft een uitdaging. Integratie van het concept van de ecosysteemdiensten kan de effectiviteit van deze managementkaders verbeteren.

De algemene doelstelling van dit werk is de ontwikkeling van methodologieën die de implementatie mogelijk maken van het ecosysteemdienstenconcept in geïntegreerd bekkenbeheer en beheer van natuurlijke hulpbronnen. Maar om deze te ontwikkelen, is een goed begrip van het rivierbekken en haar processen vereist. Daarom wordt in het eerste deel van dit werk de impact van menselijke ontwikkeling en landgebruikspatronen op stroompaden, waterkwantiteit en kwaliteit onderzocht in het stroomgebied van Nete, België. Verschillende modellering en statistische analyse worden gebruikt om ruimtelijke en temporele relaties op verschillende schalen te beoordelen. Speciale aandacht gaat naar de impact van rioleringsnetwerken op het functioneren van het bekken. De resultaten van dit proefschrift geven de fundamentele veranderingen weer die zich hebben voorgedaan in de rivierdynamiek en onthullen een aantal specifieke uitdagingen die een complex stroomgebied met zich meebrengt. In het tweede deel wordt deze systeemkennis gebruikt om methodologieën te ontwikkelen die het ecosysteemserviceconcept in integraal beheer willen integreren in zowel datarijke als data-schaarse gebieden. Indicatoren worden ontwikkeld om vraag en aanbod van verschillende ecosysteemdiensten in een stroomopwaartse-afwaartse analyse te evalueren. De ontwikkelde methodieken illustreren de kansen van een dergelijke integratie. Maar ook de resterende uitdagingen en serieuze beperkingen van een dergelijke aanpak worden besproken.

Dit proefschrift toont duidelijk de menselijke impact op het hydrologische regime van rivieren aan, met de nadruk op het belang van het rioolstelsel. De temporele en ruimtelijke schalen waarop deze veranderingen hebben plaatsgevonden, maken het moeilijk om de verschillende processen en stromen te onderzoeken. Dit belemmert de daadwerkelijke implementatie van dergelijke geïntegreerde managementconcepten. Dit proefschrift illustreert hoe menselijke ontwikkeling riviersystemen heeft beïnvloed tot een punt waarop beheer vanuit een geïntegreerd systeemcontext bijna onmogelijk is geworden.

Table of Content

Abbreviations

AEM	Asymmetric eigenvector maps
DSS	Decision support system
EIA	Effective impervious area
ES	Ecosystem services
EU	European Union
FDPM	Fully spatially distributed hydrological process model
FEA	Flemish Environment Agency
ICM	Integrated catchment management
INRM	Integrated natural resources management
IWRM	Integrated water resources management
ME	Mean error
MEM	Moran's eigenvector map
MS	Member states
NBS	Nature based solutions
NGI	National Geographic Institute
NSE	Nash–Sutcliffe efficiency
PCAIV	Principal component analysis on instrumental variables
RCC	River continuum concept
RDA	Redundancy analysis
RMSE	Root-mean-squared error
SDA	Service demanding areas
SOC	Soil organic carbon
SOD	Sewer overflow devices
SPA	Service providing areas
TIA	Total impervious area
WFD	Water Framework Directive
WQ	Water quality
WWCR	Wastewater collection region
WWTP	Wastewater treatment plant

Chapter 1 - General introduction

1. UNDERSTANDING CATCHMENT DYNAMICS

1.1 Catchments as network systems

River systems are highly complex, dynamic and hierarchical systems as each confluence of streams is the sum of its upstream properties and processes. This hierarchical perspective describes each river network as a unique, patchy continuum from headwaters to mouth (Poole, 2002). Therefore, rivers are increasingly investigated from a landscape or catchment functioning perspective (Allan, 2004). A catchment can be defined as an area of land where precipitation collects and drains off into a river. A catchment is also a hydrological response unit, a biophysical unit, and a holistic ecosystem building block to understand system behavior in terms of the materials, energy, and information that flows through it. Therefore, as well as being a useful unit for physical analyses, it can also be a suitable socio-economic-political unit for management planning and implementation (Wang et al., 2016).

Hydrological connections, such as groundwater flows, streams, but also sewer pipes and road ditches, link the different areas within a catchment across vertical, lateral and longitudinal dimensions and make up the hydrological catchment system. These different connections can span spatial and temporal scales (Covino, 2017; Ward, 1989). Unidirectional, longitudinal hydrological connections, from upstream to downstream through streams are the easiest understood. But also vertical (e.g. groundwater seepage) and lateral movement (e.g. run-off) make part of the hydrological system within a catchment (Figure 1-1).

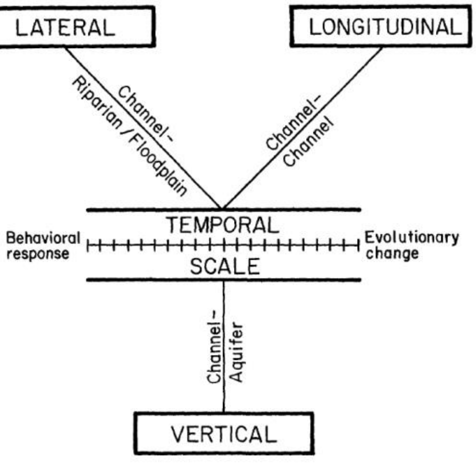

Figure 1-1: Conceptualization of the four-dimensional nature of lotic ecosystems (Ward, 1989).

Natural systems, tend to buffer and stabilize flows and make the system more resilient against disruptive events (Gunderson, 2000). Anthropogenic developments, such as agricultural expansion and urbanization, do not only change the land use type on a local level, but also affect hydrological flow paths in many aspects (Carey et al., 2013; Pickett et al., 2011). These artificial flow paths, such as sewer systems, generally act as rapid transport systems towards the river, reducing residence time of water and solutes and changing the flow regime and the occurrence of different biogeochemical processes within the catchment (Figure 1-2) (Kaushal et al., 2012). For example, urbanization reduces infiltration and recharge due to the increase in impervious areas, increasing run-off, peak flows and reducing river baseflow (Kauffman et al., 2009; Price, 2011; Simmons et al., 1982). But this decrease in infiltration and seepage, also reduces nitrification within the infiltration areas and denitrification in groundwater upwelling areas (Ward, 1996). The decline of these processes can increase eutrophication and oxygen depletion in streams, resulting in an overall degradation of the rivers ecology (Smith, 2003). Understanding how anthropogenic disturbances interact and affect these different flow paths and natural processes within a catchment can help us to sustainable manage these systems. But this understanding requires continued research across different scales and time periods.

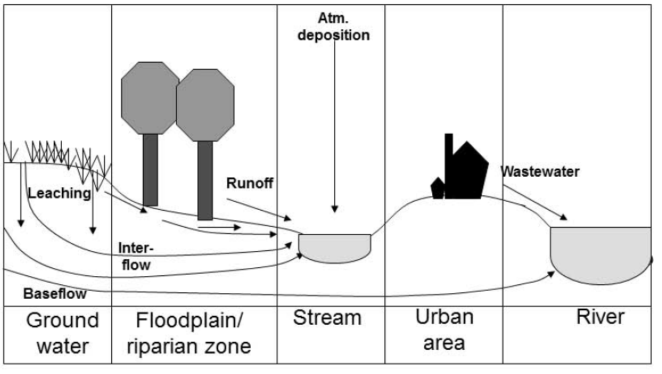

Figure 1-2: Schematic of the hydrological system in river basins, including the soil, groundwater, riparian zone and floodplain, and the stream or river, and with wastewater flows from urban areas (Bouwman et al., 2013).

Where undisturbed steams provide dynamic, hierarchically nested habitats with many feedbacks, human influenced catchments incorporate additional anthropogenic processes, which lead to perturbed flows and more extreme, pulsed responses of river systems to rain events (Kaushal et al., 2015). As such these catchments have become more complex, making it more difficult to predict whether a stream is still dominated by its natural processes or whether, and to what extent, anthropogenic effects will take over. Today it is increasingly recognized that human actions at the landscape scale are a principal threat to the ecological integrity of river ecosystems, impacting flow regime, habitat, water quality, and the biota via numerous and complex pathways (Allan, 2004; Allan et al., 1997). Catchment disturbance, from the conversion of natural ecosystem to human dominated land uses such as urban areas or changes in flow, affects the physical and chemical conditions of streams with inevitable consequences for stream life (McGrane, 2016). However, freshwater aquatic ecosystems are connected by flowing water, which transports these effects throughout the catchment. Therefore, protection and restoration of these ecosystems needs to be done with regard for upstream, downstream or upland areas (Nel et al., 2011).

1.2 Catchments: building knowledge

Impacts of land use changes on river systems have long been studied (de la Crétaz et al., 2007). More than 2000 years ago, Plato already described the effects of deforestation on springs drying out and increases in soil erosion and flooding downstream (Plato, 360). Over the last two centuries scientist have tried to understand both the natural processes that occur within a catchment and river system, as well as the different effects of human activities on these processes and the scales at which they occur. Detecting human impact on riverine systems is challenging because of the daunting diversity in biological, chemical, hydrological and geophysical components that can influence the results (Gergel et al., 2002). Interactions, additive and/or synergetic, between different disturbances within a catchment or subcatchment make it difficult to locate and rank the different sources of water quality degradation (MacDonald, 2000).

To investigate these cause – effect relationships, different types of analyses are available today. Initial scientific research in the 19th century focused on observing trends in catchments. Later on, paired-watershed designs were used to test and identify the roles of forest cover and other land uses on different stream processes (Andreassian, 2004). From these initial experiments a whole range of research fields has grown from the empirical statistical analysis of nested upstream land use indicators to assess stream ecosystem health and water quality, biodiversity and river functioning (Figure 1-3) (Uuemaa et al., 2013) to the use of complex computer models to simulate a wide range of natural and anthropogenic processes (Boyle et al., 2001; Salvadore et al., 2015). Advancement in scientific understanding and increasing computational power has resulted in a wide range of models that describe in detail the catchments hydrology, the transfer of nutrients from soil to aquatic ecosystems or the biogeochemical processes within aquatic systems (Bouwman et al., 2013; Salvadore et al., 2015). Today these different fields still evolve with ongoing developments in for example monitoring systems, statistics, geographical information systems and model design. As a result, we continue to improve our understanding of these complex systems.

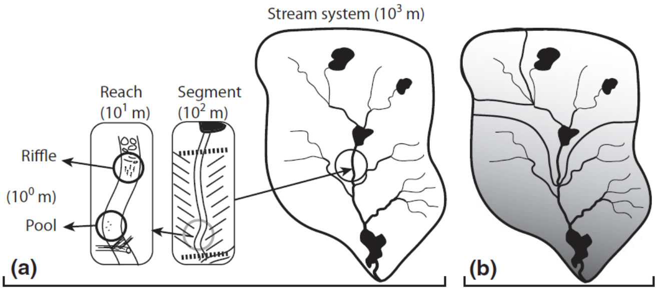

Figure 1-3: (a) Frissell et al. (1986) hierarchy showing all stream segments, reaches, riffles / pools and microhabitats nested within the entire sub-catchment. (b) Example of directionally nested and overlapping catchments typically used in hydrological analysis (figures adapted from Frissell et al., 1986; Seelbach et al., 2006) taken from Melles et al. (2012)).

1.3 From knowledge to concept to management

With expanding knowledge on streams and catchments, scientist have tried to translate the observed dynamics into theoretical concepts (Melles et al., 2012). Most of these concepts have in common that they emphasize the role of physical processes in ordering biological systems and the role of spatial and temporal scales in understanding these processes (Bouwman et al., 2013). They find their origins in the fields of geography and stream geomorphology (e.g. Hack, 1960; Strahler, 1957) as well as in ecology (e.g. Vannote et al., 1980). Vannote et al. (1980) observed that physical variables within a river system changed along a continuous gradient from headwaters to mouth, determining producer and consumer communities (River Continuum Concept). In the following decades, other authors developed additional concepts which increased our understanding of the complex relationships between different river related processes and the changes in physical and ecological characteristics that can be observed along the river downstream.

As our understanding of the complex relationships between natural processes in catchments has grown tremendously, the integration of anthropogenic influences in the related concepts is still progressing. Statzner et al. (1986) already suggested that expected, natural zonation patterns over long stream reaches are often obscured by anthropogenic influences on the stream or its valley. As a result, concepts for natural systems are less applicable to human influenced river systems and catchments. Concepts such as the “Urban Stream Syndrome” (Walsh et al., 2005) and “Urban Watershed Continuum” (Kaushal et al., 2012) attempt to integrate and adapt the theoretical concepts for natural systems to conceptual frameworks that can be applied to urbanized areas. As such the Urban Watershed Continuum recognizes that urban development influences natural downstream fluxes and transformations of carbon, contaminants, energy, and nutrients across the 4 space and time dimensions which were also used by Ward (1989). For example, the replacement of first order streams by urban infrastructure or vertical interactions between leaky pipes and groundwater can change river hydrology significant.

These theoretical concepts are used to steer the development of new hypothesis, experiments and integrated models and indicators to further expand our knowledge of these systems (Bouwman et al., 2013; Lorenz et al., 1997). Through merging perspectives, concepts, and modeling techniques, integrated model approaches are developed that encompass both aquatic and terrestrial components in heterogeneous landscapes (Bouwman et al., 2013). But, they have also changed the manner in which catchments are managed today and how riverine ecosystems are restored. For example, the serial discontinuity concept by Wards et al. (1983) influenced our understanding of how disruptive processes, such as dams build on rivers, impact river processes and how such rivers should be managed (Ellis et al., 2013). Parts of these concepts are also integrated in management frameworks such as Integrated Catchment Management (ICM) and Integrated Natural Resources Management (INRM). Nevertheless, management and restoration projects are often developed without making use of the large body of ecological theory (Lake et al., 2007). Therefore, a further integration and good implementation of these concepts within management is needed.

2. CATCHMENT MANAGEMENT

2.1 Managing complexity

Catchments are examples of complicated natural systems that are becoming even more difficult to comprehend because of the socio-economic systems and their associated processes built within them. As catchments affect and are affected by society, water resource issues get more common and severe. Reducing the impact of these problems, is commonly achieved by bringing ecosystems under control, while neglecting underlying ecological processes and feedbacks (Holling et al., 1996). Catchment managers seek to stabilize resource outputs and diminish environmental variability and natural disturbances by bringing these natural processes under command and control (Carpenter et al., 2001). But many of these processes can never be forced into a strict regime as they are part of an intricate web of interactions which operates on various scales in time and space (Jakeman et al., 2003). This pathology of ‘command and control’ affects the resilience of the ecosystems, which leads to a degradation or collapse of the ecosystem and the services they provide (Briggs, 2003; Fisher et al., 2009; Holling et al., 1996).

System complexity, variation and uncertainty are today recognized as intrinsic features of linked social and natural processes. Natural resource management strategies should incorporate these features in their methodologies to move towards sustainability (Medema et al., 2008; Rammel et al., 2007). In order to restore resilience of aquatic ecosystems in catchments and ensure the sustainable provisioning of aquatic services, social and economic activities have to be consistent with each other and with the hydrological and ecological characteristics of the catchment. It is necessary to recognize catchments as interlinked systems in which the different land uses and anthropogenic activities within the catchment have direct and indirect effect on the water quantity and quality and the ecological status of the river system (Allan, 2004; Allan et al., 1997). As a result, upstream and downstream activities have to be compatible to each other (Falkenmark, 2004).

To sustainably manage these systems, land use and management has to be integrated and coordinated at a catchment scale instead of the existing administrative boundaries (Jewitt, 2002; Staes et al., 2008a). At the same time both social and economic activities within a catchment have to be consistent with hydrological and ecological needs of human well-being (Lundqvist et al., 2000). Concepts such as integrated water resource management, integrated natural resources management and integrated catchment management focuses on integration and coordination of the different activities on the relevant scale, catchment or landscape, and the integration of ecosystem and society based on public participation (Downs et al., 1991; Gardiner, 1994; Staes et al., 2008b). They provide a management with more synergy and less conflict (Jamieson, 1986; Mostert, 1999). These concepts have received much attention in recent years, but have been criticized as well for their amorphous definition and limited translation in to practice (Biswas, 2004; Gallego-Ayala, 2013; Medema et al., 2008; Reeves et al., 2009).

2.2 Integrated management

As our knowledge of ecosystems and society increases, so does the information that needs be integrated in decision making. Policy and decision makers rely therefore more and more on formal decision support at every level, from guidance through best practice to often software driven models and decision support systems (DSS) (Clark 2002). Many different DSS's have been developed in recent years to help in the implementation of ICM and the Water Framework Directive (WFD) (e.g. de Kok et al., 2009; Giupponi, 2007; Holzkämper et al., 2012; Maurel et al., 2007).

The WFD is the European legislation which implements ICM across the European Union (EU). It was adopted by the European Commission (EC) in 2000 (2000/60/EC), and implemented by the Member States (MS) in the following years. The main aims of the WFD are to protect and restore the water environment within the EU and to manage water resources sustainable, taking into account environmental, economic and social considerations. The WFD is implemented through the development of "river basin management plans" grouping all management actions designed to achieve a good ecological and chemical status of all natural surface water bodies within the catchment (Staes et al., 2003). Although the WFD was designed to integrate ICM within European policy and it has transformed water management within the EU, criticism has been given regarding its inability to fully implement a catchment-based system approach. In order to better align the WFD with its initial goals and better integrate and communicate the importance of environmental quality, the concept of "ecosystem services" (ES) has a high potential (Everard, 2012; Vlachopoulou et al., 2014).

3. ECOSYSTEM SERVICES

3.1 Concept and classification

Ecosystem services are the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life (Daily, 1997). The concept of ecosystem services aims to clarify the societal dependence of different ecological processes and stop the further degradation of ecosystems and the ES they deliver. Although earlier developed (e.g. Ehrlich et al., 1981; Westman, 1977), the concept gained momentum in the 90's with the publication of Costanza et al. (1997) and got further attention with publications such as the Millennium Ecosystem Assessment (MEA, 2005) and the Economics of Ecosystems of Biodiversity (TEEB, 2010). Today the concept is widely accepted, however many challenges for successful implementation remain.

Several types of ES are recognized, although the classification and typology between ES lists can differ (e.g. Haines-Young et al., 2013; MEA, 2005; TEEB, 2010). The three main categories; provisioning services (e.g. fisheries), regulating services (e.g. flood control) and cultural services (e.g. recreation), can be found in all present ES classifications. However, there is still much debate on how to take account of habitat or supporting services, which are mainly structures and processes that support other services, but can also be important on their own. Aquatic and river related ES can be found in all four categories (Jin et al., 2015). Not only do aquatic ecosystems provide direct ecosystem services within the first three categories, the underlying hydrological processes are directly or indirectly involved in the generation of almost every ES. The hydrological ecosystem efficiently acts on flow regulation and filtration, crucial aspects of which involve the control of mean surface runoff, peak or flood flows, base or dry season flow, and erosion and sediment load, as well as recharge of groundwater and soil moisture dynamics (Jin et al., 2015). The benefits provided by these services are seldom expressed as prices for market commodities, but can be valued as avoided costs (e.g. water purification) and/or avoided risks (e.g. floods). Improved ecological functioning through ecological restoration or by reduction of pressures to increase performance, thus generates ES such as water quality improvement, safety, nature experience, etc. and can be considered as benefits to society. But as aquatic ecosystem functioning affects almost every ES, almost every land use change in a catchment has an impact on the aquatic ecosystems as well (Brauman et al., 2007). This interconnection makes aquatic ES sensitive to anthropogenic impacts within the whole catchment (Jin et al., 2015).

3.2 Flows of services

Rationalization of the ecosystem services framework requires different aspects and dimensions to be taken into account (Seppelt et al., 2011; Seppelt et al., 2012). To quantify the delivery of ecosystem services both supply, and demand need to be adequately assessed and quantified (Boerema et al., 2017). Supply refers to the capacity of a natural area the provide one or a bundle of ecosystem services. While demand refers to the consummation or use of these services at the same or a different location (Burkhard et al., 2012). For example, recreation will take place within the natural area that provides the service. But the use of wood products will take place outside of the woods that provide them. One of many aspects relevant to ES assessments, is how ES supply and demand can be separated in time and space (Luck et al., 2009). Different types of flow mechanisms can deliver the service from supply (or provisioning area) to the demand areas (or benefiting areas) (Figure 1-4). These flows can be mediated by both natural (e.g. water flow) and human induced processes (e.g. movement of people) (Fisher et al., 2009). As ES are often used at different locations from where they are produced, possible positive of negative spill over effects can also take place on locations located along those flow paths or elsewhere (J. Liu et al., 2013).

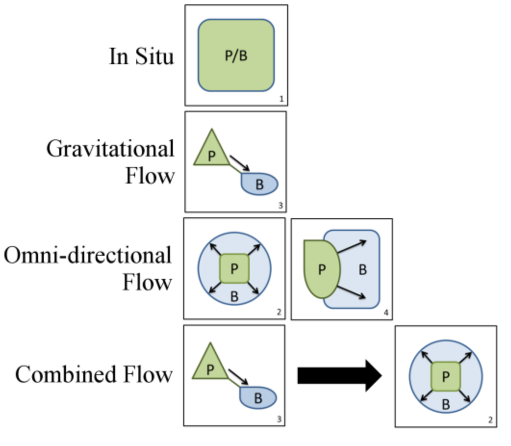

Figure 1-4: Overview of the different flow mechanisms between provisioning (P) and benefiting areas (B) evaluated within this study (Symbols after (Fisher et al., 2009)).

3.3 Integration in management

Despite the increasing attention for ecosystem services in nature and habitat conservation, the integration of ecosystem services into conservation strategies for freshwater and marine systems is limited. Surprisingly few published examples exist where a comprehensive assessment of ecosystem services supported development of conservation plans (Boulton et al., 2016). In the ecosystem services research most attention has gone to land protection options (Egoh et al., 2007), but the approach is equally appropriate for targeting incentives for best management practices or other management choices that can co-occur with developed uses of the land or water (Wainger et al., 2010). In heavily altered systems which have currently a low ecological value, protecting existing habitats and their functions and services, will not suffice. In order to fully fulfil the potential of the ES concept in intensely managed catchments, we should not only be able to assess current ES delivery and protect it. In addition, we should be able to locate those areas that have a high potential for ES supply. Based on societal needs the delivery of ES can be restored through habitat restoration, at the most adequate locations (Staes et al., 2010).

Although discussion remains regarding the position of the ecosystem service concept towards management frameworks such as INRM (Cook et al., 2012), it has the potential to improve INRM, Integrated Water Resources Management (IWRM) and ICM (Doherty et al., 2014; Grizzetti et al., 2016; S. Liu et al., 2013) and is often considered to be helpful in natural resources management. The concept can bridge the gap between research and management (Sitas et al., 2014) and provide a common language for managers, researchers and different stakeholders (Granek et al., 2010). But its place within management as a whole (Norgaard, 2010) and specific management and impact assessment frameworks is still disputed (Baker et al., 2013; Cook et al., 2012).

"Natura based solutions" (NBS) is a recent management concept which can help to integrate ecosystem services in management practices. Although the concept has different definitions, NBS is understood, at a European level, as "living solutions inspired by, continuously supported by and using nature, which are designed to address various societal challenges in a resource-efficient and adaptable manner and to provide simultaneously economic, social, and environmental benefits" (European Commission, 2015; Nesshöver et al., 2017). It is a concept introduced specifically to promote nature as a means for providing solutions to climate mitigation and adaptation challenges. (Keesstra et al., 2018). It integrates explicitly ecosystem complexity in its framework and acknowledges that the underlying natural processes need to be used in a sustainable way.

Where and how ES or NBS can be integrated in natural resource management remains a topic of discussion. Besides theoretical constraints, the implementation of ES within models and DSS for ICM remains limited. For example, hydro-economic models for water allocation between competing water do not apply any systematic approach to identify potential environmental impacts or effects on ES (Momblanch et al., 2016). To fully develop tools that help with the implementation of ES in ICM and INRM numerous challenges remain.

4. RESEARCH OBJECTIVES

The overall objective of this thesis is the development of methodologies that allow the implementation of ecosystem services concept in ICM and INRM, taking into account the spatial flow paths between supply and demand areas. To develop and apply these concepts in a credible/rigorous manner, we tested them on a catchment where we have datasets with adequate spatial, thematic and temporal resolution. But also developed a methodology for data-scarce regions. However, to implement these concepts substantiated within a specific catchment or landscape, first a good understanding of the system, different flow paths and processes within that catchment is needed.

Therefore, this general objective is divided into two more detailed objectives.

Objective 1: Improve our understanding regarding the hydrological functioning of a catchment.

The overall composition of a catchment: soil characteristics, climate, ecology, etc. makes that every catchment and stream has their unique biophysical properties and hydrological and hydro chemical behavior. On the other hand, anthropogenic developments have changed the hydrological functioning of many catchments, impacting both quantity and quality of water flows. To develop sound ICM methodologies that represent the system correctly, a good understanding of these changes and the current functioning of the catchment in question is needed.

Objective 2: Integrate ecosystem services into catchment management and planning, taking into account upstream-downstream interactions.

To integrate ecosystem services into catchment management and planning, methodologies need to be developed which can facilitate this process. To make this possible, various challenges need to be tackled, amongst others: 1) ES demand and supply needs to be mapped with high spatial accuracy; 2) demand and supply patterns and dependencies need to be assessed on a catchment level through a stream (sub-catchment) network analysis; 3) methodologies need to be suited for the amount and type of available information within the catchment; 4) drawing relevant information for management and policy from highly complex spatial-temporal analysis.

5. THESIS OUTLINE

This thesis is divided in two parts, addressing the two main research objectives. To implement ICM within a catchment and use ecosystem services supply and demand as a management concept, a good understanding of the catchment functioning is required. Therefore, the aim of **part 1** is to get a better understanding of the impact of land use patterns on the flow pathways, water quantity and quality within the Nete catchment as outlined in **Objective 1.** Both modelling and statistical analysis are used to assess spatial and temporal relationships on different scales. This revealed a number of specific challenges a complex catchment system poses. Special attention goes to the impact of wastewater treatment infrastructure on catchment functioning.

The relationship between land use, with a focus on wastewater treatment infrastructure, and the hydrological regime is assessed in **Chapter 2**. The interaction of artificial wastewater treatment catchments with natural hydrological drainage catchments is investigated. The impact of these water transfers on hydrological regimes is evaluated with a spatially distributed, physically based hydrological model.

Land uses, including wastewater treatment infrastructure, also affect the water quality of a river system. In **Chapter 3** the impact of upstream land use on various water quality parameters is analyzed. The importance of a correct representation of flow paths, including wastewater flows is examined for the entire Nete catchment.

In **Chapter 4** a more extensive water quality dataset is used to assess spatial patterns for a part of the Nete catchment. Aim of this study is to understand how land use affects a broad range of water quality parameters and at which scales these water quality patterns vary within the river system. Specific statistical tests are used to investigate whether upstream – downstream patterns of water quality are present within the catchment.

Part 2 of the thesis aims to develop methodologies that can be used to integrate the ES-concept in ICM and INRM. Goal is to integrate ecosystem services within ICM and INRM in both data rich and data scarce regions in line with **Objective 2.**

Chapter 5 presents a methodology to select the most effective sites for improving the ecosystem service “water quality regulation” within a catchment setting, by taking both demand for WQ improvement and supply through ecosystem restoration options into account. It explores to what extend upstream-downstream analyses can be utilized for ICM and how reliable ES demand indicators can be developed for branched network systems such as river systems.

Although the same principles for natural resources management should be applied, regardless of the location, available knowledge and data influences the type of analysis that can be applied within a region. **Chapter 6** presents an INRM study in a data-scarce region located next to the Rwenzori Mountains (Uganda). In this study a methodology was developed to evaluate demand and supply flows for ecosystem services in a spatial explicit manner within a region that lacks detailed information and large datasets.

The main findings presented in the previous chapters are further discussed in a broader context and the implications for integrated management are explored in **Chapter 7**. An overview or the different applied methods and research areas is given in Table 1-1.

6. RESEARCH AREAS

6.1 The Nete catchment

The research of this thesis was conducted in two different catchments: the Nete catchment (Belgium) and the Lake George catchment (Uganda).

The Nete catchment (approximately 1.673 km²) is situated in the central Campine region in Northern Belgium (Figure 3-1). It has a marine, temperate climate with an average precipitation of 800 mm/year. The dominant soil type is sand, with loamy sand occurring in the floodplains. Topographic height ranges between 3 m and 82 m above sea level, making it a typical lowland river system.

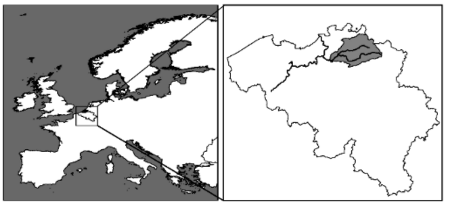

Figure 1-5: Location of the Nete catchment.

The Nete yearly discharges on average 389 million m³ water into the Rupel. The Nete has two main tributaries, dividing most of the catchment into two different systems. The northern part of the catchment is drained by the Kleine Nete (+/- 44km long) and the southern part by the Grote Nete (+/- 80km). Both rivers confluence 15km upstream of the Nete discharge point into the Rupel river. Land use in the Nete catchment consists of a patchwork of croplands, pastures, broadleaved woodland and evergreen needle leaf forests, roads and villages and cities of different sizes. Because of this land use and historical changes in the river system such as straightening and embankments, the hydrological regime and water chemistry are, spatially and temporally, highly variable.

Table 1-1: Overview of the chapters and their content.

Chapter	Relationships	Time period	Data	Methods/Statistics	Catchment area
2	Land use - hydrology	2004 - 2008	Land use map, soil map, subcatchment and river delineations,… Rainfall and river flows	Empirical data analysis MIKE-SHE/MIKE 11 modelling	Grote Nete
3	Land use - water quality	2003 - 2010	Land use map, soil map, subcatchment and river delineations,… Water quality data (NO_2^-, NO_3^-, NH_4^+ and Cl^-)	Partial correlation statistics	Nete
4	Land use - water quality - spatial variation	2010 - 2012	Land use map, soil map, subcatchment and river delineations,… Water quality data (25 parameters)	Spatial predictor calculation PCA-IV statistics	Kleine Nete
5	Ecosystem services - water quality	2007 - 2016	Land use map, soil map, subcatchment and river delineations,… Water quality data (NO_2^-, NO_3^-, NH_4^+ and Cl^-)	Ecosystem service models Upstream search algorithms	Nete
6	Ecosystem services demand and supply	2010 + future scenarios	Land cover map, river and subcatchment delineation	Ecosystem services scoring Different ES flow algorithms	Lake George

In total 55 cities and municipalities are fully or partially located within the catchment, with an average population density of 420 inh/km². At the moment 29 WWTPs are situated within the catchment. Wastewater treatment zones do not coincide with the natural catchments and wastewater is actively transported from one (sub) catchment to another for treatment. Although the Nete catchment is considered to be one of the most natural catchments within Flanders (northern part of Belgium), almost none of the rivers and streams within the catchment meet all of the European Water Framework Directive standards and more investments to reach these standards are needed. For example, in recent years N and P related WQ has improved, increasing the number of sample points which meet several of these WQ standards (Figure 1-6). However, many do not comply with all of them. Although P still is problem within the catchment. he analysis in this thesis focusses on total N and its different components. The relationships between land use and P concentrations are more difficult to establish because of high iron concentrations in parts of the Nete catchment and its impact on P chemistry

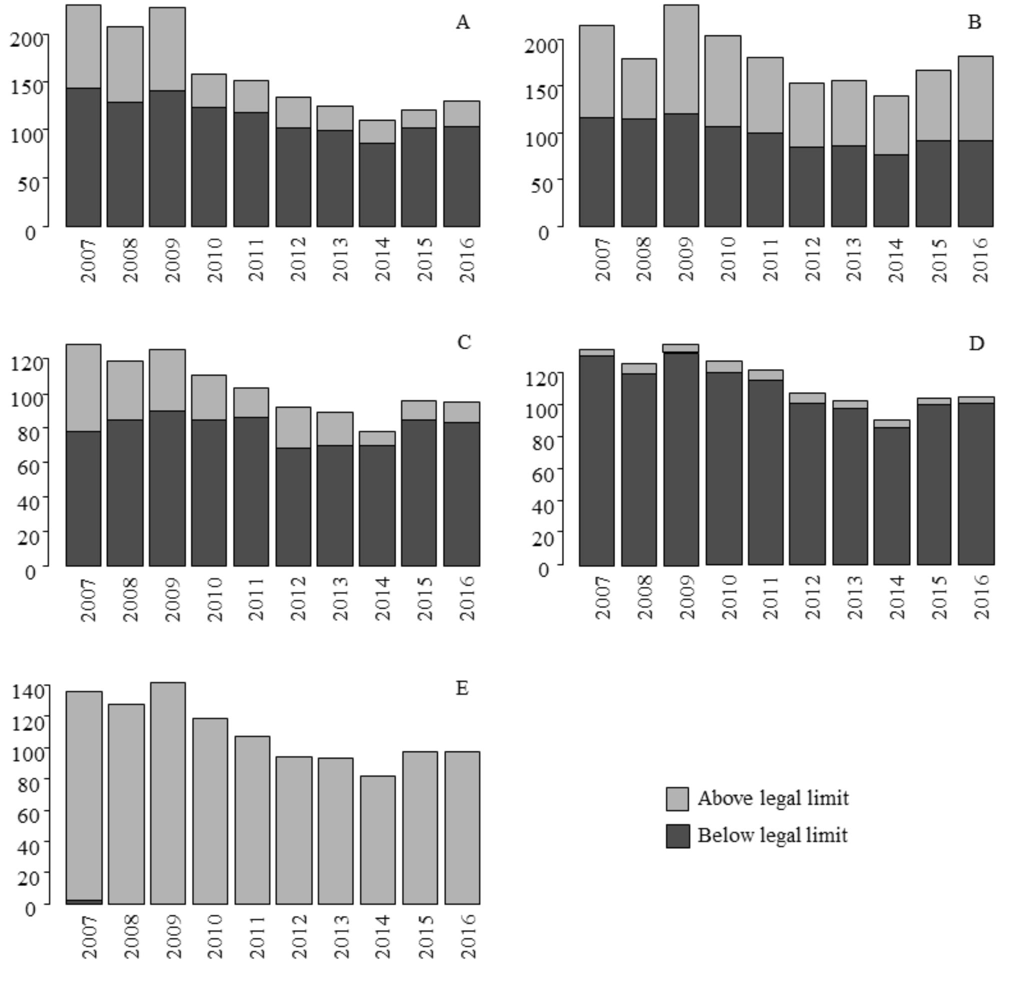

Figure 1-6: Overview WQ standard evaluation in the Nete catchment for different N and P parameters: A NO_3^-) , B) KjN, C) total N, D) oPO_4 and E) total P.

Because of its variation in land uses, the catchment is well suited to study and evaluate the differences between natural and anthropogenic processes and how they affect each other. The alternation in small and larger villages with natural areas and agricultural land in between makes it an interesting area to evaluate the effects of increasing urbanization on natural systems processes and to understand how these systems should be managed. The natural and anthropogenic characteristics of the catchment are not only found in the Nete catchment, making the results of these chapters also of interest for other regions. Results of chapter 2 and 3 are of relevance to areas with a complex land use pattern, where sewer infrastructure might have a significant impact on the river system. The results of Chapter 4 and 5 are more important for sandy regions, where mostly groundwater fed rivers are present. Although the methodology can also be applied to other types of river and catchment systems.

Although part 1 of this thesis aims to understand the system at a catchment level, not all research questions could be investigated at this scale (Figure 1-7). This because of the scale at which data and models were available. Chapter 2, which explores the hydrological regime, encompasses only a part of the Grote Nete, as no fully spatially distributed hydrological process models are available on a larger scale. Chapter 4 explores only the Kleine Nete catchments. Practical constraints limited the number of WQ sample points which could be collected and analyzed for a wide range of WQ parameters. A high density of sample points was considered to be more important to the analysis, then a catchment wide dataset with a lower SP density for this kind of analysis. Chapters 2 and 5 made use of data that were collected by Flemish government agencies throughout the region. As a result, these analyses were not constrained to parts of the catchment and complete catchment analyses could be performed.

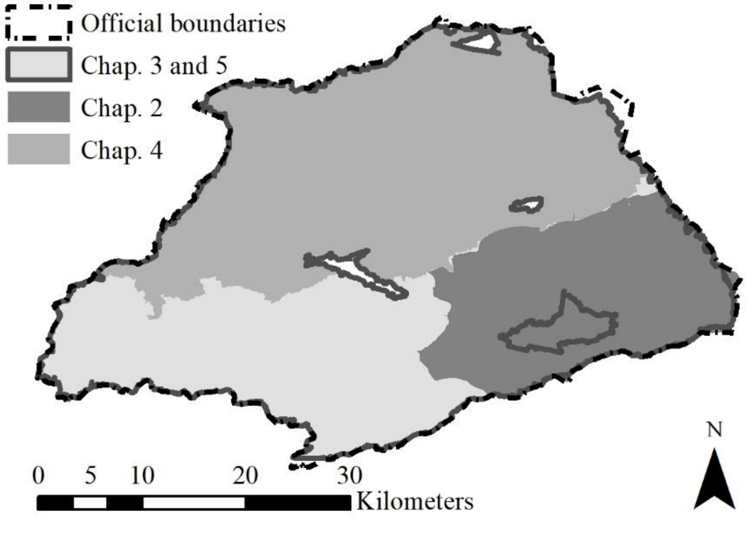

Figure 1-7: Overview of the research areas for each of the chapters (2-5).

6.2 Lake George basin

The methodology for the INRM study in Chapter 6 was developed in the Lake George basin, located in Western Uganda. The area consists of a series of rivers that are situated around and drain into Lake George. The basin is part of the Great Rift Valley and is characterized by diverse geophysical and ecological systems: high mountains, tropical high forest, savannah and papyrus wetlands.

Population densities in Western Uganda are already high and at the same time the region is confronted with a high population growth (3.4% per year). This, combined with low agricultural efficiency, has resulted in an ever increasing demand for natural resources (e.g. wood, water, etc.) and additional agricultural land in Uganda, deforestation, wetland degradation and soil erosion.

The area is typical example of a fast changing, data scarce region which is confronted with numerous ecological and other challenges. The methodology of this chapter is specifically developed for these kind of areas and can relatively easily be applied to similar regions.

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Chapter 2 - Water displacement by sewer infrastructure in the Grote Nete catchment, Belgium, and its hydrological regime effects.

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Abstract. Urbanization and especially increases in impervious areas, in combination with the installation of wastewater treatment infrastructure, can impact the runoff from a catchment and river flows in a significant way. These effects were studied for the Grote Nete catchment in Belgium based on a combination of empirical and model-based approaches. Effective impervious area, combined with the extent of the wastewater collection regions, was considered as an indicator for urbanization pressure. It was found that wastewater collection regions ranging outside the boundaries of the natural catchment boundaries caused changes in upstream catchment area between −16 and +3%, and upstream impervious areas between −99 and +64%. These changes lead to important intercatchment water transfers. Simulations with a physically based and spatially distributed hydrological catchment model revealed not only significant impacts of effective impervious area on seasonal runoff volumes but also low and peak river flows. Our results show the importance, as well as the difficulty, of explicitly accounting for these artificial pressures and processes in the hydrological modeling of urbanized catchments.

1. INTRODUCTION

Urbanization significantly impacts flow regimes and water quality of river systems (Jacobson, 2011; Paul et al., 2001). In particular, impervious areas exert several pressures on the hydrological cycle of catchments (Shuster et al., 2005). They affect infiltration, surface runoff and evapotranspiration, making the lateral processes potentially more important in urban settings than the vertical processes (Arnold et al., 1996; Becker et al., 1999; Brabec, 2009). These alterations in hydrological processes increase runoff peak flows and flood flashiness in rivers (Baker et al., 2004; Sheeder et al., 2002).

The effects of urbanization on peak flows have been studied in more detail compared to its effects on baseflow and low-flow events (Price, 2011). Baseflow represents stream flow fed from deep subsurface and delayed shallow subsurface storage (Ward et al., 1989), while low flow addresses dry season minimum flows (Price, 2011; Smakhtin, 2001). As urbanization reduces infiltration and recharge, it is generally expected that river baseflow is affected as well (Kauffman et al., 2009; Simmons et al., 1982). Baseflow can, however, also be strongly influenced by various types of anthropogenic activities in the catchment, such as water abstractions, sewer leakage or groundwater intrusion (Brandes et al., 2005; Seiler et al., 1999; Smakhtin, 2001; Wittenberg et al., 2010). Hence, the impact of urbanization on the different flow regimes is difficult to detect due to the strong temporal flow variations (weak signal-to-noise ratio). Although a weak tendency in baseflow decline and peak flow increase has been identified by some authors (Price, 2011), the combined effect of several anthropogenic and natural processes that influence baseflow and differences in assessment methodologies result in many remaining uncertainties in our understanding of baseflow behavior in periurban catchments (Hamel et al., 2013).

Sewers collect wastewater and, for combined sewer systems, also rainstorm water from pavements. They can also receive groundwater or leak wastewater to the groundwater system (Dirckx et al., 2009). The collected water is transported to a wastewater treatment plant (WWTP) and, after treatment, discharged into a receiving river. The WWTP thus aggregates water from the entire wastewater collection region (WWCR) and returns it to the environment at one single river location. Moreover, the WWCRs usually do not coincide with the catchment boundaries as they are mostly based on administrative borders. Consequently, the associated sewer infrastructure might transfer water between different natural (sub)catchments and further affect the natural hydrological processes in the catchment (Simmons and Reynolds, 1982). Recent research on catchment delineation considered incorporation of these changes in hydrological flow paths using semi-automated procedures (Jankowfsky et al., 2012). Such delineations, however, remain largely data dependent and time consuming.

Total impervious area (TIA) is considered to be an important indicator of the urban disturbance and an important land use characteristic. Imperviousness of urban areas is, however, very heterogeneous. Infiltration of impervious areas may not always be zero (Ragab et al., 2003). Impervious areas that are directly connected to the receiving river have a much larger effect on that receiving river (Boyd et al., 1994; Walsh et al., 2009). Some studies therefore suggest that the subset of impervious surfaces that route storm water runoff directly to streams via storm water pipes, also called effective impervious area (EIA), may be a better predictor of stream flow alteration (Roy et al., 2009; Shuster et al., 2005). Measurements of EIA are, however, much more difficult to obtain and therefore less commonly used in hydrological studies (Walsh et al., 2009).

Some studies have accounted for the difference between TIA and EIA in impact studies (Lee et al., 2003; Shuster et al., 2005). The traditional calculation of TIA and EIA might, however, be erroneous since the difference in boundaries between the natural river catchment and the WWCRs is typically disregarded. When impervious areas are situated within a river catchment, the surface runoff from these areas might drain to a WWTP located outside the catchment. The impervious areas in that case do not contribute to the runoff of the considered catchment.

Next to empirical statistical analysis, hydrological models can offer additional methods for studying the impact of changes in pervious and impervious areas on catchment hydrology. Such models can indeed help in complementing existing data and obtaining a better insight in the hydrological behavior of a catchment and the hydrological impact of urbanization. To allow the impact of spatial (e.g., landuse- related) scenarios to be assessed, fully spatially distributed hydrological process models (FDPM) are required. Such models give a spatially detailed and potentially reliable description of the hydrological processes in the catchment (Abbott et al., 1996; Ajami et al., 2004; Boyle et al., 2001; Carpenter et al., 2006; Refsgaard et al., 1996), but require a high amount of spatially explicit input data. After calibration of the large set of parameters in such models, a better match between simulated and observed hydrological variables may be obtained, but this does not necessarily mean that the model has a good accuracy. Model over parameterization and related parameter identifiability problems are well-known pitfalls (Beven, 1989; Jakeman et al., 2003; Muleta et al., 2005). These problems limit the applicability of such models. The FDPMs perform well in catchments where the hydrological processes are still close to natural runoff conditions, but are typically less accurate in urban areas due to the several (unknown or difficult to model) human influences (Vansteenkiste et al., 2013).

Discarding these anthropogenic influences can lead to a significant model bias and related impact assessments. In an urbanized environment with extensive sewer infrastructure, this might not only affect the performance of the catchment runoff and river flow simulation; it can also have indirect effects on parameterization of other land uses and over- or underestimate individual runoff components (Vansteenkiste et al., 2013). It has previously been demonstrated that if one does not differentiate between TIA and EIA in the hydrological model, this may result in a large model bias (Alley et al., 1983; Brabec, 2009). EIA is the most sensitive flow parameter in urban drainage models. Some authors have shown that calibration of this parameter may completely eliminate the bias in the results of these models (Kleidorfer et al., 2009; Willems et al., 1999).

This paper aims to quantify the importance of interbasin transfers and WWTP effluent flow contributions to downstream river flows for a selected river catchment in Belgium. The study makes use of measured river flows and effluent discharges from the different WWTPs installed in and outside the catchment. We evaluate the relative contribution of these WWTPs to the river flow, including peak and low flows. To understand the origin of the WWTP effluent discharges and the WWTP-induced water transfers between catchments, the sewer infrastructure and the EIA are assessed in a GIS environment, and compared with FDPM simulations for the study catchment. When implementing the FDPM, the abovementioned modeling issues (e.g., impervious area calculation and interbasin transfers) are considered. Based on empirical data analysis, model-based results and the comparison between both; we demonstrate the magnitude and importance of:

- water transfers across the catchment boundaries,
- water transfers across subcatchments within the catchment,
- the impact of EIA on river high and low flows and the performance of an FDPM.

We also discuss the implications the water transfers have on the FDPM-based impact analysis.

2. MATERIAL AND METHODS

2.1 Study area

The study catchment, the Grote Nete river (350 km^2), is situated in the north of Belgium. It has a maritime, temperate climate with average precipitation of 800mm/year. The catchment is composed of a mosaic of semi-natural, agricultural and urbanized areas, with a total population of 218 815 (Statistics Belgium, 2011). Urbanized areas are mainly situated around the town centers, but important parts of the urbanization are spread along the main roads connecting the different towns. As a result, the development of the sewer infrastructure is difficult, costly and time consuming.

Although the first WWTP in the catchment dates back from 1964, major investments in the sewer infrastructure only started 15 to 20 years ago (Dirckx et al., 2009). Nevertheless, large numbers of households are yet to be connected to the sewer infrastructure. The sewer system consists mostly of a combined system that collects both rain- and wastewater and is connected to sewer overflow devices (SOD) that are present at several locations in the catchment. Only a small, more recent part of the sewer system separates rain- and wastewater. Houses that are not yet connected to a sewer usually have a septic tank for basic treatment, after which the overflow drains to the nearest stream. The historical developments in the region and of the sewer system have led to a complex situation of connected and non-connected houses, roads and other impervious areas with or without rainwater separation.

2.2 Overview of the research approach

The approach and procedure to demonstrate the impacts of EIA connected to WWTPs and how they interlink in order to answer the research questions is visualized in Figure 2-1. (A) As a first step, empirical quantification and analysis of the impervious areas as well as the river and WWTP discharges was carried out (further referred to as the empirical analysis). (B) The empirical data of both EIA and WWTP discharges were used to develop three reduced rainfall scenarios and simulated in an FDPM to model the river flows in the catchment. (C) The empirical and modeled river flows and impacts of the WWTP impacts were intercompared in order to obtain an improved understanding of both catchment and model behavior.

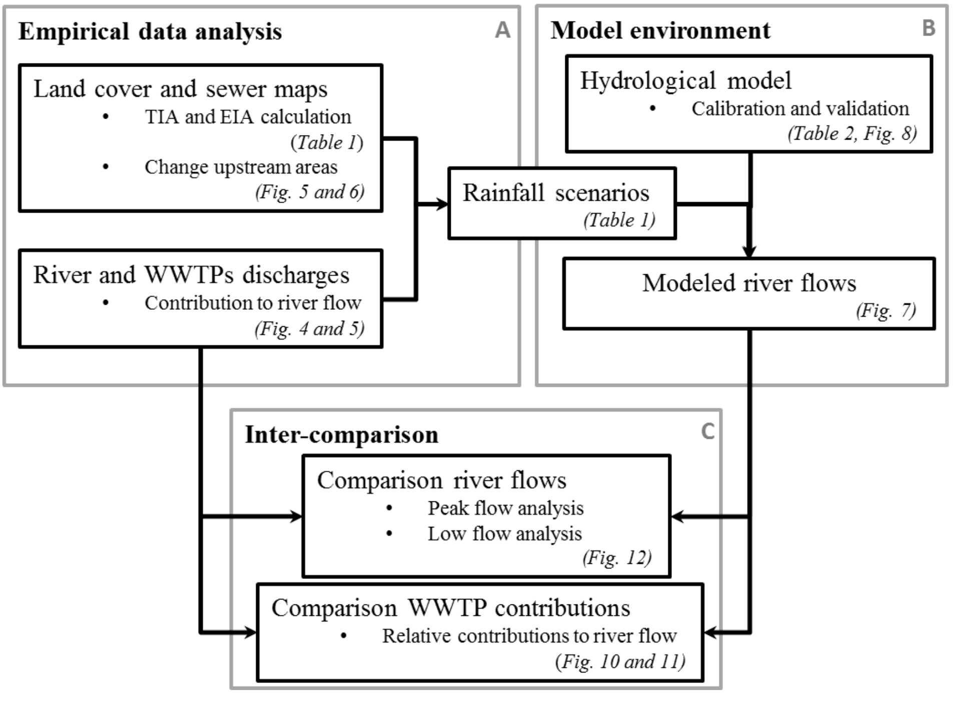

Figure 2-1: General overview of the different steps of the research methodology and their interlinks.

2.3 River flow and WWTP discharges

The Flemish Environment Agency (FEA) provided both hourly and daily mean river flow data (m^3/s) for the river gauging station situated at the outlet of the catchment (Varendonk) for the period 2004–2008, as well as effluent discharge data for the different WWTPs that are related to the catchment (Figure 2-2). To evaluate the overall impact of the WWTPs that discharge into the catchment (Mol and Geel), relative contributions of the WWTPs effluent discharges to the daily discharge of the Grote Nete river were calculated for the period 2004–2008. No discharge data were available for the WWTP of the military camp of Leopoldsburg. However, because of its small size (0.7% of total EIA), its impact on the river system is considered to be negligible. The hourly river flow data were used for model calibration and validation.

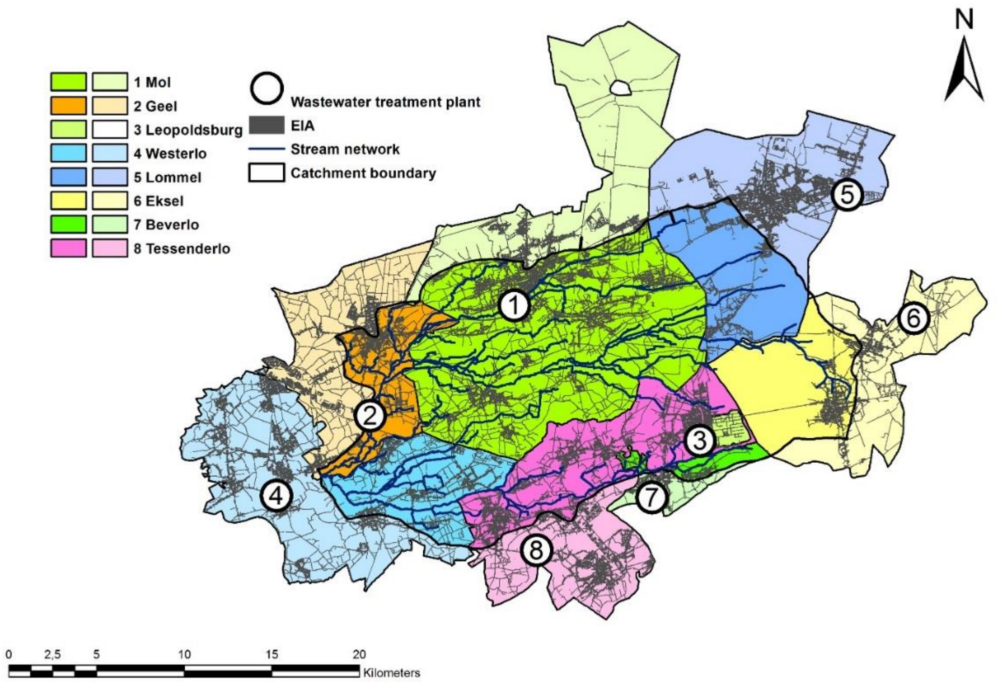

Figure 2-2: Overview of the different WWTPs and their WWCRs that are situated within the Grote Nete catchment (1–3) as well as the WWTPs that receive wastewater from impervious areas that are situated within the Grote Nete catchment but discharge into another catchment (4–8). Parts of the WWCR that are situated within the natural catchment boundary are shown with a brighter shade, illustrating the discrepancy between the areas covered by the natural catchment and the sewer systems.

2.4 Land use map and impervious area

The land use map used in this study was obtained from the National Geographical Institute (NGI) (NGI, 2007) and has a spatial accuracy of 1m. This land use map (1:10 000 vector layers) is based on aerial photographs from 1998 (1:21 000), ground-truthed and adjusted in the following years until 2007, when the map was published. It contains 47 different land uses. For the hydrological simulation purposes, a reduced set of nine categories was considered: evergreen needleleaf forest, broad-leaved woodland, mixed forest, open scrublands, grasslands, permanent wetlands, croplands, impervious area and water bodies (Table 2-1). These nine classes follow the International Global Biosphere Programme (IGBP) classification system based on their relationship to the modeled hydrological processes (Liu et al., 2004). Impervious areas include only completely sealed soils. Areas that can have reduced infiltration rates such as sandy roads (sand) or gardens are not considered to be part of the impervious areas. They are classified based on the most common characteristics of each land use type in the region (e.g., gardens are most often lawns).

The EIA is estimated on the basis of two thematic GIS layers representing the sewer system areas and are based on field observations done by different administrations. Houses connected to the sewer system are shown in zoning maps (one for each municipality). These maps indicate the connection of all individual households to the sewer systems and which ones drain directly to a nearby stream. The zoning maps also indicate which houses will be connected in the future and which buildings will have to install individual wastewater treatment plants (FEA, 2008b). In order to conduct all analyses based on the same input data, the zoning maps were used to identify the buildings, present in the NGI land use map, connected to each sewer system.

Streets that are connected to the sewer system were identified based on the polylines describing each sewer system (FEA, 2008a). Streets in the NGI land use map that overlap with the sewer system were assumed to contribute to the EIA. Sewers that separate waste- and rainwater were left out from this part of the analysis. Combining both methods resulted in one map from which the EIA of each WWCR was derived.

Table 2-1: Overview of the different land use classes used in the NGI map and the reclassification to 9 categories.

NGI description	IGBP vegetation
Coniferous trees	Evergreen needleleaf forest
Orchard Evergreen	needleleaf forest
Tree nursery	Evergreen needleleaf forest
Deciduous trees	Broad-leaved woodland
Poplar plantation	Broad-leaved woodland
Mixed deciduous and coniferous trees without dominance	Mixed forest
Mixed deciduous and coniferous trees with dominance of deciduous trees	Mixed forest
Mixed deciduous and coniferous trees with dominance of coniferous trees	Mixed forest
Sand	Open scrublands
Bare ground	Open scrublands
Coppice	Open scrublands
Heath	Open scrublands
Heath with deciduous trees	Open scrublands
Heath with coniferous trees	Open scrublands
Scrubs	Open scrublands
Brushwood	Open scrublands
Brushwood with scrubs	Open scrublands
Cemetery	Grasslands
Beds	Grasslands
Pasture	Grasslands
Gardens	Grasslands
Deep swamp	Permanent wetlands
Reedland	Permanent wetlands
Cropland	Croplands
Transformer station	Impervious area
Railway	Impervious area
Road	Impervious area
Crossroad	Impervious area
Industrial building (in use)	Impervious area
Industrial building (abandoned)	Impervious area
Warehouse	Impervious area
Silo	Impervious area
Greenhouse	Impervious area
Cooling tower	Impervious area
Non-university hospital	Impervious area

Town hall	Impervious area
Schoolhouse	Impervious area
Firehouse	Impervious area
Commercial building	Impervious area
Religious building	Impervious area
Sports hall	Impervious area
Covered grandstand	Impervious area
Non-covered grandstand	Impervious area
Indoor swimming pool	Impervious area
Building for drinking water supply	Impervious area
Normal building	Impervious area
Building for public use	Impervious area
Watercourse	Water bodies
Pond	Water bodies
Sluice	Water bodies

2.5 Upstream area calculations

Subcatchments were delineated based on a 1 : 5000 digital elevation model expressed as a 5m raster (FEA, 2005, 2006). For each stream junction (n = 131), upstream areas were calculated using the method discussed in Jenson et al. (1988) (further referred to as the runoff method). By combining these upstream areas with the 1m raster of the land use map, we calculated upstream impervious area for each stream junction.

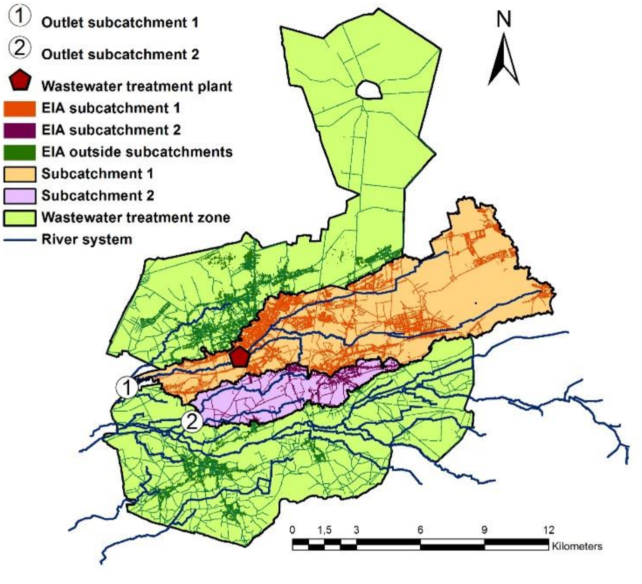

Figure 2-3: Example on the calculation of the upstream areas. The WWTP discharges into subcatchment 1 (orange colored). Therefore, the EIAs within the WWCR, but outside subcatchment 1 (green and purple colored), are included in the upstream area of subcatchment 1. As a result, the area of this subcatchment increases by 404 ha of impervious area or by 5.1% of the total area. Because the EIA is removed from subcatchment 2 (purple color), the area of subcatchment 2 decreases by 69 ha or 4.3% of the total area.

Next the sewer infrastructure was considered (further referred to as the sewer method) (Figure 2-3). In this method, the upstream areas were recalculated by removing the EIAs from their natural subcatchments and adding these EIAs to the subcatchment of the river reach into which the WWTP discharges. As zoning maps also indicate which houses yet have to be connected to the WWTPs, expected upstream areas for the near future were obtained as well. Differences in upstream impervious areas and total areas between the runoff method and the sewer method were considered as indicators of how strongly the catchment is affected by the sewer infrastructure. All GIS calculations were performed in ArcGIS 9.3 (ESRI Inc., 2009).

To make an evaluation of the impact of the sewer infrastructure on the catchment's overall water balance, the changes in upstream area and upstream impervious areas (TIA) between both methods were calculated for the 131 stream junctions in the catchment. The relative changes (%) were analyzed by means of histograms.

2.6 MIKE SHE model set-up

MIKE SHE is a spatially distributed, physically based hydrological model (Abbott et al., 1996). It simulates the terrestrial water cycle including evapotranspiration (ET), overland flow, unsaturated soil water and groundwater storage and movements (DHI Water and Environment, 2008; Feyen et al., 2000; Refsgaard et al., 1995). The MIKE SHE model has been used worldwide for a wide range of applications (Refsgaard, 1997; Sahoo et al., 2006; Sun et al., 1998; Thompson et al., 2004; Zhang et al., 2008). For this study, a spatially distributed, physically based hydrological model was selected over other types of hydrological models as it can simulate the effect of spatially differentiated scenarios. It allowed us to evaluate the effect of changes in spatial patterns of surface runoff on the hydrological regime. The model was also used for other research purposes (Vansteenkiste et al., 2013). The representation of catchment characteristics and input data (digital terrain model, land use, soil) in MIKE SHE are provided through raster information. The MIKE SHE model for the Grote Nete catchment was built on a 250m grid. It was developed with physics-based flow descriptions only for those processes that are relevant for the purposes of this study, i.e., overland and unsaturated flow. Given that the study focusses on spatial scenarios at the surface (changes in surface runoff), groundwater flow processes are considered to be secondary. Therefore, the saturated zone was implemented through simplified lumped process descriptions, while surface processes were modeled in a spatially variable way (see Vansteenkiste et al. (2013), for details). The applied model configuration is schematized in Figure 2-4 (Graham et al., 2005).

Hourly data from six rain gauges were used to describe the spatial variability of the rainfall over the catchment and used for meteorological input after applying Thiessen polygons. Only one potential evapotranspiration series was acquired from the national meteorological station located at Uccle, 30 km west of the study area, and applied. The growing cycle of the different crops was considered by means of a vegetation database that included leaf area index (LAI) and root depth (RD) and was based on Rubarenzya et al. (2007). Additional empirical parameters for determining the evapotranspiration of the crops were assessed from the literature (DHI Water and Environment, 2008; Kristensen et al., 1975). The overland flow component was determined by the Strickler roughness coefficient, detention storage and initial water depths. The surface roughness was based on values from the literature (Chow, 1964) as a function of land use. Standard values were taken for the detention storage and initial water depths and are considered constant over the entire catchment (DHI Water and Environment, 2008). The MIKE SHE model was coupled to a full hydrodynamic river model, implemented in MIKE 11 (DHI Water and Environment, 2009) to route MIKE SHEs overland flow to the catchment outlet and account for the hydraulic effects of the river network and its infrastructure. The river network comprised the main branches in the catchment, which were extracted from the Flemish hydrological atlas (FEA, 2005). The geometry of each river branch was specified in terms of cross sections obtained from field survey data. All infrastructures that were expected to have a significant impact on the river flow, such as bridges, culverts and weirs, were implemented in the model. For the unsaturated soil water component of the catchment model, soil moisture characteristics were defined by means of the model by Brooks et al. (1966) for soil retention curves, and the equation by Averjanov (1950) for the soil hydraulic conductivities. The unsaturated zone parameters, needed to identify these relations, were based on the USDA – United States Department of Agriculture – soil information database. As mentioned above, the saturated zone was implemented through baseflow reservoirs applying simplified lumped storage and flow descriptions and parameters. More specifically, the entire groundwater system was divided into a series of shallow interflow reservoirs plus two deep baseflow reservoirs. These reservoirs allowed for differentiating between fast and slow components of baseflow discharge and storage. An overview of the most important model parameters in the considered model configuration is presented in Table 2-2.

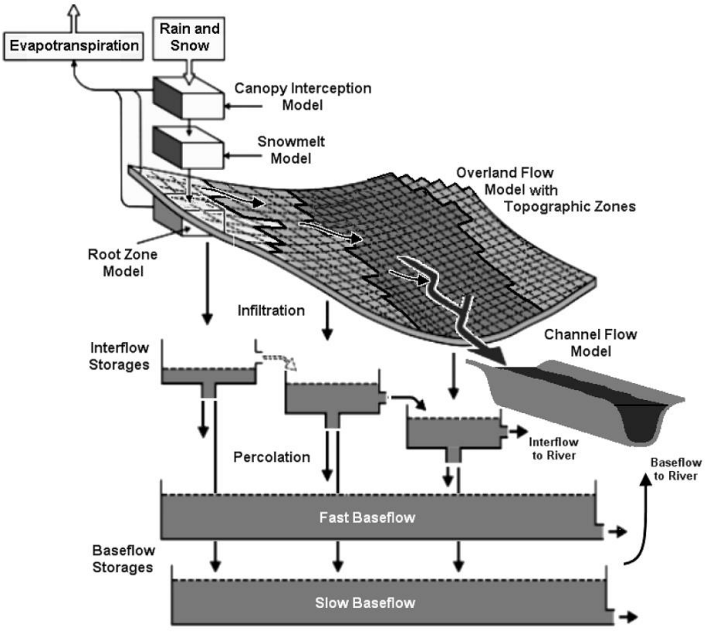

Figure 2-4: Schematic representation of the applied MIKE SHE model configuration (Graham and Butts, 2006).

Soil characteristics were derived from the USDA soil parameters classification system (Graham et al., 2005). Saturated zone flow was simulated using the linear reservoir method. The entire groundwater system was divided into a series of shallow interflow reservoirs plus two deep baseflow reservoirs. These reservoirs allowed for differentiating between fast and slow components of baseflow discharge and storage. Water was routed through the linear reservoirs as interflow and baseflow and subsequently added to the MIKE 11 river network as lateral inflow in the lowest interflow reservoir (Graham et al., 2005).

Table 2-2: Overview of the MIKE SHE model parameters in the considered model configuration.

Component	Parameter	Description	Unit
Evaporation	Cint	Canopy interception	mm
	c1, c2, c3	Evapotranspiration empirical parameters	mm/day
	Aroot	Root distribution index	1/m
	Kc	Crop coefficient	-
	LAI	Leaf area index	-
	RD	Root depth	mm
	M	Strickler roughness coefficient	$(m^3/s)-1$
Overland flow	DS	Detention storage of the ground surface	mm
	Hini_OF	Initial water depth on the ground surface	mm
	θsat	Saturated soil water content	m^3 m−3
Unsaturated flow	θFC	Soil water content at field capacity	m^3 m−3
	θWP	Soil water content at field wilting point	m^3 m−3
	θres	Residual soil water content	m^3 m−3
	Ksat	Saturated hydraulic conductivity	ms−1
	n	Averjanov empirical constant	-
	Hini_CF	Initial water level	mm
Channel flow	Qini_CF	Initial discharge	m^3/s
	n	Bed resistance	$(m^3/s)-1$
	Sy_IF	Specific yield for interflow reservoir	-
Groundwater flow	Hini_IF	Initial depth of the interflow reservoir	m
	Htreshold_IF	Threshold depth of the interflow reservoir	m
	Hbottom_IF	Bottom depth of the interflow reservoir	m
	tpercolation	Percolation time	days
	RCIF	Interflow time constant	days
	αS	Fraction of percolation to the baseflow reservoir	-
	Sy_BF	Specific yield for the baseflow reservoir	-
	RCBF	Baseflow time constant	days
	αUZ	Unsaturated zone feedback fraction for baseflow	-
	Hini_BF	Initial depth of the baseflow reservoir	m
	Htreshold_BF	Threshold depth of the baseflow reservoir	m
	Hbottom_BF	Bottom depth of the baseflow reservoir	m

2.7 Implementing the hydrological influence of the sewer infrastructure in MIKE SHE

To model the effect of the EIA on the hydrological regime of the Grote Nete catchment, the detailed land use map (1:5000) and EIA had to be resampled to the MIKE SHE model grid specifications (250m). Despite the careful resampling for preserving the catchment land use in the model, an overestimation of TIA by 4.2% remained. For each WWCR, the urban area and EIA were extracted and the percentage EIA per urban area WWCR calculated. These calculations were used in combination with the resampled urban area per WWCR in MIKE SHE to define the fraction of rainfall discharged by the sewer infrastructure to the river. Table 2-3 presents the percentage of EIA per WWCR and its urban area.

Incorporating the impact of the sewer infrastructure within the MIKE SHE model can be done in two different ways. This basically involves the removal of the surface runoff that is going to the sewer network from the total catchment runoff. This surface runoff to the sewage system can then be added to the river network at the WWTP discharge location as a point source, after accounting for the sewer WWTP routing time delay. To remove the sewer runoff from the catchment runoff, one of the first solutions is to take out, from the modeling domain, the grid cells that cover the impervious areas and that contribute to the sewer system. The problem encountered here in this study is that none of the 250m grid cells are fully covered by that type of impervious surfaces. Only fractions of the grid cell areas contribute to the sewer system, making the removal of the grid cells impossible. Therefore, we opted for the second solution: reducing the rainfall input proportional to the fraction of the sewer runoff contribution. This allowed us to take better in to account the fractions of impervious areas within each grid cell.

Three different rainfall scenarios were developed to assess the impact of the sewer system. For each scenario, the total measured rainfall in the catchment was reduced in relation to the assessed WWTP discharges. The rainfall reductions were spatially differentiated within the catchment by reducing the different rainfall series based on the overlap between the Thiessen polygons (different point rainfall input series) and the different WWCR regions (amount of EIA). The different scenarios of reduced rainfall were applied within the model to assess its impact in the model. Scenario 1 considered a reduction in rainfall within the WWCRs that discharge within the catchment to assess the impact of the sewer infrastructure on the river flows. Scenario 2 implemented a reduction in rainfall within the WWCRs that discharge outside the catchment to assess the impact of water transport outside the catchment. Scenario 3 took a reduction in rainfall across the entire catchment to evaluate the impact of all the sewers on the river system (Table 2-3). The original measured rainfall input series, applied to calibrate the model, is further referred to as the reference scenario.

The differences in runoff discharges between the initial model result and the simulations with reduced rainfall input gave us indications of the impact of the sewer infrastructure on the catchment runoff. The model results were compared for the different scenarios and assessed on an hourly, daily and monthly basis. The reductions in flow because of reduced rain were compared to the measured WWTP discharges as well as their relative contributions to the total river flows. Differences in relative contributions were calculated between the reference scenario and the rainfall scenarios 1 and 3. Changes in peak and low flows were evaluated in relation to the empirical return period (mean recurrence interval of these flows).

Table 2-3: Different variables used to implement the reduced rain scenarios: EIA in the catchment (ha), EIA per WWCR (%) and EIA per urban area unit (%) based on the NGI data.

WWTP	EIA per WWCR (ha)	EIA per WWCR (%)	EIA per urban area WWCR (%)	Scenario 1	Scenario 2	Scenario 3
Mol	782.07	5.17	72.77	*		*
Geel	284.31	7.2	67.42	*		*
Leopoldsburg	32.04	6.99	48.5	*		*
Tessenderlo	418.85	6.58	76.21		*	*
Westerlo	182.96	4.77	67.22		*	*
Beverlo	41.14	5.43	87.34		*	*
Lommel	145.43	2.75	47.57		*	*
Eksel	101.6	2.27	74.04		*	*

* indicates the WWRCs for which the rain was reduced in each scenario.

2.8 MIKE SHE model calibration

After completing the model setup, the MIKE SHE model was calibrated. Note that the MIKE SHE model code comprises numerous free parameters, whereas the guiding principle for complex models like MIKE SHE is to calibrate the model on as few free parameters as possible (Refsgaard et al., 1995). Therefore, the calibration parameters were reduced by a parameter sensitivity analysis similar to Xevi et al. (1997) and Thompson et al. (2004). The results of this sensitivity analysis for the Grote Nete model are not presented here, but can be found in Vansteenkiste et al. (2013). They show that the most sensitive parameters are the surface roughness and saturated zone parameters. These parameters are mainly related to the groundwater computations, but also have a strong influence on both low-flow and peak-flow magnitudes. In the end the model was calibrated using 14 parameters per grid cell related to the distributed raster information, and 20 catchment-wide parameters related to the groundwater flow and evapotranspiration processes.

Calibration of the model was done against hourly stream flow measurements at the catchment outlet for the time period 2004–2006, while the years 2007 and 2008 were used for model validation. The most sensitive parameters were iteratively and manually adjusted between predefined limits until maximal correspondence between measured and predicted discharge runoff downstream of the catchment was achieved. The predefined parameter value limits represent the physically acceptable intervals and have been assessed on the basis of previous modeling studies of the Grote Nete catchment (Rubarenzya et al., 2007; Woldeamlak et al., 2007) and the literature (Anderson et al., 1991; Chow et al., 1988; DHI Water and Environment, 2008).

The model correspondence was evaluated both qualitatively by visual inspection of the runoff results and quantitatively using goodness-of-fit statistics, including mean error (ME), root-mean-squared error (RMSE), correlation coefficient (R) and Nash–Sutcliffe efficiency (NSE) (Nash et al., 1970). Because the aim of this study was to investigate the impact on both high and low river flow conditions, independent peak and low flows, extracted from the time series using the method of Willems (2009), were also explicitly validated. This was done in scatterplots of simulated versus observed values as well as by means of empirical frequency distributions (peak and low flows versus return periods). The return periods of peak and low flows were calculated empirically as the total length of the available time series (in years) over the peak- and low-flow rank (1 for highest, 2 for second highest, etc.). Box–Cox transformation was applied to the simulated and observed peak and low flows to reach homoscedastic model residuals (Willems, 2009). This means that the model residuals can be represented by one distribution and equal weight is given to the peak- and low-flow values. The RMSE of the model residuals after transformation was optimized during model calibration.

3. RESULTS

3.1 River flow contribution of WWTPs

Between January 2004 and December 2008, the Grote Nete had an average observed discharge of 3.95m³/s at the catchment outlet. The upstream WWTPs discharged for the same period on average 0.31 m³/s of wastewater to the Grote Nete, or 7.9% of the river flow. Discharges of both the river and WWTPs, however, varied substantially in time (Figure 2-5). Rain events always lead to strong changes in river flow. For example, in 2007 there was a noticeable reduction in baseflow during spring and summer, followed by a strong increase during the winter period. In 2008 several rain periods led to a higher average flow during spring and summer. Monthly mean discharges of WWTPs and monthly mean river flows were found to be well correlated ($r^2 = 0.72$, $p < 0.001$). Correlation between daily mean WWTP and daily mean river discharges was, however, lower ($r^2 = 0.60$, $p < 0.001$).

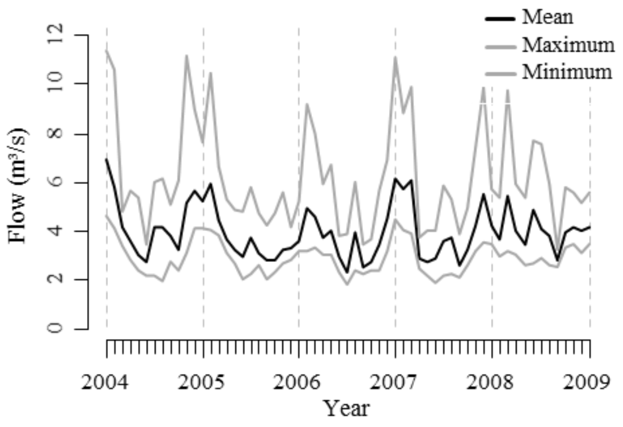

Figure 2-5: Monthly mean, minimum and maximum of the daily measured flow (m^3/s) for the period 2004–2008 at the outlet station (Varendonk) of the Grote Nete catchment.

In general, the WWTPs were found to contribute between 5.5 and 13.1% of the monthly average river flow at the Grote Nete catchment outlet (Figure 2-6). On a daily basis the contribution of the WWTPs to the river flow can decrease to 5.5% during wet periods or increase up to 23.6% during dry summer periods. The highest relative contributions were observed for rain events that occur during low river flow periods (e.g., convective thunderstorm periods after long, dry summer periods).

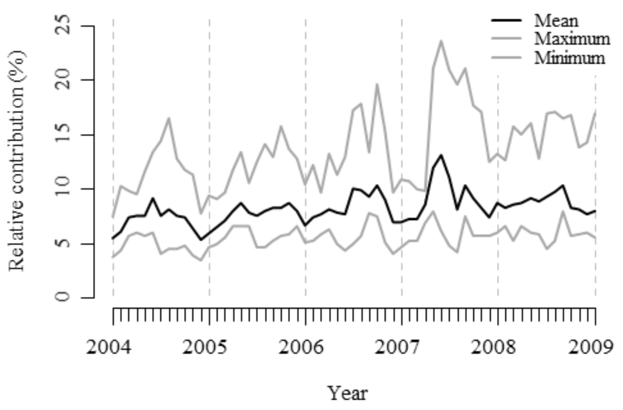

Figure 2-6: Monthly mean, minimum and maximum relative contribution of the WWTPs (Mol and Geel) to the river flow at the outlet station (Varendonk) of the Grote Nete catchment.

3.2 Water transfers between catchments and subcatchments

3.2.1 *Current situation*

From the analysis of the WWCRs we conclude that there are significant water transfers between the Grote Nete catchment and adjacent catchments. Of the total of 2836 ha TIA in the catchment, 1661 ha are currently connected to the WWTPs. This gives an initial ratio of 0.6 between TIA and EIA. Only 54.0% of the EIA drains water that remains inside the catchment, the rest, mostly situated in the southern part of the catchment, drains water outside the catchment. At the same time waste-, ground- and rainwater from 461 ha, mostly from the north, is transported from outside to inside the catchment. If the difference in boundaries between catchment and WWCRs is taken into account, the EIA for the catchment is considered to be 1361 ha. This represents 4.0% of the entire catchment area.

Upstream impervious areas and total upstream area change substantially when the WWCRs are incorporated into the calculations. A comparison between both calculation methods for 131 river junctions illustrates this impact. By taking the WWCRs into account, total upstream impervious areas decrease up to 99 %, as for most subcatchments impervious areas are connected to a WWTP located outside the subcatchment. For other river junctions, the upstream areas increase up to 64% because a WWTP is situated upstream of the junction (Figure 2-7a). For the same reason, the change in total upstream area varies strongly (Figure 2-7b).

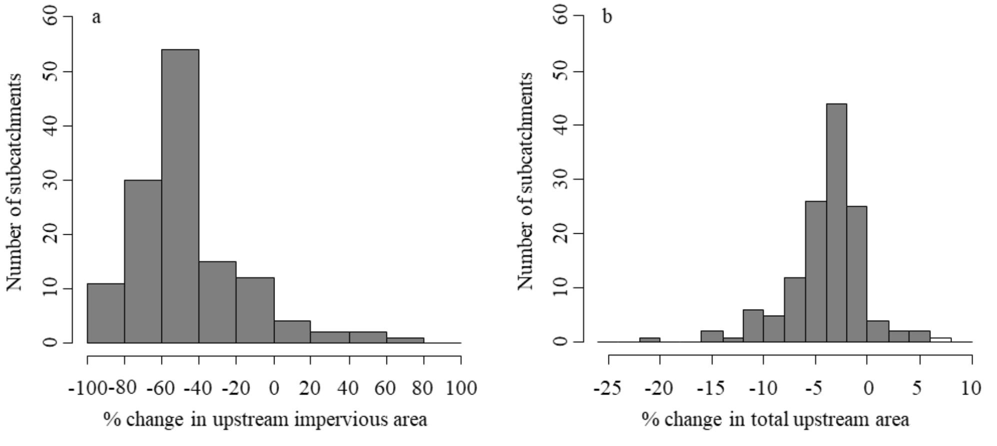

Figure 2-7: (a) Histogram of the change in upstream impervious areas that demonstrates the impact of the sewer system on both upstream area calculations. For different stream junction points (n = 131) in the catchment, upstream impervious areas were calculated based on the natural catchment and after taking the sewer system into account. Differences between both types of upstream impervious areas were calculated as percentage change for each junction. (b) Histogram of the change in total upstream areas that demonstrates the impact of the sewer system on both upstream area calculations. For different stream junction points (n = 131) in the catchment, total upstream areas were calculated based on the natural catchment and after taking the sewer system into account. Differences between both types total upstream areas were calculated as percentage change for each junction.

3.2.2 *Future developments*

When the WWCR zoning plans are fully implemented in the future, another 245 ha of impervious areas will be connected to the WWTPs. Of those 245 ha, the surface runoff of 141 ha will be transported to other catchments, while the surface runoff of 148 ha will be imported from neighboring catchments.

When all subcatchments are evaluated, it is seen that most of the river junctions will experience an extra reduction in upstream area by 1 or 2% (Figure 2-8). Ten river junctions will, however, experience an increase of their upstream area by 1 or 2% because of the upstream presence of a WWTP.

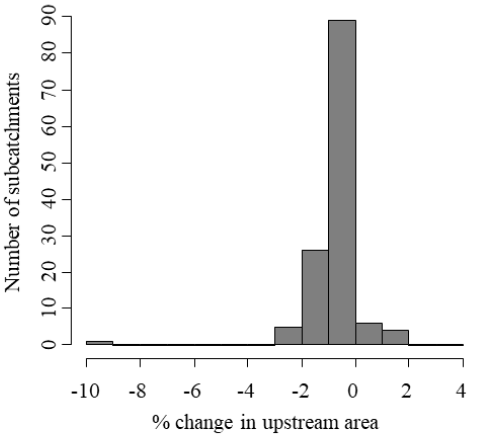

Figure 2-8: Histogram of the change in upstream areas after full implementation of the zoning plans. For different stream junction points (n = 131) in the catchment, total upstream areas were calculated by comparing the current sewer system and the future sewer system after full implementation of the zoning plans. Differences in total upstream areas are given as percentage change for each stream junction.

3.3 Model calibration and validation

In comparison with Vansteenkiste et al. (2013) the main difference in the parameterization of the MIKE SHE model was related to the saturated zone component (Table 2-4) and the fine tuning of the overland flow roughness parameters (Table 2-5)

Table 2-4: MIKE SHE calibrated parameters for the saturated zone.

	Interflow reservoir	Baseflow reservoir 1	Baseflow reservoir 2
Specific yield [−]	0.22	0.2	0.2
Initial depth [m]	15	30	30
Bottom depth [m]	15.065	40	40
Threshold depth [m]	15	30	30
Time constant [days]	4	12	12
Percolation time constant [days]	1.5	–	–
Fraction of percolation [−]	0.82	–	–
UZ feedback fraction [−]	–	0.32	0.12

Table 2-5: MIKE SHE calibrated parameters for the surface.

	Strickler roughness coefficient [$m^{1/3}/s$]
Deciduous needleleaf forest	2.5
Deciduous broadleaf forest	1.25
Mixed forest	1.82
Grasslands	6
Permanent wetlands	2
Croplands	13
Urban and built-up	90
Water bodies	90

Table 2-6 shows the model performance statistics ME, RMSE, R and NSE. These demonstrate the good general model performance. The statistics, however, demonstrate that the model performance is slightly better in the calibration period than in the validation period. Figure 2-9 shows the observed and simulated hourly runoff series for the calibration period.

Additional verification of the model performance for the high- and low-flow extremes is presented in Figure 2-10. The observed independent high- and low-flow extremes are plotted against the simulated ones after Box–Cox transformation. These validation plots allow for evaluation of the model for its ability to predict extreme conditions. The model is able to simulate the extreme peak flows well, while the low-flow extremes are slightly overestimated by the model. The ME is very small for the peak flows (0.05 m^3/s) and larger for the independent low-flow extremes (0.14 m^3/s). Based on the good general model performance for total flows in both calibration and validation periods and for peak flows, the model was considered applicable for assessing the impact of the water transfers on these flow variables as a result of the sewer infrastructure.

Table 2-6: Statistical performance of total hourly river flows for the model calibration and validation periods at the outlet station of the Grote Nete catchment.

	Calibration	Validation
ME [m^3/s]	0.6	0.72
RMSE [m^3/s]	0.84	0.93
R [−]	0.88	0.84
NSE [−]	0.72	0.63

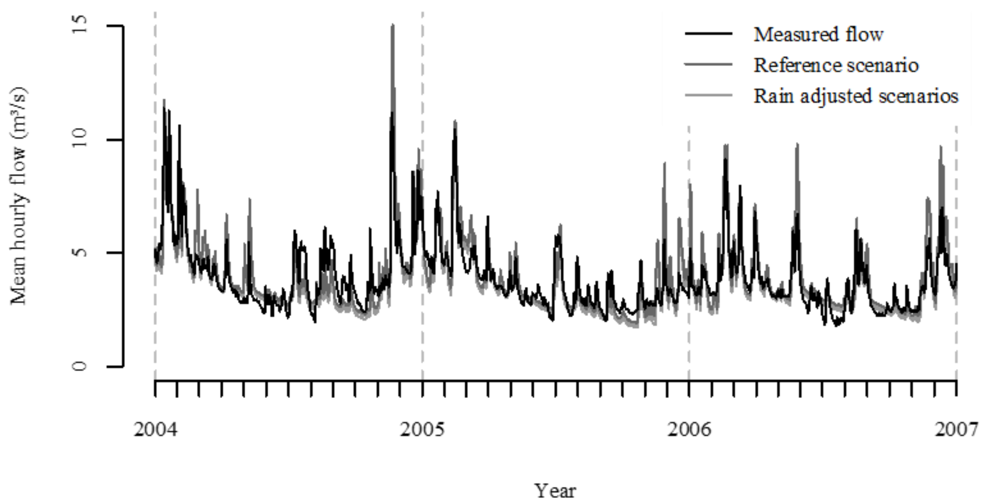

Figure 2-9: Observed and simulated hourly river flow series for the model calibration period on a daily time step.

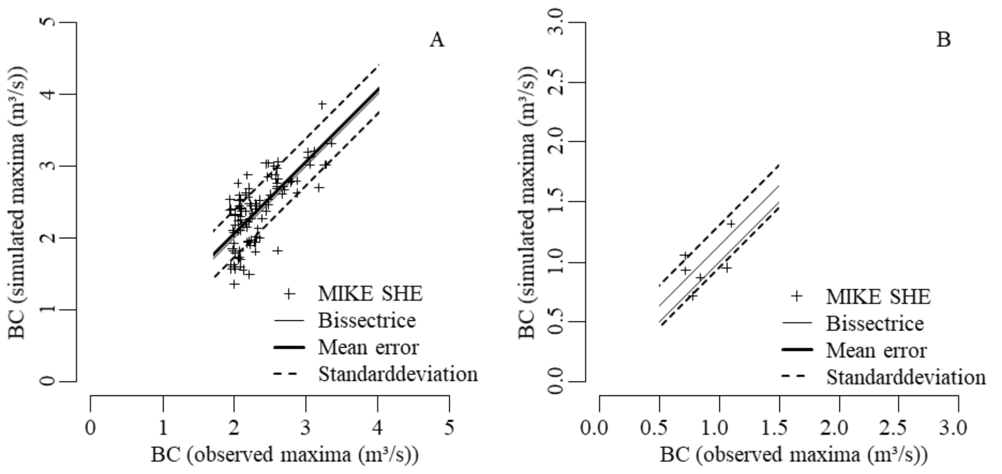

Figure 2-10: A) Scatter plot of simulated versus observed independent hourly peak flows at the outlet station of the Grote Nete catchment after Box–Cox transformation ($\lambda = 0.25$). B) Scatter plot of simulated versus observed independent hourly low flows at the outlet station of the Grote Nete catchment after Box–Cox transformation ($\lambda = 0.25$).

3.4 Comparison with model impact results

3.4.1 *River flow impact of WWTPs*

As a first step, the different rainfall scenarios are compared with the reference scenario. Figure 2-11 shows the model based differences in mean monthly river flows between the reference scenario and the adjusted rainfall scenarios. Based on this difference, the relative contributions of the EIA to the total river flow were obtained. These relative contributions vary between 2.2 and 7.2% for scenario 1. For scenario 2 these contributions vary between 2.8 and 6.1 %.

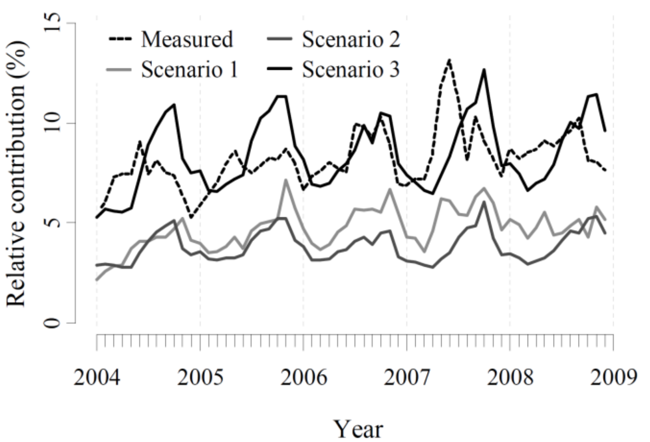

Figure 2-11: Relative contribution (% monthly mean total flow) of the WWTPs to the total river flow. Measured refers to the empirical results obtained in Sect. 3.1.

3.4.2 *Seasonal variation in river flow impact*

A seasonal change in relative contribution of the WWTP infrastructure to the river flow was found. The largest contributions to the overall flow were found during summer and lowest during winter periods. The effect is, however, again less pronounced compared to the relative contribution based on the empirical analysis (Figure 2-11). A comparison is made between the model-based impact results and the empirical analysis of Sect. 3.1, where the river flows at the outlet station are adjusted for the connected areas. Results of that comparison show strong seasonal patterns in differences between the model-based and empirical analysis results (Figure 2-12). Especially during the period of declining flows (flow recession periods) in spring and the beginning of summer, the model simulates much lower relative contributions of WWTP discharges compared to the empirical analysis. This difference is less pronounced or absent for the summer of 2008.

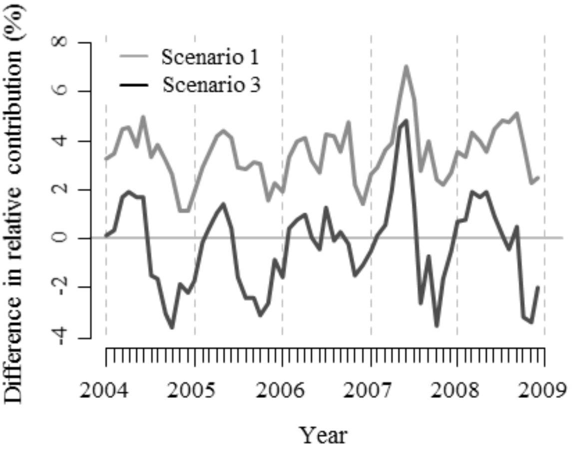

Figure 2-12: Difference in relative contribution of the WWTPs to the total river flow at the outlet station of the Grote Nete catchment (% difference in monthly mean total flow) between the model-based and empirical analysis results (rainfall scenario 1 and 3). The difference in contribution increases during flow recession periods.

3.4.3 *Impact on peak and low flows*

The model-based impact results of the scenarios result in a decrease of both peak (Figure 2-13a) and low flows (Figure 2-13b) for given return periods. For events with an empirical return period higher than 1 year, both peak and low flows decrease proportional to the reduced rain scenarios. The effects thus are stronger for the low-flow event compared to the peak flow events (Table 2-7).

Table 2-7: Absolute and relative changes in peak and low flows at the outlet station of the Grote Nete catchment for empirical return periods higher than 1 year and the different rainfall scenarios compared with the reference scenario.

	Peak flows		Low flows	
	m^3/s	%	m^3/s	%
Scenario 1	0.33	3	0.3	5.5
Scenario 2	0.3	2.8	0.11	4.9
Scenario 3	0.62	6	0.23	10.6

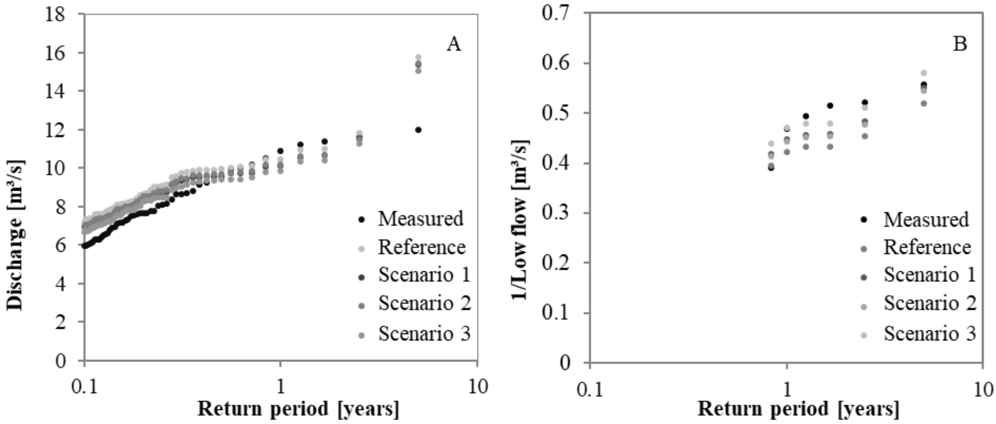

Figure 2-13: Return period of hourly peak flow (A) and low flow (B) extremes for the Grote Nete catchment between 2004 and 2008.

4. DISCUSSION

4.1 Impact of WWTPs on the river baseflow

The overall impact of the WWTPs on the river flow depends on the timescale (daily, monthly or yearly). On average, the WWTPs are responsible for about 10% of the catchment's discharge, but the relative WWTP contributions to the river flow can be significantly higher at short timescales and during dry periods (Figure 2-6). The sewer infrastructure in the catchment is hence found to be an important point source of water in the rivers.

The high WWTPs effluent contribution to the total mean river discharge is due to the combined effect of different sewer-infrastructure-related processes: wastewater collection, rainwater runoff and groundwater intrusion. That parasitic groundwater, due to the groundwater intrusion, can be high in the region as has also been shown before by Dirckx et al. (2009). Due to this draining of the groundwater table, drought-related problems induced by the urbanization will further increase. Climate change scenarios for Flanders predict a strong decrease in river low flows during summer (Baguis et al., 2010; Vansteenkiste et al., 2013). The impact of the WWTPs on the overall flow is thus expected to increase in the future.

While the impact of the WWTPs on a river flow can be evaluated rather easily, the impact of connected impervious area on the flow regime of smaller reaches within the WWCR is more difficult to quantify. Often the roofs of buildings and pavements of a catchment are connected to sewers that transport storm and wastewater to a WWTP, which might be located outside the natural catchment boundary. If we would like to evaluate these changes, long-term river flow data need to be available that encompass also river flow data prior to the sewer development. Also a detailed inventory of the gradual expansion of the sewer infrastructure would be required.

4.2 TIA versus EIA

Impervious area is a landscape metric that is widely used as an indicator of water quality, quantity and river ecosystem health (Jacobson, 2011). In the empirical analysis, conducted in this study, the impact of the sewer infrastructure on the impervious areas within the catchment was evaluated. The proposed method allowed us to make a distinction between TIA and EIA and to evaluate both transfers between catchments and subcatchments. Whereas both upstream TIA and EIA were found to be useful indicators of the hydrological and ecological disturbance, the EIA is in general considered to be a better indicator for the anthropogenic impact on the hydrological regime (Roy et al., 2009). The EIA of the catchment decreased significantly when the different WWCRs were incorporated into the calculation. Large parts of the connected impervious areas within the catchment do not contribute to the river flows inside the catchment but are drained to a neighboring catchment.

As a result, the overall impact of urbanization within the catchment can be over- or underestimated. At the same time large amounts of wastewater are transported from outside the catchment. These changes to the natural system affect both the spatial and temporal distribution of runoff water in the catchments: from spatially distributed runoff to point inflows, from one catchment to another, increased surface runoff, reduced groundwater infiltration, shorter travel time and hence higher temporal and spatial aggregation, etc.

The changes in the total upstream areas, between −16 and +3 %, and upstream impervious areas, between −99 and +64 %, in our study were found to be large. For most of the subcatchments in our research area, both total upstream area and upstream impervious area decreased significantly. Although these subcatchments do not have actual upstream EIA, they are affected by the reduction in upstream impervious areas and the resulting decreased total upstream area. These reductions in impervious areas and the transfer of storm and wastewater to other reaches can lead to reductions in the flow regime and changes in related river characteristics. The actual absence of upstream impervious areas, due to wastewater allocation, might be more important than the presence of only a small portion of upstream impervious area.

In contrast to many other studies we were able to use high resolution data that are based on manual field observations instead of less accurate remote sensing data. The use of proxies for impervious areas, as used in other studies (Chabaeva et al., 2009), was not necessary. The same counts for the calculation of the EIA. Despite the high resolution of the data, some uncertainties remain in the impervious area classification (e.g., use of NGI classes and its influence on the TIA calculation) and EIA calculation (e.g., actual connection between road surfaces and the underlying sewer system). However, we expect that both metrics, TIA and EIA, are close to the actual situation in 2008 during low-flow periods. Nevertheless, due to increasing urbanization and sewer development, both indicators require a regular update.

SODs can have a profound impact on the hydrology, especially during extreme rain events. But their responses to these rain events can vary widely and are difficult to predict. Although relevant to the study area, the available data were not sufficient to incorporate SODs into the river flow analysis or into the EIA calculations or model development. It is expected that incorporating SODs in the analysis would result in a reduction of the EIA during extreme rain events. If such extreme rain events were to be analyzed explicitly, SODs should be integrated in the EIA calculations.

4.3 Model impact results

As opposed to the empirical analysis, the model-based results allow for explicit consideration of the catchment runoff dynamics, the highly non-linear hydrological responses to the changes in impervious areas and the interactions between different runoff components. However, the use of hydrological models has, as is the case for all models, limitations. Traditional hydrological models impose restrictions on how to deal with sewer infrastructure. In this study we evaluated the impact of the WWTPs by means of an existing, calibrated MIKE SHE hydrological model. Rainfall series were reduced proportional to the EIAs within the catchment to enable simulation of the impact of the surface runoff from impervious areas to the sewer system. This method is an alternative to the actual integration of a sewer model in the catchment model, which was not possible for reasons of data availability and model characteristics. Despite these simplifications of reality, the consistencies between the empirical and model-based results gave us confidence in the impact results. However, using this method it is not possible to evaluate the impact of the EIA situated outside the catchment. The latter impact evaluation would require sewer models to be integrated with the catchment hydrological model.

Previous studies have demonstrated the difficulties models can have to describe baseflows in (peri-)urbanized areas (Elliott et al., 2010). Our model results revealed overestimations of low flow, while other recent studies have reported underestimations (Furusho et al., 2013). Hydrology in periurban catchments is typically a combination of fast and slow hydrological responses (Braud et al., 2013). The specific weighing of both responses is in general case specific and a result of the historical developments in the anthropogenic system. Although some characteristics in our catchment (e.g., fragmented land uses and related sewer system) are specific for the region, interbasin transfers of waste- and storm water are not rare and can play an important role at different scales in many peri-urban catchments. Impacts of these transfers in peri-urban catchments should therefore be analyzed and if necessary incorporated during model development. Bach et al. (2013) concluded, based on a semi-distributed model, that the representation of the flow processes in peri-urban models should take place at a high temporal and spatial resolution. Although FDPMs give a spatially detailed description of the hydrological processes, our results illustrate the challenges for sewer system integration in FDPMs, though they might be less if an open-source modeling system were used.

As expected, the different scenarios resulted in a decreased flow, compared to the reference scenario, proportional to the amount of EIA taken into account. The scenario with the lowest amount of EIA (scenario 2: transportation outside the catchment) resulted in the smallest change in flow. Besides an overall decrease in flow, both peak and low flows decreased. But low flows were proportionally more heavily affected by the rainfall reduction scenarios. These results confirm the higher impact of EIA during summer low flows found by the empirical analysis and again illustrate the high impact of EIA on the flow regime and its importance to be considered in the model. After consideration of the rainfall scenarios, the modeled rivers were higher than the measured river flow adjusted for the EIA, this despite the fact that the rainfall input was reduced for a similar amount within the different scenarios. Apparently, other processes like evapotranspiration within the model compensate for the reduced rainfall, leading to a lower reduction in flow.

Another problem is the coarse spatial resolution of the model. Due to this resolution, there is a general overestimation of both TIA and EIA implemented in the model compared to the high-resolution data. Nevertheless, similar values were obtained for the WWTP discharges and the reduction in flow in scenario 3 (full reduction based on all WWTPs). Because of the overestimation in the actual TIA and EIA, similar errors might be made to when no distinction is made between EIA and TIA (Alley et al., 1983). At the same time the overestimation of the impervious surfaces may have biased the hydrological model parameters during the calibration (e.g., underestimation of the surface runoff coefficient). This means that if the model were to be used for impact analysis of urbanization and climate change scenarios, the impacts on peak flows and flood frequencies may be underestimated. This problem is further investigated by Vansteenkiste et al. (2013) for the MIKE SHE model of the same catchment considered in this study.

An important aspect is the seasonal variation in the relative contribution of the EIA compared to the empirical data. Both scenario 1 (reduction for WWTPs inside the catchment) and 3 (full reduction based on all WWTPs) resulted in an underestimation of the EIA impact on the river flow during months with low flow and an overestimation during months with high flow. Hydrological models are often used to evaluate peak discharges and related flood risks. Climate change scenarios for Flanders, however, indicate an increase in frequency and duration of dry periods, making low-flow events more common (Boukhris et al., 2009). Therefore, the importance of these low-flow events and their evaluation in hydrological models will become more important. A better incorporation of both impervious areas and WWTPs might be crucial for a better performance of the models in evaluating the effect of climate change on peak and low-flow events. Hydrological models are frequently used to predict changes in the hydrological regime. But if we want to use these to assess changes in climate, land use or other future developments within the catchment, consideration of the sewer transfers discussed in this paper becomes increasingly important. Our results show that the further development of the sewer infrastructure will have a profound impact on the upstream areas.

5. CONCLUSIONS

This paper presented a methodology to calculate EIA in a way that incorporates the effects of WWCRs that do not coincide with natural catchment boundaries. The methodology allows us to evaluate storm- and wastewater transfers between different catchments and indicate how strongly the catchment’s hydrology is impacted by the sewer infrastructure. Comparisons between histograms or differences in histograms of catchment areas can display the vulnerability of the catchments to impervious area impacts and potential peak and low-flow events. The method also allows for study of how rivers that have no WWTP upstream are impacted by the upstream presence of EIA. These upstream impervious areas can have profound impacts on infiltration, surface runoff and the river flow regime. We also simulated the impacts of the changes in impervious areas and WWTPs in FDPMs. By applying different rainfall scenarios, the impacts of wastewater transfers in the catchment were simulated and evaluated. At the same time, we were able analyze the impervious area parameterization within the model. Our results show that water displacements in and between catchments may severely impact the hydrological model results. Hence it may also be important to take these displacements into account in the hydrological model development. Although we used high-resolution data, the limited integration of all sewer processes (e.g., SODs) in the analysis prevented us from completely assessing the impact of the sewer system on the model performance.

The correct incorporation of impervious areas in models is of utmost importance as impervious areas have an impact on catchment delineation and different aspects of the flow regime. With increasing urbanization and sewer development, the impact of these processes on the hydrological regime are expected to further increase in the future. Important areas of further research remain, amongst others, as to (a) how to incorporate impervious areas from outside the catchment into the model, (b) how to remove the areas that are transported outside the catchment from the model domain, (c) how to better represent the seasonal variation in impervious area and WWTP impact in the model, and (d) how to integrate these processes based on less detailed data sets.

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Chapter 3 - Water displacement by sewer infrastructure and its effect on the water quality in rivers.

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Abstract. Water quality is affected by a complex combination of natural and anthropogenic factors. To assess watershed integrity on a larger scale and for an optimal, cost-effective integrated watershed management, defining linkages between upstream watershed land cover and riverine water quality is essential. A correct upstream area calculation is an absolute necessity to reach conclusive results, but remains problematic in human influenced catchments. Especially sewer infrastructures (including wastewater treatment plants) are difficult to incorporate. We developed a method that allows us to integrate the sewer system in the upstream calculations and applied it on the Nete catchment in Belgium. Our results show strong changes in results compared to standard runoff methods. We conclude that if sewer systems are not incorporated in upstream area calculation, the impact of human activities on the water quality at a catchment scale estimates will be severely biased. A thorough understanding of the evaluated catchment and a correct translation of the different hydrological flow paths in the upstream area calculation is absolutely necessary to gain reliable results.

1. INTRODUCTION

Water quality is affected by a complex combination of natural and anthropogenic factors (Allan, 2004; Baker, 2003). Understanding the anthropogenic impacts is important to implement effective measures to improve water quality and stream ecosystem health (Booth et al., 2004). To assess watershed integrity on a larger scale, defining linkages between watershed land cover and river characteristics can provide interesting insights leading to cost-effective measure programs (Gergel et al., 2002; Oneill et al., 1997). Upstream landscape metrics are widely used as predictors of stream ecosystem health and water quality, biodiversity and river functioning (Jones et al., 2001; Stanfield et al., 2009; Van Hulle et al., 2010). The developed methods are frequently improved to better represent the different catchment processes (e.g. Baker et al., 2007; Sponseller et al., 2001; Strayer et al., 2003; Van Sickle, 2005; Van Sickle et al., 2008).

Upstream urban and impervious areas are important contributors to anthropogenic impact on both landscape and aquatic ecosystems (Arnold et al., 1996; Booth et al., 1997; Jacobson, 2011). Usually, urban and impervious land uses encompass only a low percentage of the catchment area. Still, they have a disproportionately large influence on both the hydrology and biogeochemistry of receiving streams (Cunningham et al., 2009; Feminella et al., 2005; Paul et al., 2001; Vrebos et al., 2014). Urbanization changes hydrological flow paths in many different aspects (Carey et al., 2013). An accurate integration of urban areas and their specific hydrological processes in upstream area calculations is hence an absolute requirement to study human influence on nutrient balances of catchments (Brabec et al., 2002).

Upstream area calculations that represent the actual catchment are dependent on correct incorporation of hydrological flow paths and solute deliveries (Gergel et al., 2002). Catchment areas are generally delineated based on computer rendered upstream areas that reflect the natural runoff conditions (Baker et al., 2006), while manmade structures are usually neglected. As a result, upstream areas are often inaccurately delineated (Hammond et al., 2006). One of the most important artificial structures is the sewer infrastructure: it drastically impacts the water flow paths in sub-urban catchments (Bernhardt et al., 2007). Rain and wastewater run through underground pipes, crossing streams above and beneath: water is pumped upstream, downstream and between sub-catchments. Areas that, under natural conditions, used to be part of one (sub)catchment have become part of another. Such manmade hydrological changes have profound effects on different explaining variables, like impervious area, used in land use indicator tests.

Instead of integrating sewer infrastructure in the upstream area calculation, most land use indicator studies incorporate sewer infrastructure and wastewater treatment plants (WWTP) as a separate factor with proper mean discharges and not as part of the upstream land use classes (e.g. Meynendonckx et al., 2006; Rothwell et al., 2010). A limited number of studies have considered connected or "effective" impervious areas (EIA) as a separate factor (e.g. Hatt et al., 2004; Wang et al., 2000). Still, the potential impact of the sewer system on upstream area calculation and land use distributions has been overlooked. The incorrect estimation of land use impacts can thus lead to erroneous conclusions. Yet, a correct calculation of upstream areas is an absolute necessity. Because of the large amount of explaining variables (e.g. upstream land uses, soil characteristics) used in statistical models, significant results are likely to be obtained, even if some explaining variables are incorrect.

In this paper, we assessed the effect of sewers and WWTPs on the upstream area calculation and the associated impact on land use indicator studies, in a case-study for the Nete catchment (Belgium). We compared the classic method of upstream area calculation with an adapted approach in which the sewer system complexity was incorporated. Both methods were used to explore correlations between upstream watershed characteristics and chloride and nitrogen concentrations over a time period of 8 years.

2. MATERIAL AND METHODS

2.1 Study area

The Nete catchment (approximately 1.673 km²) is situated in the central Campine region in Northern Belgium (Figure 3-1). It has a marine, temperate climate with an average precipitation of 800 mm/year. The dominant soil type is sand, with loamy sand occurring in the floodplains and a small area in the south that mainly consists of sandy loam and loamy sand. Topographic height ranges between 3 m and 82 m above sea level.

A major part of the streams within the catchment have been straightened, deepened and embanked. As a result, these streams no longer follow their natural flow path. At the same time large obstructions like canals and high ways have been constructed. These constructions are bypassed by siphons, which allow water to run under obstructions in the ground.

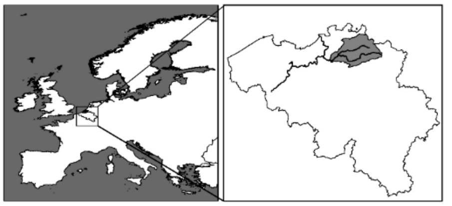

Figure 3-1: Location of the study area.

The Nete yearly discharges on average 389 million m³ water into the Rupel. Water chemistry is, spatially and temporally, highly variable because of different anthropogenic activities like agriculture, households, industry, etc. Land use in the Nete catchment consists mainly of cropland (20%), pasture (22%), broadleaved woodland and evergreen needle leaf forests (23%) and impervious area (8%). The other 27% mainly consists out of open water, gardens, bare land, etc. The land cover is highly fragmented with a median parcel of 0.10 ha.

In total 55 cities and municipalities are fully or partially located within the catchment. Population densities for these ranged in 2010 between 152 inh/km² and 1530 inh/km² with an average of 420 inh/km² for the entire catchment. The urbanized areas are mainly situated around the town centers. Typical ribbon development is present between the different villages, making the development of sewer systems costly and time consuming (De Decker, 2011).

At the moment 29 WWTPs are situated within the catchment. In 2008 74% of the households in the catchment were connected to a WWTP. Only during the 90's and the beginning of the 21st century most of these WWTPs have been expanded with tertiary treatment systems. Wastewater treatment zones do not coincide with the natural catchments and wastewater is actively transported from one (sub) catchment to another for treatment. Although the Nete catchment is considered to be one of the most natural catchments within Flanders (northern part of Belgium), almost none of the rivers and streams within the catchment meet the European Water Framework Directive standards and more investments to reach these standards are needed.

2.2 Water quality data

Water quality data from 2003 until 2010 were obtained from the Flemish Environmental Agency (FEA). During this period, nitrate (NO_3^--N, mg/l), nitrite (NO_2^--N, mg/l), ammonium (NH_4^+-N, mg/l) and chloride (Cl^-, mg/l) concentrations in streamflow were measured at 345 different locations in the catchment. Sampling frequency differs between the sample locations. Some points were sampled monthly for all parameters, others only once a year for one parameter. For nitrate 16762 samples were available for analysis, for nitrite 14836 samples, ammonium 14646 samples and for chloride 13877 samples.

Seasonal means, i.e. winter (December-February), spring (March-May), summer (June-August) and autumn (September-November) were calculated per year for each sample point, only when at least 2 samples were present for each season. As a consequence, incorporated data points vary for the different variables and between years. Because of the origin of the data, detection limits varied between years and location and some measurements were below a relatively high detection limit. In order to take these measurements into account and incorporate also the sample points with low concentrations in the analysis, measurements below the detection limit, were given a value half of the detection limit.

2.3 Wastewater treatment plants

There are 29 active WWTPs situated in the catchment. The oldest dates back to 1957, the most recent was built in 2007. While most of the WWTPs have been renovated in last 15 years, large parts of the sewer infrastructure are relatively old and most of the sewer system still collects waste as well as rainwater, but also parasitic groundwater (Dirckx et al., 2009). Yearly WWTPs influent and effluent data were obtained from the FEA for the period 2003-2010. For one, small, WWTP (Leopoldsburg) no data were available. These influent and effluent data encompass the following information: yearly flow (m³/year), NO_3^--N load (kg/year), NO_2^--N load (kg/year), NH_4^+-N load (kg/year) and chloride load (Cl^- kg/year).

2.4 Geographic analysis

2.4.1 *Land use map*

Land use maps (1:10.000 vector-layers) were obtained from the National Geographic Institute (NGI) and consist of 49 different categories (NGI, 2007). They have a high accuracy and are based on aerial photographs from 1998 (1:21.000) and on site verification and adjustment in the following years until 2007. The land use vector layer was converted to a 1m-raster and the land use categories were aggregated to 8 different classes: woodland (VE111, VE113, VE114, VE120, VE131, VE132, VE133, VE140, VE150, VE220), cropland (VE340), pasture (VE320), buildings (EL221, ST111, ST112, ST113, ST120, ST211, ST220, ST230, ST240, ST250, ST311, ST411, ST414, ST416, ST720, ST911, ST912, ST914), paved area (RA112, RO112, RO113), water (HY112, HY120, HY131), greenhouses (ST131) and others (GS110, GS300, ST931, VE211, VE212, VE213, VE214, VE231, VE232, VE240, VE310, VE330). A description of the different land use categories can be found in the Appendices (Table S3.1). A distinction was made between buildings (area buildings) and other impervious areas (roads, concrete areas, etc.) because buildings can be an important source of wastewater, while other impervious areas mainly collect rainwater. All GIS-calculations were performed in ArcGIS 9.3.

2.4.2 *Soil map*

Soil properties were obtained from the digital soil map (1:20.000) created by the Flemish Land Agency (AGIV, 2006). 1m-rasters for soil texture and soil drainage were calculated. Soil texture characteristics were aggregated to 8 classes: sand (Z), dunes (X), loamy sand (S), sandy loam (P and L), loam (symbol A), clay (symbol E and U) and peat (symbol V). Soil drainage characteristics were aggregated to five classes: well drained (symbol a and b), moderately drained (symbol c and d), poorly drained (symbol e, f, and g) and poorly drained with stagnating water (symbol h and i) and others (areas without symbol).

2.5 Upstream area calculations

Two different methods were used to calculate the upstream areas. The ‘runoff method’ is the commonly used method based on a runoff model. The ‘sewer method’ is an adaption of this, incorporating the sewer system.

2.5.1 *Runoff method*

First, sub-catchments were delineated for each FEA sample point from a 1:5000 digital elevation model (DEM) expressed as a 5m-raster (FEA, 2006) using a D8-runoff model (Jenson and Domingue, 1988). The DEM was modified by lowering the elevation values based on mapped stream channels and siphons and by elevating values based on mapped dikes (NGI, 2007). This allowed us to force flow-direction maps to match existing streams. Sub-catchments were calculated using the hydrology tools in ArcGIS 9.3 (Figure 3-2). Areas that drain into the artificial water navigation canal system were removed from the analysis (Figure 3-2).

The 5m-subcatchment raster was resampled to a 1m-raster for the further analysis. With this list of sub-catchments, upstream areas were calculated for the different sample points (“runoff method”). By combining the 1m-raster with the land-use, soil texture and soil drainage raster, datasets were compiled that allowed the calculation of upstream land-use, soil texture and soil drainage class percentages for each FEA sample point.

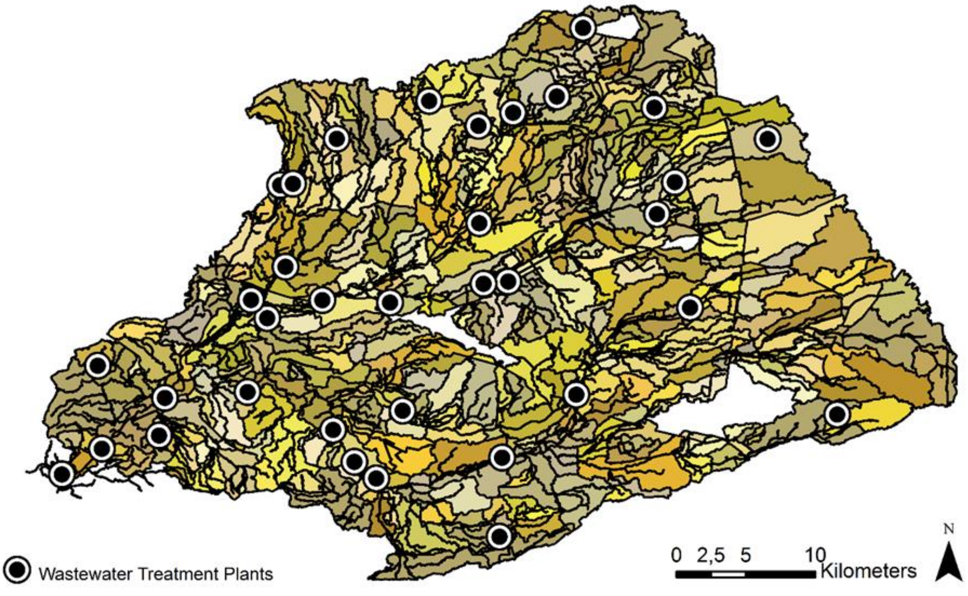

Figure 3-2: Different subcatchments within the Nete catchment based on the FEA sample points and location of the WWTPs. This network is used to calculate the upstream areas. The white zones within the catchment drain into canals that run through the catchment.

2.5.2 *Sewer method*

In a second step upstream land use areas were recalculated including the sewer infrastructure. In 2008, the FEA developed maps that indicate which buildings currently are connected to a WWTP, which will be connected in the future and which buildings will never be connected and will be responsible for their own wastewater treatment. These maps were used to assign each building from the land use maps to a WWTP. The methodology was previously used to assess the importance of sewer systems in hydrological modelling (Vrebos et al., 2014).

Buildings connected to a WWTP effectively drain into the river that receives the WWTP discharge. These surfaces were virtually removed from their sub-catchment and added to the WWTP receiving sub-catchment. For each sample point total upstream areas and subsequent land use percentages were recalculated.

To recalculate the soil properties consequently we assumed the infiltration capacity of impervious areas to be zero. Therefore, all building areas, connected and not connected, were removed from the 1m-rasters. Both upstream soil texture and soil drainage percentages were calculated.

2.6 Data analysis

Relative changes (%) were calculated for upstream areas and upstream building areas between the “runoff method” and “sewer method. Normality of distribution of all explaining variables was tested with the Kolmogorov-Smirnov goodness of fit test. When necessary the dependent water quality variables were log-transformed. Spearman rank correlations were calculated between the connected building areas and yearly means of both influent and effluent loads (n=29) to assess the predictive power of “connected buildings” for WWTP impact (Hollander et al., 1973).

To examine the effect of different catchment properties on water quality both multiple regression and partial correlation statistics are often used in land use – water quality studies (e.g.

Daniel et al., 2010; King et al., 2005). Partial correlations are less sensitive to outliers and can be used with non-normal distributed data. Therefore, partial correlations are a better solution for water quality assessments which are often characterized by outliers and non-normal distributed datasets (Van Sickle, 2003).

To calculate the partial correlation, several variable classes were removed from the dataset (other land use, other soil texture, other soil drainage, open water and loam). This is necessary as spearman rank partial correlations do not work correctly when the total of the explaining variables is always the same, 100% in this case. Spearman rank partial correlations were calculated between seasonal mean water quality data and the different upstream catchment variables (Johnson et al., 2007). In this study goodness of fit tests and spearman rank correlations were performed with the "stats" library of functions and spearman rank partial correlations with the "ppcor" library of functions in R (R Core Team, 2008).

3. RESULTS

3.1 Water quality data

Water quality concentrations in the Nete catchment can differ strongly between location, season and year (Figure 3-3). The different variables display different seasonal patterns. Ammonium and nitrate have lower median concentrations during summer and autumn (Figure 3-3a and 3c), while nitrite generally has its lowest median values during winter period (Figure 3-3b).

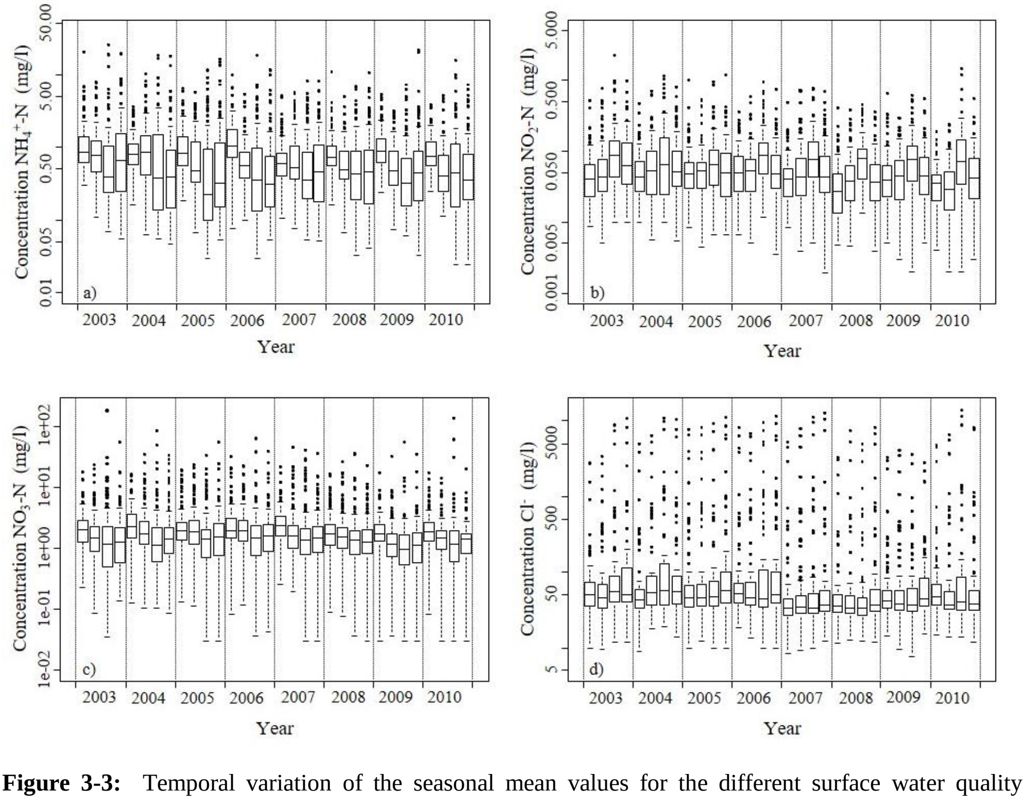

Figure 3-3: Temporal variation of the seasonal mean values for the different surface water quality parameters on a log scale NH_4^+-N (a, n = 3592), NO_2^--N (b, n = 3628), NO_3^--N (c, n = 4024) and Cl^- (d, n = 3392). For each season the boxplot represents the median, 25th and 75th quantile and standard deviations of the seasonal means. Whiskers on the boxes give the largest values within 1.5 times interquartile range above 75th percentile range and the smallest value within 1.5 times interquartile range below the 75th percentile range.

3.2 WWTP processes and connected areas

Between 2003 and 2010 the 29 active WWTPs in the Nete catchment discharged on average 55 million m³ per year of wastewater or 14.1% of the total yearly discharge of the Nete river. The monthly WWTP discharge (Q) is positively correlated with the connected building areas (Table 3-1).

The WWTPs reduced the total NH_4^+-N loads by 91%, from 1.340.464 kg/year (influent) to 118.398 kg/year (effluent). At the same time mean total NO_3^--N loads increased by 814% from 43.766 kg/year (influent) to 356.277 kg/year (effluent). The impact of WWTPs processes on NO_2^- loads is less pronounced. Loads decreased by 25%, from 12.375 kg (influent) to 9.321 kg (effluent).

Spearman rank correlations between NH_4^+-N loads and connected building areas (m²) decreased between influent and effluent as a result of differences in WWTP processing methods and processing efficiency. Correlations between NO_3^--N loads and connected building areas (m²) increased (Table 3-1). Correlations between NO_2^--N and connected building areas (m²) remained stable. Based on these significant results, we assumed that connected building area is a good indicator for the impact of WWTPs on the surface water quality.

Table 3-1: Spearman rank correlations between connected building area (m²) and WWTP monthly influent and effluent data between 2003 and 2010 are given for discharge Q (m³), NH_4^+-N (kg), NO_3^--N (kg), NO_2^--N (kg), NO_3^--N (kg) and Cl^- (kg).

n = 2784	Influent		Effluent	
	Rho	p-value	Rho	p-value
Q			0.95	<0.001
NH_4^+-N	0.94	<0.001	0.7	<0.001
NO_2^--N	0.72	<0.001	0.6	<0.001
NO_3^--N	0.54	<0.001	0.77	<0.001
Cl^-	0.9	<0.001	0.91	<0.001

3.3 Impact of sewer system on the upstream area calculation

As a result of the "sewer method" calculation upstream building area and change in total upstream area changed substantially. A reduction in the upstream area for 281 of the 345 sample points was displayed. Conversely 64 sample points displayed an increase in upstream area and 33 sample points experienced no change compared to the "runoff method". The change in total upstream building area ranged between a decrease of 98% and an increase of 525% (Figure 3-4a). This results in a maximum decrease of 9% and a maximum increase of 24% of the total upstream area (Figure 3-4b).

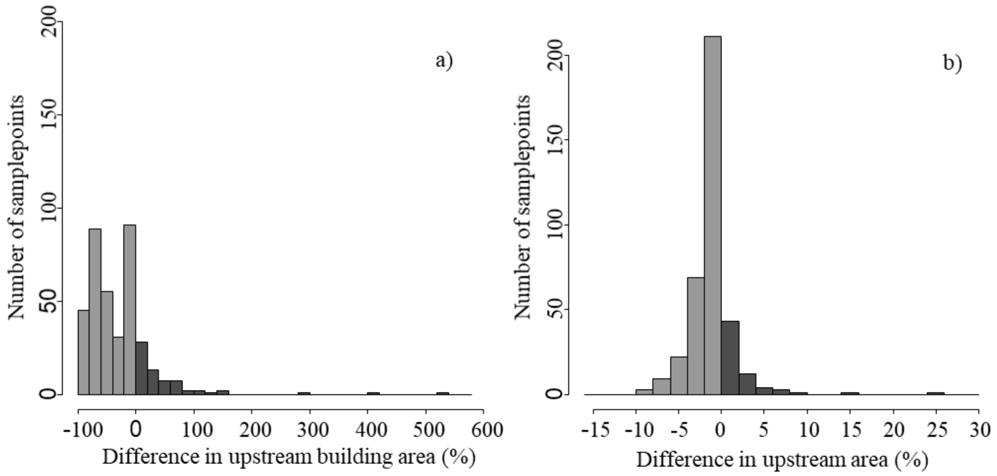

Figure 3-4: Histograms display differences in area between the "runoff method" and "sewer method": (a) difference in upstream building area, (b) difference in total upstream area (n = 345).

3.4 Correlations between upstream area and water quality parameters

3.4.1 *"Runoff method" versus "Sewer method"*

We have focused our results on the impact of connected and not-connected building area on different water quality parameters and its relevance to the upstream area calculation. The full analysis result can be found in addendum A of Vrebos et al. (2014).

To evaluate both methods of upstream area calculations partial correlation coefficients between upstream land use areas and water quality parameters were calculated (Table 3-2). Only the results for mean values for 2007 for the parameters NH_4^+-N, NO_3^--N, NO_2^--N and Cl^- are shown. Similar results were found for the other years.

Using the "runoff method", no significant results could be found between the different water quality variables and the upstream building. However, when the "sewer method" was used and a distinction was made between connected and not-connected buildings, significant results were found between connected building areas and Cl^-, NO_3^--N and NO_2^--N. NO_2^--N also showed a significant correlation with the unconnected houses during summer and winter. For NH_4-N, a significant correlation with not connected buildings was only found in winter.

While no significant results were found between the different water quality parameters and buildings from the "runoff method", other partial correlations with land use and soil properties were found (Table 3-3). Many of the significant results found with the "runoff method" decreased using the "sewer method" or disappeared.

Table 3-2: Spearman rank partial correlations between different water quality parameters (NH_4^+-N, NO_3^--N, NO_2^--N and Cl^-) and upstream buildings calculated with the "runoff method" and "sewer method". Mean seasonal values were calculated for the year 2007. (* = p-value < 0.05, ** = p-value < 0.01, *** = p-value < 0.001)

n = 133	Season	Parameter	Runoff method		Sewer method	
NH_4-N	Winter	Building / Connected	---	---	0.39	***
		Not-connected			0.22	***
	Spring	Building / Connected	---	---	---	---
		Not-connected			---	---
	Summer	Building / Connected	---	---	---	---
		Not-connected			---	---
	Autumn	Building / Connected	---	---	---	---
		Not-connected			---	---
n = 143						
NO_2^--N	Winter	Building / Connected	---	---	0.42	***
		Not-connected			---	---
	Spring	Building / Connected	---	---	0.44	***
		Not-connected			---	---
	Summer	Building / Connected	---	---	0.32	**
		Not-connected			0.23	*
	Autumn	Building / Connected	---	---	0.42	***
		Not-connected			0.2	*
n = 133						
NO_3^--N	Winter	Building / Connected	---	---	0.21	*
		Not-connected			---	---
	Spring	Building / Connected	---	---	0.41	***
		Not-connected			---	---
	Summer	Building / Connected	---	---	0.42	***
		Not-connected			---	---
	Autumn	Building / Connected	---	---	0.42	***
		Not-connected			---	---
n = 127						
Cl^-	Winter	Building / Connected	---	---	0.25	*
		Not-connected			0.21	*
	Spring	Building / Connected	---	---	0.38	***
		Not-connected			---	---
	Summer	Building / Connected	---	---	0.32	**
		Not-connected			---	---
	Autumn	Building / Connected	-0.21	*	0.38	***
		Not-connected			---	---

Table 3-3: Spearman rank partial correlations between different water quality parameters (NH_4^+-N, NO_3^--N, NO_2^--N and Cl^-) and selected upstream characteristics calculated with the "runoff method"". Mean seasonal values were calculated for the year 2007. (* = p-value < 0.05, ** = p-value < 0.01, *** = p-value < 0.001).

Samples	Variable	Winter		Spring		Summer		Autumn	
		Rho	p-value	Rho	p-value	Rho	p-value	Rho	p-value
	NH_4^+-N								
n = 133	Well drained	0.18	*	0.32	**	---	---	0.19	*
	NO_3^--N								
n = 143	Green -houses	0.38	***	0.25	*	0.29	**	0.22	*
	Loamy sand	-0.24	*	-0.23	*	-0.23	*	---	---
	NO_2^--N $^-$								
n = 133	Green -houses	0.26	*	0.23	*	0.33	**	0.31	**
	Clay	---	---	0.22	*	0.22	*	0.19	*
	Well drained	0.22	*	0.23	*	0.25	*	---	---
	Cl^-								
n = 127	Clay	0.20	*	0.20	*	0.30	**	0.28	*
	Well drained	0.28	*	0.28	*	0.31	**	0.32	**

3.4.2 *Differences in seasons and years*

Strong seasonal trends were found between different land use and water quality parameters. When significant results were found throughout the different years rho and p-value varied between seasons and displayed a clear pattern. Partial correlations between WWTP connected buildings and NO_3-N and NO_2-N showed strong seasonal trends. Partial correlations for NO_2-N peaked during winter period and spring with rho values up to 0.52 (p-value < 0.001) and lowest rho in summer and autumn down to 0.23 (p-value < 0.05) (Figure 3-5a). Rho values for NO_3-N were at their highest during spring and summer with a maximum rho of 0.54 (p-value < 0.001) and lowest during winter with a lowest rho of 0.19 (p-value < 0.05) (Figure 3-5b). Similar seasonal trends were not found for $NH4^+$-N and Cl^-.

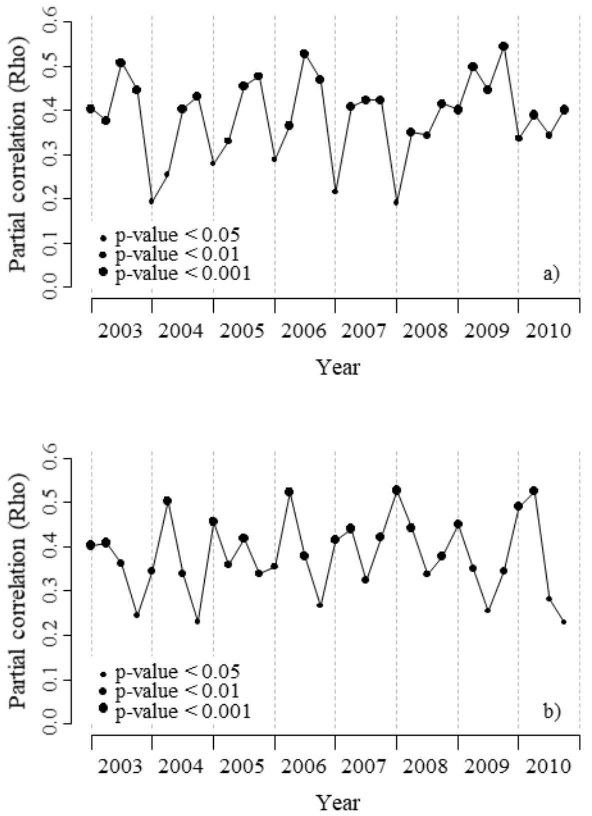

Figure 3-5: Seasonal variation in spearman rank partial correlations between WWTP connected buildings and yearly seasonal mean values of NO_2^--N (a) and NO_3^--N (b).

4. DISCUSSION

Our results clearly demonstrate the importance of a correct incorporation of sewer hydrological flow paths in the upstream area calculation. Results obtained from the "runoff method" initially confirmed the findings in the Nete catchment of Meynendonckx et al. (2006). Buildings were concluded to have no general impact on nutrient concentrations in the catchment and the defining upstream variable for nitrate concentrations was selected from the geophysical explaining variables. The use of these conclusions in management decisions could result in inaction, because of the underestimation of anthropogenic impacts.

The integration of the sewer infrastructure in the upstream area calculations changed both upstream area composition and statistical outcomes completely. It can now be concluded that building areas have a determining impact on the water quality. NH_4-N, NO_3-N and NO_2-N are affected differently and the impact can change between seasons and years. Contrarily to the "run-off method" these results provide information to correctly focus management on the anthropogenic sources like sewer development and small individual treatment plants.

Previous studies proved the significance of a good delineation of the catchment (Baker et al., 2006). However artificial hydrological flow paths like sewer systems are difficult to incorporate in an automated process. Therefore, methods developed to characterize rural watersheds based on DEMs should not automatically be applied directly to urban terrain, and consequently different methodologies have been developed for urban areas in recent years (Gironas et al., 2010). However, more and more areas have a mixed land use in which agriculture, urban areas, sewer infrastructure, direct discharges of households and other activities can all have an impact on the river health. Analyzing the possible impact of this mixed land use requires a correct incorporation of both natural and artificial flow paths in the upstream land use calculation. Evaluating the different flow paths and adapting the upstream areas is a labor intensive and time consuming process. However, as demonstrated by our outcomes, the alternative, a fast and less detailed analysis, can lead to misleading results and bad, counterproductive conclusions and actions.

Complementary with the upstream area calculation is the issue of the "impervious area" classification. In landscape indicator studies urban land uses are generally aggregated to one class, but impervious areas are often heterogeneous and generally consist of roads, houses (or roofs), industry, gardens etc… showing it is far from a uniform land use class. Depending on the research question, e.g. impact of urban area on water flow, nutrients, metals, etc. a subdivision of the class "impervious area" is required to assure a good representation of the classes (Alley et al., 1983). In this study an initial subdivision of the impervious areas in 'paved areas' and 'buildings' did not generate reliable results. A further separation in connected and not-connected buildings and the incorporation of the sewer system in the area calculation however changed the results completely. The subdivision of impervious areas and aggregation of the data on a sample point level should be handled with care and combined with the related hydrological flow paths.

The strong change in significant results that can be seen between both methods can be attributed to the effect of adding one extra class in combination with the recalculation of the upstream area. Instead of just adding one class, e.g. WWTP discharges, our method not only changed the building area distribution but in fact changed the distribution of all the explaining variables. Increasing or adding one class in absolute values will result in an increase in percentages, but also in a decrease of all the other land use variable percentages. This results in complete changes in significance from one class to the others.

Over time, progressive land use change and other developments can have an impact on both the different flow paths and land use distributions. Outdated land use maps or older versions of the river system should therefore be handled with care. The datasets that are incorporated in the study have to be relevant and reliable for the investigated time period. In our dataset, the 8-year study strongly confirms the significant link between buildings (connected and unconnected) and N and Cl concentrations in the receiving rivers.

The relevance for implementation of the presented methodology depends on both the available data as well as the characteristics of the catchment and sewer system. Making a distinction between households and paved areas is probably most relevant in urban and sub-urban studies where sewer systems can have a large impact on both flow pathways as well as water quality.

Results from land use indicator studies give a good understanding of the overall anthropogenic influences in the catchment. However, the statistical results should always be handled with care. Absence of significant correlations with a certain land use does not mean that it has no impact on the water quality variable. The impact of diffuse pollution sources like agriculture for example appeared to be difficult to assess within this study. Although many studies found significant results between agriculture and nitrogen (e.g. Dodds et al., 2008; Jordan et al., 1997; Lassaletta et al., 2010), this is not the case in the Nete catchment. This is unexpected, especially because agriculture is considered to be the main source of nitrogen. Agriculture is assumed to have a large impact on the water quality in the region and a part of the water quality network used in this study is specifically designed to monitor the effects of agriculture on the water quality. An explanation could be that the applied land use indicators are inadequate because of a combination of land conversion, crop rotation and response delay (Cherry et al., 2008). Differences in agricultural practices between individual farmers can also have led to a spatially differentiated impact of agriculture on the nitrogen compounds.

Just like for urban areas and WWTP plants, additional information might lead to better correlative results between diffuse sources and different water quality parameters. For example, creating different agricultural land use variables based on fertilizer uses, instead of a straightforward distinction between grassland and cropland, might lead to more significant results. However, a more advanced approach would require the necessary data to be available on parcel level.

Another important remark is the translation of rho values into impact analysis. Higher rho values of a result do not necessarily mean that concentrations are higher during the period with the highest rho-value, but only indicate that the spatial relationship between both variables is stronger during that period and the impact of the variable is more consistent over the catchment. Nitrite concentrations for example were generally higher in summer then in winter. Yet the partial correlations with connected buildings were highest in winter. Therefore, the translation of the results into conclusions should always be combined with a thorough analysis of the water quality dataset.

5. CONCLUSION

Land use – water quality tests can give us valuable information on the driving forces within catchments for important factors in stream ecosystem health, such as N concentrations. But in order to get reliable results a good representation of the hydrological cycle and associated land-use in the upstream area calculation is particularly important. A good understanding of the evaluated landscape or catchment and a correct translation of the different hydrological flow paths in the upstream area calculation is absolutely necessary to gain reliable results. The presented methodology is especially of importance in urban and sub-urban studies where sewer systems can have a large impact on both flow pathways as well as water quality.

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Chapter 4 - The impact of land use and spatial mediated processes on the water quality in a river system.

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Abstract. River systems are highly complex, hierarchical and patchy systems which are greatly influenced by both catchment surroundings and in-stream processes. Natural and anthropogenic land uses and processes effect water quality (WQ) through different pathways and scales. Understanding under which conditions these different river and catchment properties become dominant towards water chemistry remains a challenge. In this study we analyzed the impact of land use and spatial scales on a range of WQ variables within the Kleine Nete catchment in Belgium. Multivariate statistics and spatial descriptors (Moran's and asymmetric eigenvector maps) were used to assess changes in water chemistry throughout the catchment. Both land use and complex mixes of spatial descriptors of different scales were found to be significantly associated to WQ parameters. However, unidirectional, upstream-downstream changes in water chemistry, often described in river systems, were not found within the Kleine Nete catchment. As different sources and processes obscure and interact with each other, it is generally difficult to understand the correct impact of different pollution sources and the predominant pathways. Our results advocate for WQ management interventions on large and small scales where needed, taking the predominant pathways in to account.

1. INTRODUCTION

River systems are complex and patchy systems which are greatly influenced by both catchment surroundings and in-stream processes (Poole, 2002). This results in a hierarchical system, ranging from the largest spatial scale of landscape or basin to successively smaller scales such as the valley segment, channel reach and sediment pools and riffles (Allan, 2004; Townsend et al., 2003). Different geomorphological, ecological and anthropogenic factors affect river water quality (WQ), with changing influence over temporal and spatial scales (Baker, 2003; Poole, 2010). Under natural conditions, river systems already demonstrate a high level of complexity which has increased further due to land development. Anthropogenic activities have disrupted and changed existing processes and/or included new water and pollution sources through different pathways (sewers, runoff, seepage, etc.). As a result, carbon, nutrient and other contaminants, such as chloride and calcium, have become more dynamic (Kaushal et al., 2012; Kaushal et al., 2014; Steele et al., 2011).

Since the beginning of the 20th century, scientists have attempted to translate these dynamics into theoretical concepts (Melles et al., 2012). Vannote et al. (1980) observed that physical variables changed along a continuous gradient from headwaters to river mouth, determining stream communities (River Continuum Concept: RCC). In the following decades, additional concepts were developed that increased our understanding of the rivers complexity, amongst others: the serial discontinuity concept (Wards et al., 1983) , nutrient spiraling (Webster et al., 1979), catchment hierarchy (Townsend, 1996) and patch dynamics (Townsend, 1989). Which were later integrated and extended by Poole (2002), Thorp et al. (2006) and Humphries et al. (2014) to explain different types of discontinuities in natural systems. Statzner et al. (1986) evidenced the universal influence of hydraulics on the longitudinal river gradient, and mentioned that natural zonation patterns over long stream reaches are usually obscured by human influences. The integration of anthropogenic influences in these concepts is still progressing; e.g. “Urban Stream Syndrome” by Walsh et al. (2005) and “Urban Watershed Continuum” by Kaushal et al. (2012). Under what conditions these concepts can be applied is often less clear.

Many river processes occur at different spatial scales, taking directional movements into account. Both natural and anthropogenic patterns and processes in river systems are generally strongly oriented to downstream reaches (longitudinal connectivity vectors) (Ward, 1989). But also lateral (e.g. runoff) and vertical (e.g. seepage) vectors can have important effects (Stanford et al., 1993; Townsend, 1996). The relative magnitude of these vectors can differ among river systems and investigating which of these vectors are predominant within a river system remains challenging.

Catchment characteristics like land uses and geomorphological properties are connected to a river system through all three vectors and have profound effects on the river characteristics. These characteristics are therefore widely used as landscape metrics to explain WQ variation and applied as predictors for ecosystem health and river functioning (Jones et al., 2001; Stanfield et al., 2009). As these landscape metrics depend on a good delineation of the upstream area and correct incorporation of the hydrological flow paths and solute deliveries (Gergel et al., 2002), much research has gone to improve their calculation Baker et al., 2007; Van Sickle et al., 2008; Vrebos et al., 2015). Nevertheless, landscape metrics can only explain part of the observed variation, as they describe only a limited portion of all processes present in a catchment. For example, in-stream processes can have a profound impact on different WQ parameters (e.g. Caissie, 2006; Withers et al., 2008) and also groundwater contributions to stream chemistry can be scale dependent (Peralta-Tapia et al., 2015). Up to now, it remains difficult to define and select structures, connectivity vectors and scales on which these processes take place and to translate them into functional indicator metrics.

In general, ecological variables strongly depend on environmental conditions which change often gradually in space (Dray et al., 2012; Legendre, 1993). Although a catchment can extend over a large area and can be marked by strong upstream-downstream gradients, sub-gradients at smaller scales can result from sub-catchment properties, local pollution sources or in-stream processes. As a result, the WQ pattern of the whole catchment might encompass various spatial scales. In this respect, WQ descriptors can exhibit different “spatial waves”, where these descriptors increase and decrease along a river stretch. These changes depend on particular physical, chemical or human-mediated drivers and/or local independent processes. Basically, there is no single scale at which ecological phenomena should be studied as the observer imposes a perceptual bias (i.e. the spatial extent of his work) through which the system is viewed (Levin, 1992). Many scales can be perceived in any spatial extent and recent developments in spatial analysis enlarged exploratory perspectives in scale-dependent processes (Dray et al., 2012). Scales can be materialized by spatial variables derived from geographic coordinates of sampling units in order to explore the spatial nature of ecological variables by a correlative approach. Two types of spatial predictors were recently developed: those accounting for spatial patterns resulting from multi-directional processes (Griffith et al., 2006) and those accounting for unidirectional ones (F. Guillaume Blanchet et al., 2008). Both types can be combined within one analysis, to test a community or species for both directional and non-directional processes and flows at different spatial scales (Blanchet et al., 2011).

Natural lotic systems are expected to generate longitudinal gradients. However, intensive land uses should strongly disrupt the spatial extent of natural processes and these longitudinal gradients. Focusing on a strongly anthropogenically affected river system in northern Belgium, we hypothesized that (1) land use should drive a substantial amount of WQ variance; (2) spatial variables accounting for unidirectional processes should weakly predict WQ patterns; and (3) human-mediated processes should take place over complex mixes of spatial scales.

2. MATERIAL AND METHODS

2.1 Study Area

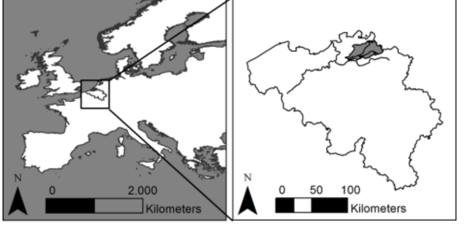

Figure 4-1: Location of the study area.

The Kleine Nete Catchment (approximately 780 km²) is situated in in Northern Belgium and is a sub-catchment of the Scheldt river basin (Figure 4-1). It has a marine temperate climate with an average precipitation of 800 mm/year. Topographic heights range between 3m and 57m above sea level. As a result of weak elevation gradients, average water current velocities are limited (0.2 m/s in winter, 0.06 m/s in summer) (De Doncker et al., 2009). The catchment consists mainly of sandy soils, with loamy sand soils in the floodplains. Due to permeable soils, the river system is mainly groundwater fed and natural surface runoff takes only place during wet periods when the soils are water saturated. Some of the upstream geophysical characteristics of the catchment, used in the analysis, are summarized in the supplementary material (Figure S4.1).

Land use within the catchment is dominated by pasture (24%), croplands (20%), broadleaved woodland and evergreen needle leaf forests (22%) and houses and roads (13%). The other 21% consists mainly out of open waters, gardens, shrubs, etc. In total 27 cities and municipalities are fully or partially located within the catchment, with a total population of +/- 524 000 inhabitants in 2012 and an average population density of 455 inh./km². Typical ribbon development is present between villages in the catchment (De Decker, 2011).

The hydrological system within the catchment has been strongly transformed by human interventions. Streams no longer follow their natural flow paths and their flow regimes are heavily altered. Although the Kleine Nete is considered to be one of the lesser polluted catchments in Flanders, almost none of the streams within the catchment meet the European Water Framework Directive (Directive 2000/60/EC) quality standards. At the moment 20 wastewater treatment plants (WWTP) are located within the catchment. In 2012 75% of the households in the region were connected to a WWTP. The rest of the population discharges its wastewater directly into the river system through ditches and sewer pipes.

2.2 Water quality sampling

Between June 2010 and the end of 2012, monthly samples were taken on 73 locations within the Kleine Nete catchment (Figure S4.2). Fifty-seven of those sample points are part of the long term monitoring network of the Flemish government. An additional 16 locations were selected to better represent the different land uses and improve the spatial coverage within the catchment. Because of the number of sampling points, monthly sampling took 3 days.

Samples were analyzed to quantify 25 different parameters: temperature (T; °C), pH, oxygen concentration (O_2Conc; mg/L) and saturation (O_2Sat %) were measured on location. Concentrations (mg/L) of nitrate (NO_3), nitrite (NO_2), ammonium (NH_4), Kjeldahl nitrogen (KjN), orthophosphate (oPO_4), total phosphorus (Ptot), chloride (Cl^-), carbon dioxide (CO_2) biogenic silica (BSi), calcium (Ca), iron (Fe), potassium (K), magnesium (Mg), sodium (Na), dissolved silica (SiO2), zinc (Zn), biological oxygen demand (BOD), chemical oxygen demand (COD), suspended solids (SS), chlorophyll a concentrations (Chla; mg/L) and conductivity (Cond; µS/cm) were measured in the lab following different national and international standards. An overview of these is given in the supplementary materials.

2.3 Upstream area analysis

For each of the sampling points the characteristics of the upstream area were calculated following the procedure of Vrebos et al. (2015), taking the sewer system into account where needed. All GIS-calculations were performed in ArcGIS 9.3 (ESRI Inc., 2009).

2.3.1 *Land use and soil map*

Land use maps (1:10.000 vector-layers) were obtained from the National Geographic Institute and consist of 49 categories (NGI, 2007). These, high accuracy maps (1:21.000) were created between 1998 and 2007. The vector layers were converted to a 1m-raster and the land use categories were aggregated to 8 different classes. woodland, cropland, pasture, buildings, paved area, water, greenhouses and others. A description of the land use categories can be found in Chapter 3 – Addendum A. A distinction was made between buildings and other impervious areas (roads, concrete areas, etc.) buildings can be an important source of wastewater, while other impervious areas mainly collect rainwater.

2.3.2 *Soil map*

Soil properties were obtained from the digital soil map (1:20.000) of Flanders (AGIV, 2006). 1m-rasters for soil texture and soil drainage were calculated. Soil texture characteristics were aggregated to 8 classes: sand (Z), dunes (X), loamy sand (S), sandy loam (P and L), loam (symbol A), clay (symbol E and U) and peat (symbol V). Soil drainage characteristics were aggregated in five classes: well drained (symbol a and b), moderately drained (symbol c and d), poorly drained (symbol e, f, and g) and poorly drained with stagnating water (symbol h and i) and others (areas without symbol).

2.3.3 *Upstream area calculation*

Upstream areas and sub-catchments were delineated for each sample point from a 1:5000 digital elevation model expressed as a 5m-raster (FEA, 2006) using a D8-runoff model (Jenson and Domingue, 1988). The sub-catchment raster was used to calculate initial upstream land use, soil texture and soil drainage acreages. These datasets were then adapted to include effects of sewer infrastructure. Maps, that indicate which buildings are currently connected to a WWTP, were used to virtually remove them from their sub-catchment and add them to the connected WWTP receiving sub-catchment. As result a distinction between WWTP connected and not connected buildings could be made. For each sample point total upstream areas and subsequent land use percentages were calculated. A more detailed description of the used methodology can be found in Chapter .

Since the same area is taken into account for several WQ sample points, there is double counting in the land use conditions. This can lead to some autocorrelation in the results. Using independent sample points on different streams, often used in land use – WQ studies, was not an option. As this would not generate adequate MEMs and AEMs. To our knowledge, there are currently no statistical methods that are able to adequately address this issue of spatial autocorrelation in lotic systems.

2.4 Data Analysis

Initially WQ was statistically compared (Tukey's HSD) between seasons (winter = December, January, February; spring = March, April, May; summer = June, July, August and autumn = September, October, November). WQ parameters were found to be similar in spring, summer and autumn, but differed drastically in winter (Figure S4.3). Winter concentrations were mainly distinct due to higher chemical loads, while some parameters, such as pH, were lower. At the same time, river discharges strongly varied from winter (high) to summer (low) (Figure S4.4). Hence, data analyses were conducted separately on winter data and on averaged values of the three other seasons; hereafter called "summer". Many WQ parameters have a diurnal variability (Vandenberghe et al., 2005). For most of our samples, the sampling time was not available, although sampling always took place during the day. As a consequence, we were not able to adjust for diurnal variability. However, we computed mean values per season from monthly values spread over two years. Between-month and between-year variabilities may have affected samples more than sampling time within the few sampling days. An overview of the ranges of the different WQ variables in both periods is given in supplementary materials (Figure S4.5). Both periods were further analysed in the same way. Three data tables; land uses, water qualities and spatial predictors, were further considered and processed in two steps.

Firstly, significant relationships between land uses and water quality parameters were drawn from a Principal Component Analysis on Instrumental Variables (PCAIV; Lebreton et al., 1991; Sabatier et al., 1989) also known as Redundancy Analysis (RDA). This enabled us to consider the WQ variations due to land use related processes. The significance of these relationships was tested with a permutation test (9999 iterations; Manly, 1991).

Secondly, spatial scales of the resulting predicted pattern were determined using recent statistical developments (F. Guillaume Blanchet et al., 2008; Griffith et al., 2006). Spatial scales are not controlled by the observer, but deduced from the spatial extent of the dataset. Geographic coordinates of n sampling stations are used to build a weighted distance matrix (n stations × n stations) which is diagonalized to generate n – 1 orthogonal eigenvectors and n – 1 orthogonal associated spatial predictors composed of n observations (principal coordinates). In a conservative way, this enabled us to consider the WQ variations only due to land use related processes. The significance of these relationships were tested using a permutation test (9999 iterations; Griffith et al., 2006). This procedure offers several advantages, compared to older statistical methods. First, it builds independent predictors, a necessary condition in regression analyses. Second, the variances of these spatial predictors equal associated spatial autocorrelation following Moran's I (Moran, 1950). Third, predictors are hierarchized from the largest (largest scale, positive autocorrelation) to the smallest one (smallest scale, negative autocorrelation). Here, the spatial decomposition of the 73 stations results in 72 spatial descriptors, called Moran's Eigenvector Map (MEM; Griffith et al., 2006)). MEMs provide anisotropic spatial descriptors as they enable to identify multi-directional processes. Since lotic systems experience strong directional patterns, isotropic spatial descriptors were proposed by F. Guillaume Blanchet et al. (2008). These descriptors, called Asymmetric Eigenvector Maps (AEM), can be built from a PCA on a weighted matrix containing affinities between sampling stations and connections among them. Hereby links among sampling stations not connected by possible upstream-downstream paths are prevented. This calculation also resulted in 72 different AEMs.

Although both methods were initially developed for community ecology, they have a great potential in other research fields like river landscape ecology. In recent years, they have been used to analyse different types of species and communities within river systems (e.g.

Bertolo et al., 2012; Bourgeois et al., 2016; Landeiro et al., 2011; Liu et al., 2013; Massicotte et al., 2011). Investigating such scale-dependent processes should improve our understanding of fundamental river ecosystem functioning and underline the importance of the different concepts. For example, although the RCC is considered to be largely falsified (Poole, 2010), recent applications of the methodology of F. Guillaume Blanchet et al. (2008) seem to prove the validity of the concept in at least some rivers (e.g. Mortillaro et al., 2012). Combining both types of structures within one analysis should allow us to distinguish the relevant types (isotropic and anisotropic) and their size (Blanchet et al., 2011).

The most significantly correlated spatial scales to the predicted WQ pattern were identified by PCAIV with a forward selection following the method from F. G. Blanchet et al. (2008), based on both alpha significance level and adjusted coefficient of multiple determination (adjusted R2) as stopping criteria. This strongly conservative procedure lowers type I error inflation and was designed for variable selection in general, and more specifically for spatial variables, always in high numbers. In order to assess the relevance of land use predictors, the whole analytical procedure was also conducted on the residual pattern between land uses and WQ parameters. This procedure, using the predicted pattern of water qualities by land uses, focuses only on the spatial scales of human-mediated processes.

Figure 4-2 summarizes the analytical procedure. Computations and associated graphical representations were implemented under R version 3.2.0 (R Core Team, 2015) with the packages “ade4” (Chessel et al., 2004; Dray et al., 2007) for PCAIV, and “adespatial”, “AEM” (F. Guillaume Blanchet et al., 2008) and “packfor” packages for MEM and AEM computations and forward selection, respectively.

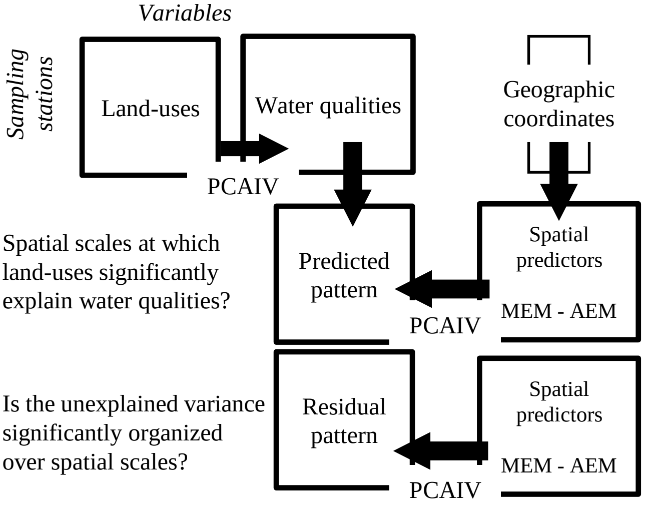

Figure 4-2: Conceptual schema of the analytical procedure.

3. RESULTS

3.1 Water quality and land uses

Water quality was significantly influenced by land uses in winter and summer (Figure 4-3). Land use explained 29% (winter) and 32% (summer) of WQ variation ($p < 0.001$). In both periods, water quality predictions were structured along three main axes on which covariances between land uses and WQ were mostly similar. Axis 1 expressed in both periods a size effect in most of the WQ parameters, explaining more of the variation in winter than in summer. This size effect was mainly explained by land uses that are related to urbanization: paved areas, WWTP connected and not-connected houses

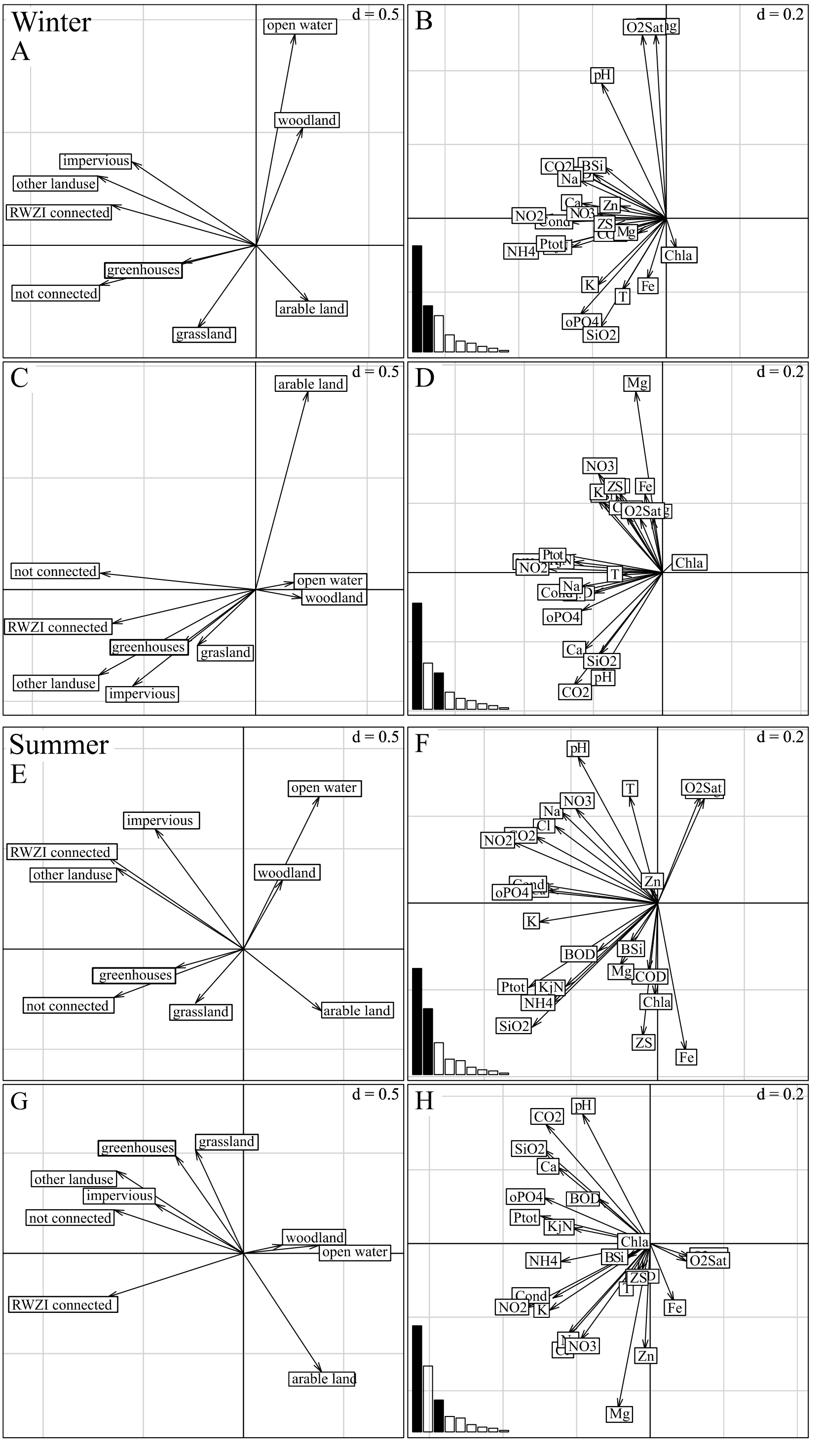

Figure 4-3: PCAIVs of the pure relationships between land uses and water qualities. A-B) and E-F) Axes 1 (horizontal) and 2 (vertical). C-D) and G-H) Axes 1 and 3 (vertical). Bar diagrams, eigenvalues. Winter: Axis 1, 45%; Axis 2, 20%; Axis 3, 15%. Summer: Axis 1, 42%; Axis 2, 26%; Axis 3, 13%. “d” 0 the grid scale.

and other land uses. These land uses covaried with a large group of WQ parameters that had a stronger clustering in winter. In winter, Axis 2 was mainly explained by open water and woodland covarying with O_2 and pH, and opposed to grassland and arable land covarying with T, PO_4, SiO_2, K and Fe. WWTP connected houses, paved areas and other land uses were opposed to not connected houses and greenhouses along the second axis. With NH_4, KjN and Ptot covarying with one group and Na, CO_2, Cl^- and BSi covarying with the other. In summer, Axis 2 described more variation compared to winter and was also explained by open water, woodland and paved areas; whereas grassland and arable land were still in opposition. Temperature shifted to covary in a positive way to O_2 and pH, and other elements such as NO_2, NO_3, CO_2, Na and Cl. The opposition between WWTP connected and not connected houses along Axis 2 appeared to be stronger compared to winter. With WTTP connected houses covarying with $NO_{3,}$ Na, Cl, CO_2 and NO_2 and not connected houses with K, KjN, NH_4, Ptot, SIO_2 and BOD.

Axis 3 expressed mainly an opposition between arable land and grassland with greenhouses, whereas Mg, O_2, K, NO_3, SS, Zn and Fe strongly covaried with arable land in both seasons. O_2 was less variable; SiO_2, CO_2, pH and Ca were mainly explained by grassland and glasshouses.

3.2 Explained water quality variation and spatial patterns

WQ within the Kleine Nete catchment was highly variable both between and within the two periods (Figure S4.5). In both periods more or less halve of the parameters was higher compared to the other period: summer, 12; winter, 13. WQ parameters alone were first spatially modelled to describe their raw pattern. As winter and summer WQ tables were significantly correlated to the geographic coordinates of the sampling station (PCAIV of water qualities on geographic coordinates; $p < 0.001$), the tables were detrended to avoid spurious correlations (Sharma et al., 2011). No AEM was found to be significant for the WQ data, while several MEMs explained only 11% of winter and summer pattern variances respectively ($p < 0.001$; Table 4-1). PCAIVs plots can be found in the supplementary materials (Figure S4.6).

Table 4-1: Results of spatial modelling displaying the significant spatial predictors explaining significantly the raw water quality patterns.

Season	Spatial predictor	R^2	Adjusted R^2	Cumulated explained variance	F	p-value
Winter	MEM9	0.052	0.052	0.039	3.91	0.002
	MEM2	0.041	0.093	0.067	3.15	0.006
	MEM7	0.040	0.133	0.095	3.15	0.003
	MEM6	0.029	0.162	0.112	2.35	0.013
Summer	MEM2	0.046	0.046	0.033	3.46	0.003
	MEM7	0.039	0.086	0.060	3.00	0.007
	MEM49	0.030	0.116	0.077	2.34	0.018
	MEM20	0.027	0.143	0.093	2.18	0.024
	MEM11	0.027	0.170	0.108	2.20	0.034

After modelling WQ and spatial predictors, also land use was taken into account. In this analysis both seasonal patterns were significantly organised in space ($p < 0.001$; Table 4-2). Space explained for both analyses circa 22% of variance. A first result was the absence of significance of AEMs in both patterns, the essential of the spatial structure being explained by multi-directional waves (MEMs). Second, the results showed that these seasonal analyses of human-mediated processes (i.e. land uses vs WQ relationships) were twice more significantly spatialized (22% of explained variance) than the WQ pattern alone (10 %). Third, broad scales had no dominant contributions to the observed patterns which were rather explained by mixes of scales ranging from a few km to the maximum, with summer having both larger (20-50km, MEM1 and MEM2; Figure 4-4 and Figure 4-5) and smaller scales (a few kilometres, MEM 59).

Table 4-2: Results of spatial modelling displaying the significant spatial predictors retained in PCAIV and their contributions to the land uses – water qualities relationships.

Season	Spatial predictor	R^2	Adjusted R^2	Cumulated explained variance	F	p-value
Winter	MEM36	0.052	0.052	0.038	3.87	0.008
	MEM7	0.041	0.092	0.066	3.13	0.020
	MEM10	0.038	0.130	0.092	3.01	0.030
	MEM4	0.038	0.168	0.119	3.08	0.021
	MEM12	0.037	0.205	0.145	3.10	0.023
	MEM2	0.035	0.240	0.171	3.08	0.017
	MEM9	0.032	0.272	0.194	2.85	0.029
	MEM48	0.031	0.303	0.215	2.80	0.027
Summer	MEM10	0.051	0.051	0.037	3.80	0.006
	MEM7	0.040	0.091	0.065	3.07	0.018
	MEM2	0.039	0.130	0.092	3.12	0.016
	MEM36	0.034	0.164	0.115	2.79	0.031
	MEM6	0.032	0.196	0.136	2.65	0.048
	MEM1	0.031	0.227	0.157	2.68	0.032
	MEM5	0.030	0.258	0.178	2.65	0.036
	MEM15	0.029	0.286	0.197	2.57	0.047
	MEM59	0.028	0.315	0.217	2.61	0.037

The winter WQ pattern was similar to previously described by the WQ – space analysis; the largest scales (MEM2, MEM4 and MEM7) were mainly expressed along the first axis, explaining the size effect which appears to be mostly related to increases in urbanization. The opposition along the second axis was mainly explained by smaller scales and the opposition along the third axis by a mix of different scales. Both second and third axis showed oppositions between different types of urbanisation, RWZI-connected and not connected houses, and types of agriculture, greenhouses, grassland and arable land. These oppositions along the second and third axis could be related to differences in nutrient concentrations, but also to other WQ parameters.

The summer WQ pattern, was also similar as previously described, but encompassed only two axes. Both size effect and opposition were explained by mixes of different scales. Compared to winter the effects were more diffuse with all WQ parameters changing over different sizes of scales.

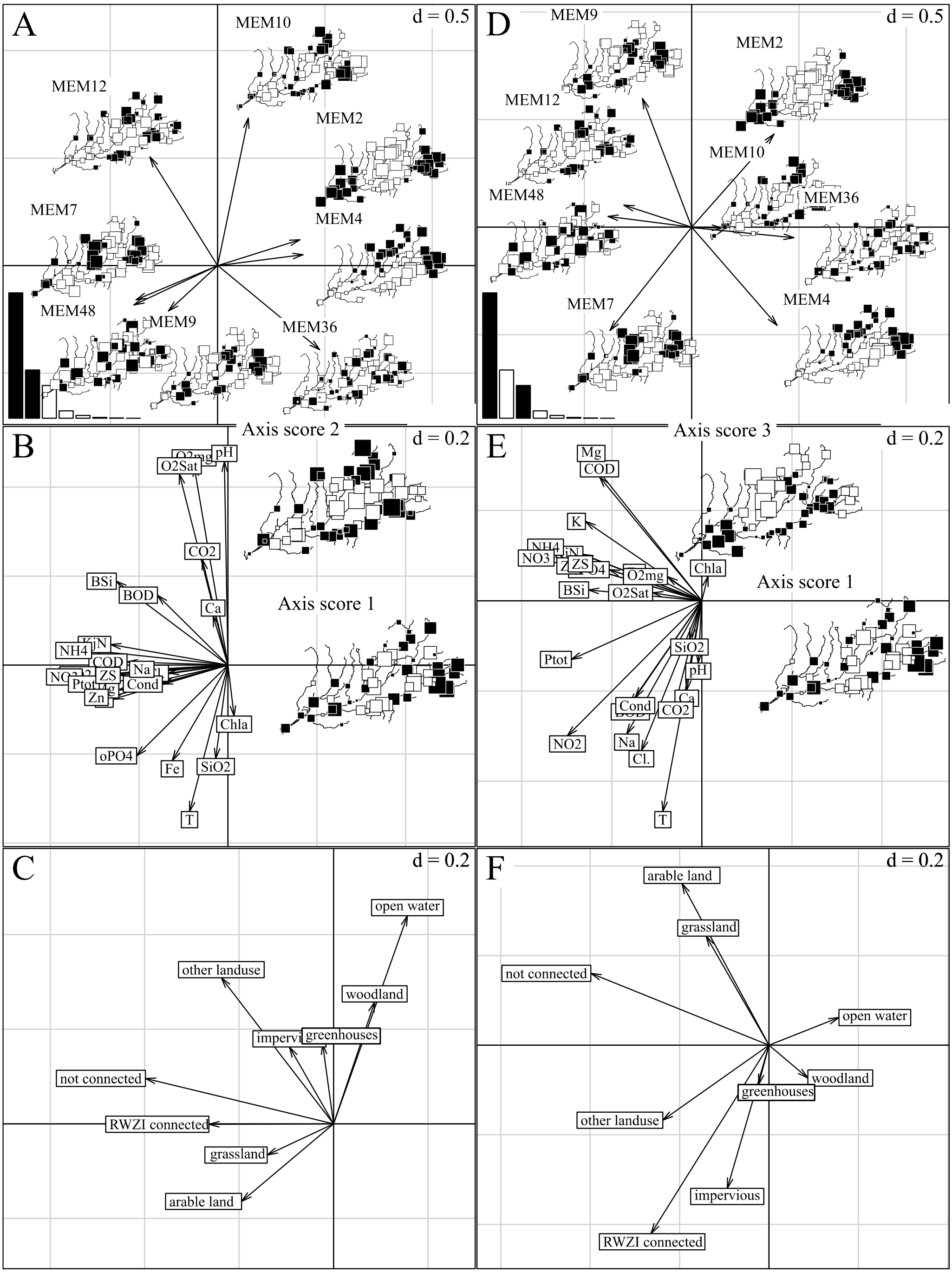

Figure 4-4: PCAIV of the winter relationships between human-mediated processes (relationships between land uses and water qualities) and spatial descriptors. A-C) Axes 1 and 2. D-F) Axes 1 and 3. Bar diagrams, eigenvalues. Axis 1 (horizontal), 58%; Axis 2 (vertical), 22%; Axis 3 (vertical), 15%; A and D) Spatial predictors; inserts illustrate spatial wave length expressed by MEM score distributions; large white squares, low values; large black squares, high values; small squares, intermediate values. “d” indicates the grid scale.

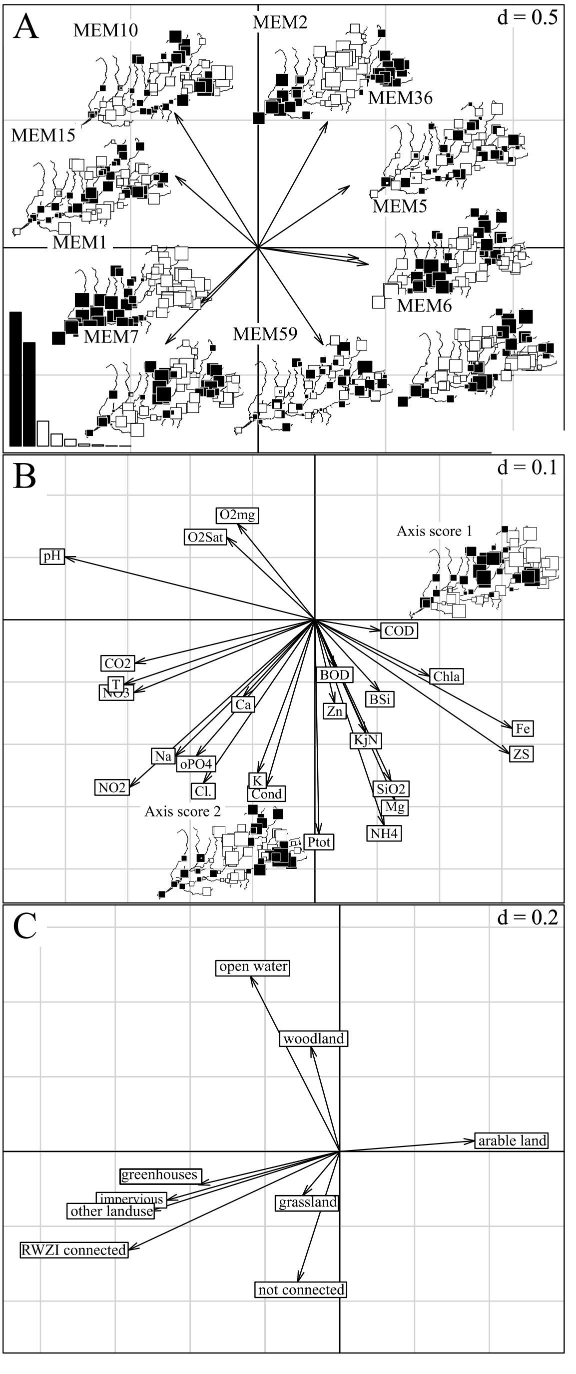

Figure 4-5: PCAIV of the summer relationships between human-mediated processes (relationships between land uses and water qualities) and spatial descriptors. Bar diagram, eigenvalues. Axis 1 (horizontal), 47%; Axis 2 (vertical), 36%. A) Spatial predictors; inserts illustrate spatial wave length expressed by MEM score distributions; large white squares, low values; large black squares, high values; small squares, intermediate values. "d" indicates the grid scale.

3.3 Unexplained water quality variation and spatial patterns

Only part of variance in the WQ data is explained by the upstream land use indicators in section 3.1. The residual variance from the winter analysis was not significantly spatialized ($p > 0.05$). In summer 15% of residual variance was explained by space (Table 4-3); again, no AEM was significant. The pattern was predominantly explained by MEMs expressing scales of a few kilometres. The structures identified in the axis scores clearly represent smaller scale variation compared to the axis scores of the previous analysis. While the clusters of WQ parameters are mostly similar to those found in the results for summer in analysis 3.3. Axis 1 is characterized by oxygen demanding parameters such as NH_4, Ptot, oPO_4 and KjN which are opposed by oxygen itself. Axis 2 is mainly composed out of Na, Cl, Cond, NO_3, and NO_2.

Again MEM2 and MEM7 were found to be significantly related. MEM2, as in analysis 3.3 - summer, covaries with most of the WQ parameters along both axes, except for oxygen and pH and additionally Ca and CO_2 along axis 2. MEM7 covaries, as in analysis 3.3 - summer, with Ca and CO_2 and additionally with oxygen and pH. PCAIVs plot of the analysis can be found in the supplementary materials (Figure S4.7).

Table 4-3: Results of spatial modelling displaying the significant spatial predictors retained in PCAIV and their contributions to the residuals from the land uses – water qualities summer relationships.

Spatial predictor	R^2	Adjusted R^2	Cumulated explained variance	F	p-value
MEM7	0.033	0.033	0.019	2.39	0.015
MEM20	0.030	0.063	0.036	2.27	0.016
MEM51	0.029	0.092	0.053	2.23	0.024
MEM8	0.029	0.121	0.070	2.25	0.017
MEM55	0.027	0.148	0.085	2.11	0.026
MEM59	0.026	0.175	0.099	2.11	0.027
MEM44	0.025	0.200	0.114	2.06	0.027
MEM49	0.025	0.224	0.128	2.03	0.027
MEM2	0.024	0.249	0.141	2.03	0.031
MEM18	0.024	0.272	0.155	2.01	0.034

4. DISCUSSION

Upstream land uses appear to be a relatively strong predictor for WQ variation, which is in line with previous results (Allan, 2004; Baker, 2003). Increases in most parameters covary in first instance along a gradient of increasing urbanization. However, there is a clear distinction between RWZI connected and not-connected houses mostly, which confirms earlier findings within the catchment (Vrebos et al., 2015). Land uses were more distinctly covariant in summer (Figure 4-3C), with possibly similar impact on water qualities among land uses. Only oxygen is negatively impacted because of break down oxygen demanding nutrients related with the anthropogenic land uses (de la Crétaz et al., 2007). While many studies only assess a limited number of parameters, our results signify that the entire chemistry balance is affected by human activities.

The WQ in the catchment appears to be highly variable: both between winter and the rest of the year and between sample points. WQ changes take place along different gradients of different sizes. When land use was included, other large and small MEMs were found to be significant in both periods. The increase in selected MEMs is most likely a result of the fragmented land use. Different processes and pollution sources might counteract each other, obscuring the patterns of the different processes. This can result in an underestimation of the actual complexity of the system. Selection of general and additional, large and small scale structures between both periods illustrate both land use impacts that are relatively stable throughout the year, as well as impacts that differ between seasons. Our results suggest that WQ varies on several scales, underpinning the catchment's complexity. Only two different spatial structures were found to be significant in all different steps of the analysis, signifying that some processes do remain constant throughout the year, working at stable scales.

The impact of land uses and space in winter seems to take place along three gradients. A first gradient is related to increases in urbanization, which takes place on larger scales, the second gradient relates to different types of urbanization and a third to different types of agriculture. Urbanization changes on larger scales with alternating areas of cities, smaller villages and open land. While types of urbanization and agriculture differ on smaller scales. Neighborhoods or houses that are connected to a WWTP or not, alternate on a small scale within the catchment due to complex spatial planning and urban sprawl. In summer these effects between land use, WQ and space are more diffuse and complex. This can be an effect of the biological processes which can differ strongly within the catchment and within the streams, Decreases in flow rate in summer can also increase the, sometimes local, impact of pollution sources with a relatively stable load discharge. The combination of both biological processes and local pollution sources results in a more complex WQ pattern, a less pronounced impact of land use across the catchment and the selection of smaller MEMs in summer.

The origin of medium and small scale structures within the unexplained water quality variation is difficult to uncover. Medium and small MEMs range within several kilometers and encompass several independent sample points that are situated on neighboring tributaries, but are fed by the same groundwater system. This indicates that vertical vectors, related to seepage, can have an important impact on a smaller scale. Considering the sandy soil and groundwater fed river system, water table exchanges likely occur within the catchment (Anibas et al., 2011). The impact of local variation in groundwater quality, due to seepage location and pollution sources is estimated to increase in summer as runoff and dilution effects decrease and deeper groundwater becomes more important in feeding the tributaries. These small MEMs are however still logically structured, creating a distinct pattern across the catchment. Yet using these structures to identify other specific properties or processes within the catchment remains difficult because of their low explanatory strength and the absence of relevant information. Why these small scale patterns are only found in summer and not in winter remains questionable.

An important result was the absence of significance of the isotropic structures in the analyses. Potential upstream-downstream effects cannot be incorporated within land use predictors and should therefore be retained within the remaining variation. As a result, we expected to find isotropic structures within the unexplained variance. Especially given the high density of sampling stations, which could have captured upstream-downstream trends in WQ characteristics. In-stream processes should after all have longitudinal isotropic effects within a stream and therefore be represented by isotropic structures in the analysis. AEMs were designed for spatial modelling in lotic systems, outcompete MEMs in such conditions (Blanchet et al., 2011) and have been successfully applied in river ecology (e.g. Wan et al., 2015). However, within our catchment, they appeared to be inadequate to explain any WQ variation. The total absence of significant unidirectional predictors raises crucial questions. Our results suggest that the upstream-downstream pattern, as described in the RCC, is not directly applicable to our catchment. Whether this is because the upstream–downstream pattern was never there or because the anthropogenic impact and cumulative effects on the river system has overruled its initial state, is impossible to answer (MacDonald, 2000). Legacy effects such as past agricultural land uses, not incorporated in our analysis, can effect WQ (Maloney et al., 2011). Also, some of the variability might be due to the diurnal cycle of many parameters, which was not incorporated fully within the analysis.

Managing WQ in heavily altered river systems is challenging. Despite large efforts during the past two decades, only a few streams in the Kleine Nete reach the European 'Good Ecological Health' status. Our results indicate that WQ if affected by both land use and complex mixes of spatial scales. As different sources and processes can obscure and interact with each other, it is generally difficult to understand the correct impact of different pollution sources and their predominant pathways. The intricacy of the river system makes the management of it a daunting challenge, requiring significant amounts of investment in both time and effort.

5. CONCLUSION

Advances in spatial statistics now give us tools to analyze different pathways in ecological studies and help us understand the spatial complexity of rivers systems. The interaction between the natural characteristics of the catchment and human activities has created a river system of which the WQ variation and its sources are difficult to analyze and understand. Our results advocate for continuous small scale monitoring to better understand these local variations across the catchment and the impact of pathways such as groundwater flows on the river WQ. Although parts of the WQ variation can be attributed to land uses, more research related to impact of groundwater and other potential pathways is complementary needed.

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APPENDIX

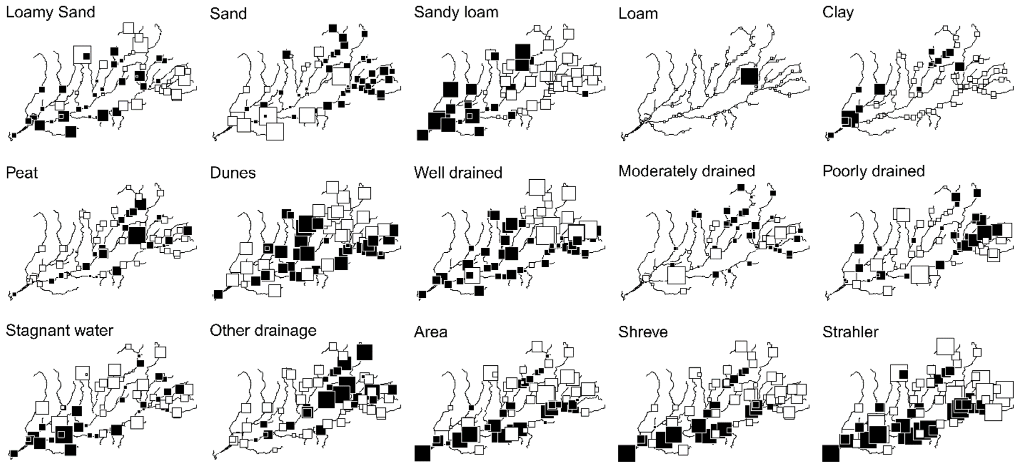

Fig.S4.1: Upstream physical characteristics of the study area at each sampling station. Values are normalized; large white squares, low values; large black squares, high values; small squares, intermediate values.

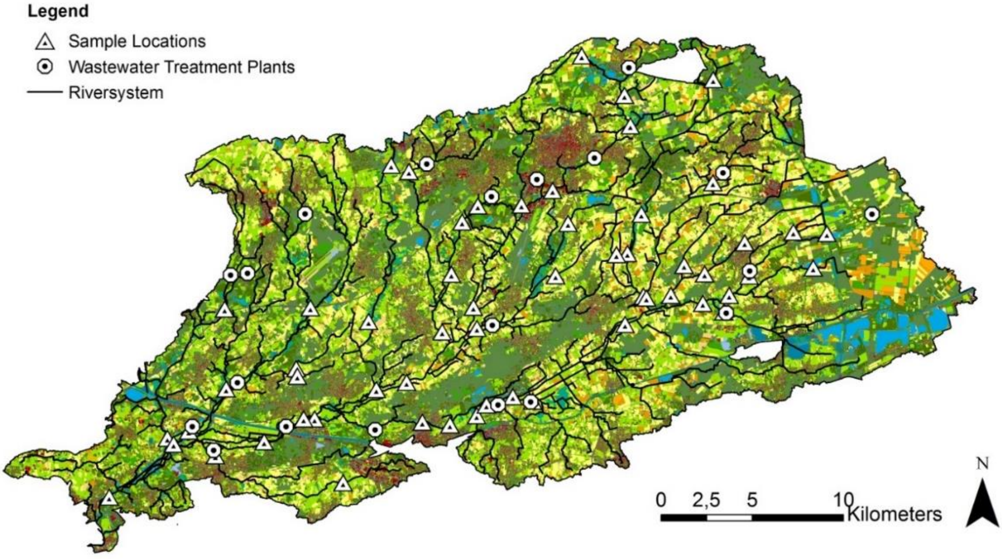

Fig. S4.2: Overview of the Kleine Nete catchment with sample locations and wastewater treatment plants.

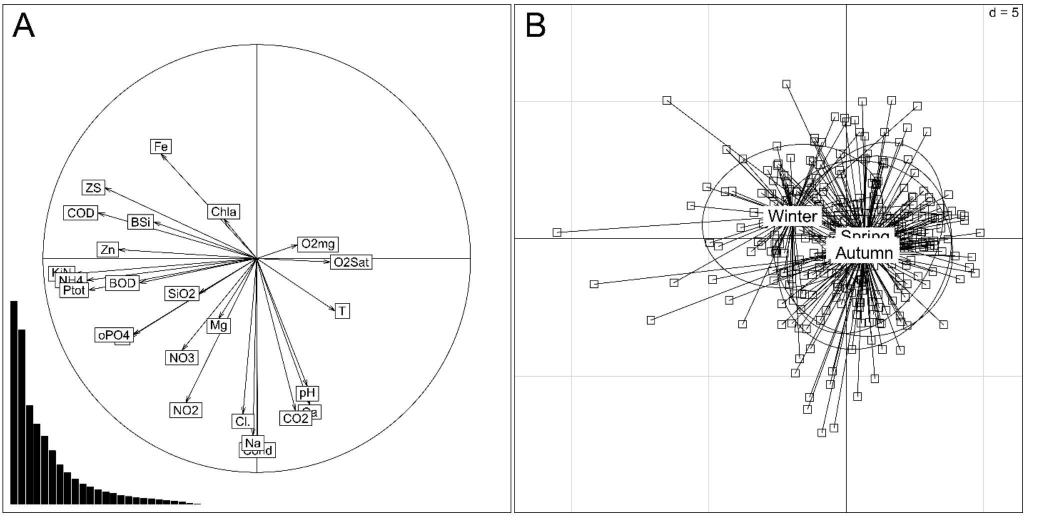

Fig. S4.3: PCA on the whole water qualities data set, with values averaged per season and per sampling station. A) Correlation circle; eigenvalue diagram: Axis 1 (horizontal), 23 %; Axis 2 (vertical), 19 %. B) Samples (black squares) grouped per season. "d" indicates the grid scale. The first axis discriminates the winter season from the three others. An analysis of variance of the seasonal effect on Axis 1 evidenced a significant effect ($p < 0.001$); a post hoc test (Tukey's HSD) revealed highly significant differences between the winter group and the other groups ($p < 0.001$) whereas no significant difference was detected among the three other groups ($p > 0.990$).

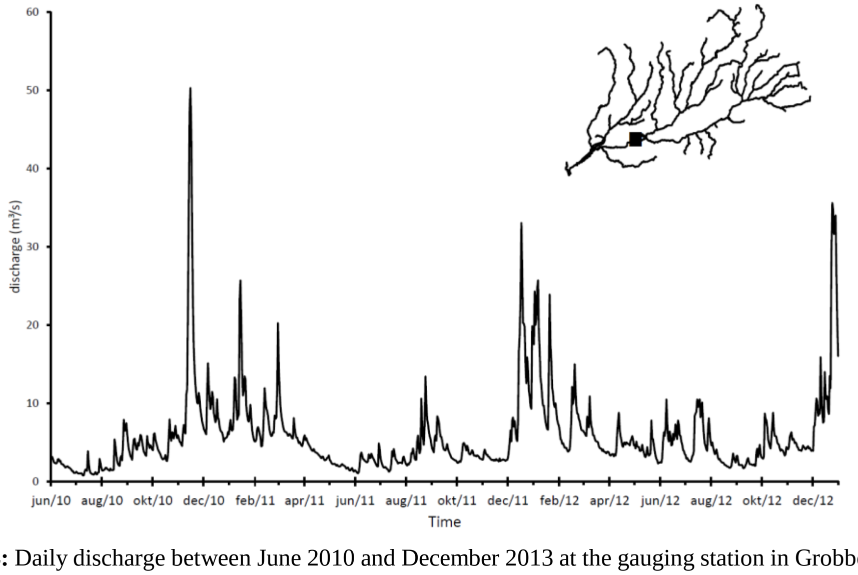

Fig. S4.4: Daily discharge between June 2010 and December 2013 at the gauging station in Grobbendonk. Flows are highest in winter season, but can be highly variable throughout the year.

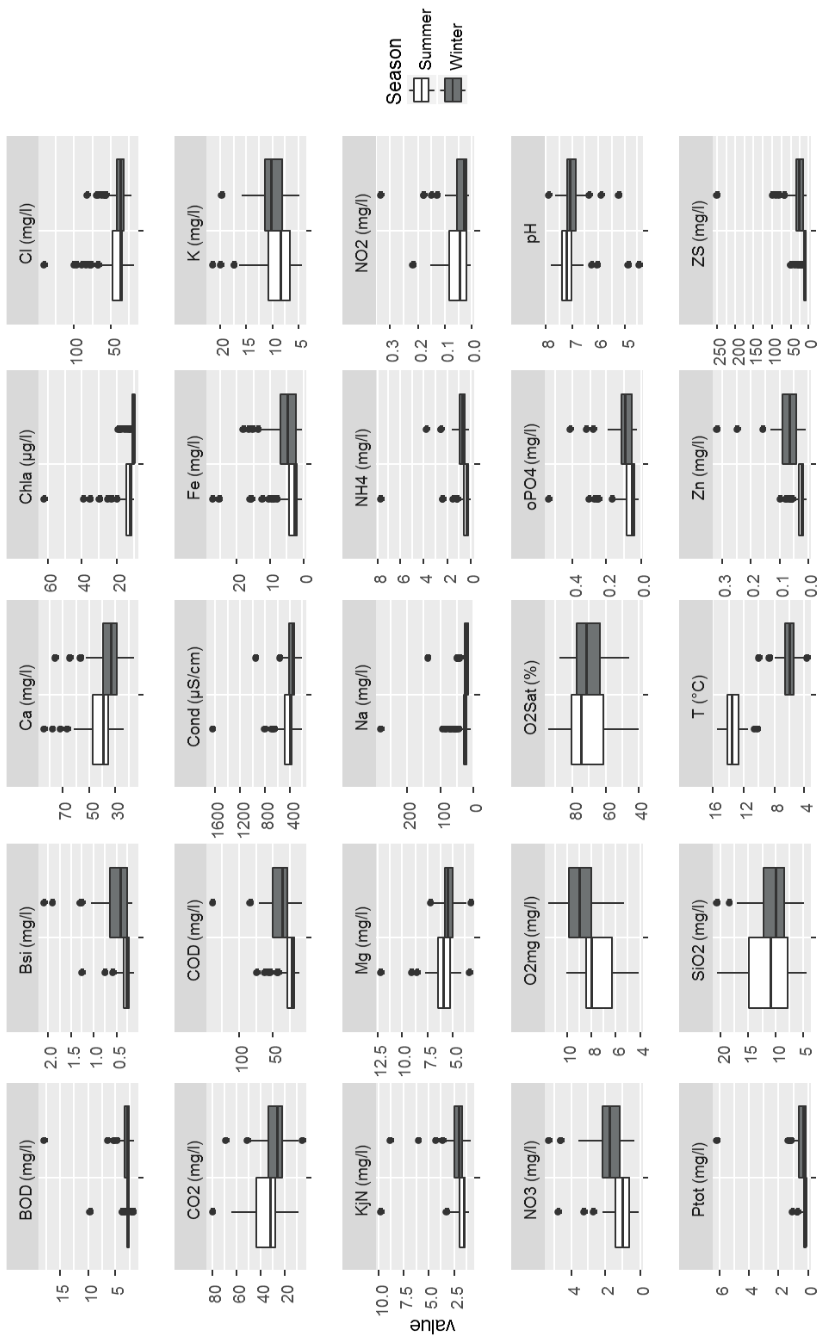

Fig. S4.5: Overview of the variation for the different WQ parameters across the catchment in summer and winter. The upper and lower "hinges" correspond to the first and third quartiles (the 25th and 75th percentiles). The upper whisker extends from the hinge to the highest value that is within 1.5 * IQR of the hinge, where IQR is the inter-quartile range, or distance between the first and third quartiles. The lower whisker extends from the hinge to the lowest value within 1.5 * IQR of the hinge. Data beyond the end of the whiskers are outliers and plotted as points (R version 3.2.0 – ggplot2).

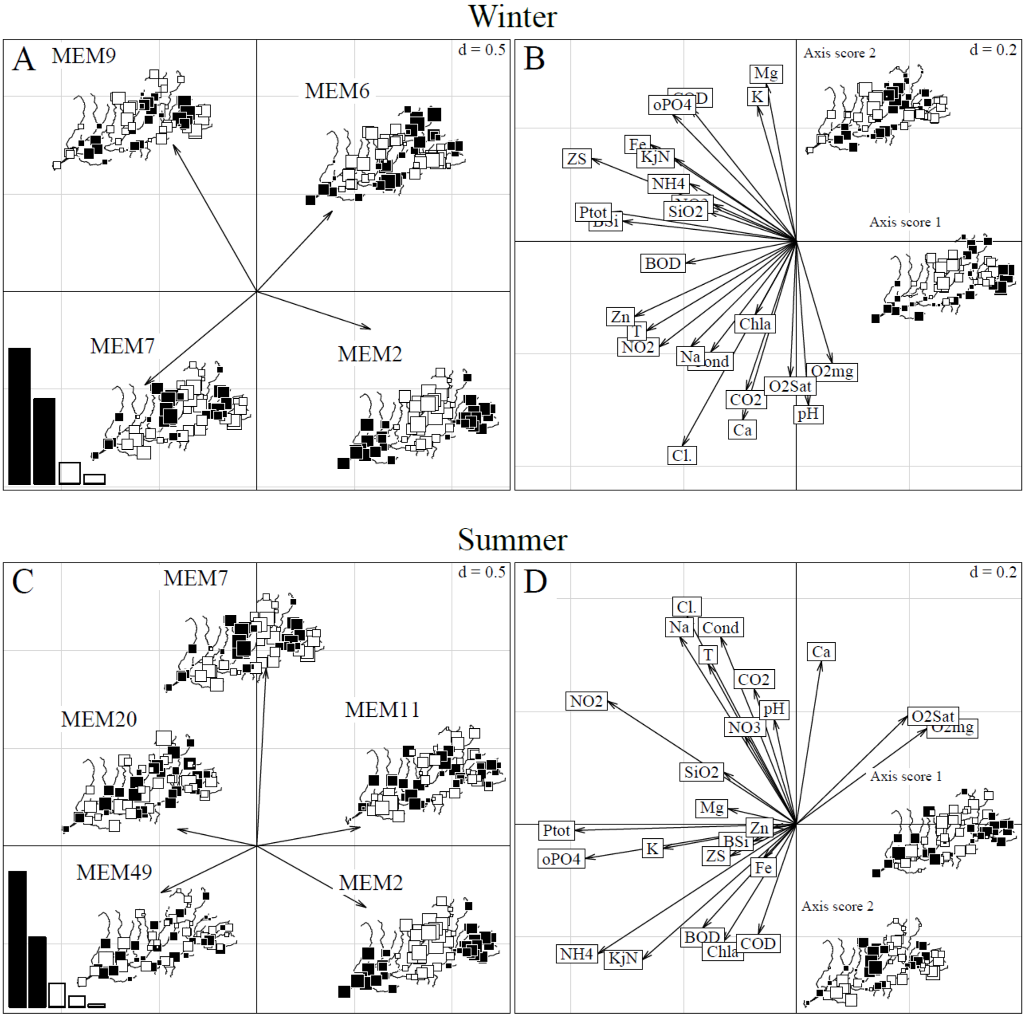

Fig. S4.6: PCAIV of water quality predictions by spatial variables. A and C) Spatial predictors; inserts illustrate spatial wave length expressed by MEM score distributions; large white squares, low values; large black squares, high values; small squares, intermediate values. B and D) Water quality descriptors; insert illustrate the axis score predictions. Bar diagrams, eigenvalues. Winter: Axis 1 (horizontal), 54%; Axis 2 (vertical), 34%. "d" indicates the grid scale. Summer: Axis 1 (horizontal), 56%; Axis 2 (vertical), 29%.

Water quality within the Kleine Nete catchment is highly variable both between and within the two periods (Fig. S3) In both periods more or less halve of the parameters is higher compared to the other periods: summer, 12; winter, 13. Water quality parameters alone were first spatially modelled to describe their raw pattern. As winter and summer WQ tables were significantly correlated to the geographic coordinates of the sampling station (PCAIV of water qualities on geographic coordinates; $p < 0.001$), the tables were detrended to avoid spurious correlations (Sharma et al., 2011). No AEM was found to be significant for the water quality data, while several MEMs significantly explained only 11% of winter and summer pattern variances respectively ($p < 0.001$; Table 1). The contributions of the different MEMs were relatively homogeneous and ranged between 3 and 5%. The main difference between the two periods was the greater spatial scale of the predictors in which water quality varied in winter, ranging from 10-15km to 20-25km whereas the summer pattern exhibited variation at much smaller spatial scales of circa a few km.

In both cases, the main part of the pattern was characterized by a size effect whereby most of the elements covaried along the first axis. This size effect coincides largely with the areas in the Kleine Nete catchment that are considered to be in good ecological health (axis score 1). In summer there is an effect of increased oxygen with lower concentrations of other parameters. This size effect occurred in both periods and was mainly related to MEM2, a large upstream-downstream wave over which different nutrients such as Ptot, oPO_4, NO_2, NH_4 and KjN expressed the main part of the variation. In winter, the second axis was characterized by an opposition between oxygen, Cl, Na, Cond, pH, Ca, pH, and CO_2 on the one hand, and Mg, K, oPO_4 and COD on the other hand, mainly explained by MEM7 and MEM9. In summer, the main part of the opposition remained however with a larger Chla variation (still explained by MEM7) and a shift of COD and oxygen in direction.

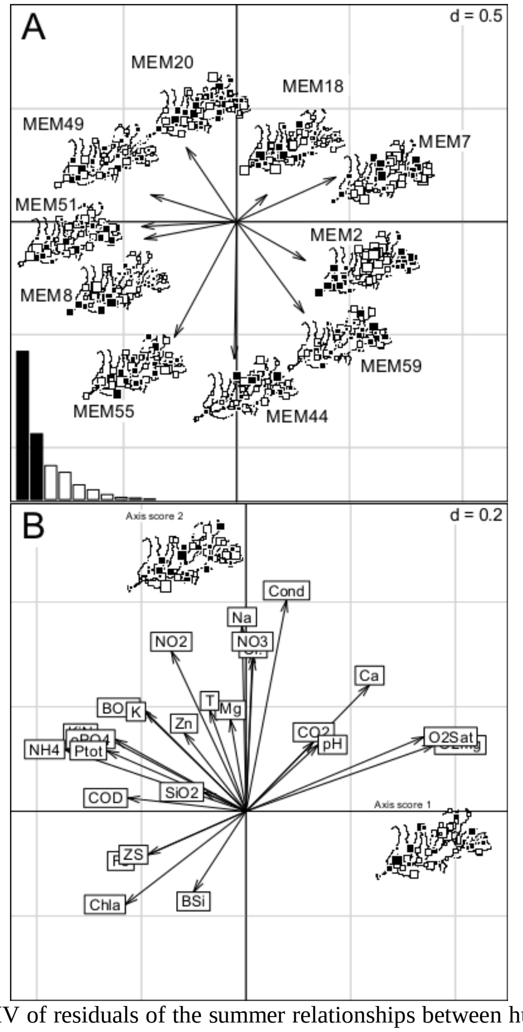

Fig. S4.7: PCAIV of residuals of the summer relationships between human-mediated processes (relationships between land uses and water qualities) and spatial descriptors. Bar diagram, eigenvalues. Axis 1 (horizontal), 48%; Axis 2 (vertical), 21%. A) Spatial predictors; inserts illustrate spatial wave length expressed by MEM score distributions; large white squares, low values; large black squares, high values; small squares, intermediate values. “d” indicates the grid scale.

Chapter 5 - Site selection for ecosystem service development within a catchment: challenges and opportunities.

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Abstract. Catchments are complex, dynamic systems, impacted by anthropogenic developments, with processes taking place on different temporal and spatial scales. To manage catchment characteristics sustainably, natural and anthropogenic properties need to be evaluated along the river continuum. Within this study we developed a methodology that allows for the upstream-downstream evaluation of catchment properties through a routed river network. It can evaluate nitrogen concentrations within the catchment and explore the possibilities for targeted nitrogen removal through ecosystem service development along the reaches. Our results illustrate the opportunities that ecosystem services present in catchment management and the importance of evaluating them along the river continuum, taking both between supply and demand into account. However, challenges remain to develop reliable indicators that can cope with changes in water quality data and sampling, and the lack of suitable flow data. The methodology was tested and evaluated for the Nete catchment in Belgium.

1. INTRODUCTION

In 2000 the Water Framework Directive (WFD) was adopted by the European Commission (EC) (2000/60/EC), and implemented by the Member States (MS) in the following years. The main aims of the WFD are to protect and restore the water environment within the European Union (EU) and to manage water resources sustainable, taking into account environmental, economic and social considerations. These goals are to be reached through a shift from previous fragmented, inefficient policies to a more holistic approach integrating all parts, interactions and interdependencies of the wider environmental system. This requires a change in management towards system thinking (Arnold et al., 2015; Voulvoulis, 2012). This shift is implemented through the development of “river basin management plans” grouping all management actions designed to achieve a good ecological status of all natural surface water bodies and good ecological potential for heavily modified water bodies within the catchment. The WFD aims to implement the concept of “Integrated Catchment Management” (ICM) within the EU (European Commision, 2000).

Historically water resources management is commonly achieved by bringing ecosystems under control, while neglecting underlying ecological complexities (Holling et al., 1996). It aims to stabilize resource outputs and to diminish environmental variability and natural disturbances, increasing their predictability and stability (Carpenter et al., 2001). However, many ecosystem processes can never be forced into a controlled regime, as they are part of a complex web of processes that operate on various scales in time and space. This pathology of ‘command

and control' affects the resilience of the ecosystems, which leads to their degradation or collapse and the services they provide (Briggs, 2003; Fischer et al., 2009; Holling et al., 1996).

Natural resource management strategies, such as ICM, try to move away from this 'command and control' paradigm and incorporate ecological complexity, variation and uncertainty in their methodologies to transfer towards sustainable development (Hjorth et al., 2006; Medema et al., 2008; Rammel et al., 2007). To achieve this and to ensure the resilience of the entire catchment, the different processes and flow paths should be recognized and taken in to consideration. Therefore, it is essential to recognize catchments as interlinked systems in which land uses and anthropogenic activities within the catchment have direct and indirect impacts on the entire river system and its ecosystems (Allan, 2004; Allan et al., 1997). As a results upstream and downstream activities have to aligned and be compatible to each other (Falkenmark, 2004).

Although the WFD was designed to integrate ICM within European policy and it has transformed water management within the EU, criticism has been given regarding its inability to fully implement a catchment-based system approach. This is partly due the overwhelming diversity in national implementation (Hering et al., 2010; Josefsson et al., 2011; Keessen et al., 2010; Liefferink et al., 2011; Moss, 2008; Voulvoulis et al., 2017). Currently the WFD fails to fully measure, assess and communicate the benefits of water management measures with regard to the value of environmental quality, jeopardizing political and social support (Everard, 2012). In order to better align the WFD with its initial goals and better integrate and communicate the importance of environmental quality, the concept of "ecosystem services" (ES) has a high potential (Everard, 2012; Vlachopoulou et al., 2014).

Ecosystem services are the used contributions of ecosystem structures and functions to human well-being (Daily, 1997; MEA, 2005). The ES concept offers a valuable methodology to link society to nature and reinforces arguments to conserve and restore ecosystems, while providing a common language for managers, researchers and different stakeholders (Granek et al., 2010). It has the potential to become helpful in integrated natural resources management (INRM) and water resources management (Grizzetti et al., 2016a; Liu et al., 2013). It can also assist in identifying and negotiating trade-offs between a range of management options and to develop policies aligning private incentives with societal objectives (Engel et al., 2013). However, its place within management frameworks such as INRM is sometimes contested (Baker et al., 2013; Cook et al., 2012). Nevertheless, recent studies have evaluated and discussed the potential of the ES concept towards its implementation within the WFD (Doherty et al., 2014; Vlachopoulou et al., 2014). Its integration within the WFD can provide a first step towards acknowledging the WFD systemic intent (Vidal-Abarca et al., 2016; Voulvoulis et al., 2017) and increase the effectiveness of management actions that are integrated in river basin management plans (RMBPs) (Terrado et al., 2016). Although the ES concept is already incorporated in several RBMPs of several countries (Grizzetti et al., 2016b), a more consistent integration across Europe is required.

To successfully implement the ES framework into ecosystem management, different aspects and dimensions need to be taken into account (Seppelt et al., 2011; Seppelt et al., 2012). The successful delivery of ES to society depends on both areas that provide ES (Service Providing Areas, SPA) and areas that benefit of these ES (Service Demanding Areas, SDA). which can be separated from each other in time and space and can be connected through different flow mechanisms (Service Connecting Areas, SCA) (Fisher et al., 2009; Luck et al., 2003; Luck et al., 2009; Syrbe et al., 2012). Often mismatches between ES supply and demand are caused by the spatial distribution of the SPA and SDA and the absence of appropriate SCA (Wei et al., 2017). To successfully implement the ES concept within the WFD, these aspects of space and time that connect SPA with SDA need to be taken into account (Bastian et al., 2012).

Many ES are strongly related to the hydrological cycle, streams and rivers (Brauman et al., 2007). To sustain stream ecosystems and their services, processes such as minimal stream flow, water quality etc. have to be maintained, to ensure the necessary stream ecological and hydrological functions (Alan Yeakley et al., 2016; Baron et al., 2002; Keeler et al., 2012). While water quality in itself is contested as a final ES (Keeler et al., 2012), it is generally considered to be part of the “regulating and maintenance” ES (Haines-Young et al., 2013). At the same time maintaining a good water quality is part of the WFD and imperative to comply with its goal of “good ecological health”. As such the demand for water quality regulation is clearly defined within the European Union. Generally, water quality improvements are gained by, often expensive, technological measurements such as sewer infrastructure. However, in line with the ideas of ICM, restoration of natural systems, such upstream riparian buffers, can significantly improve downstream water quality and the delivery of a range of additional ecosystem services (Mitchell et al., 2013).

However, selecting upstream areas that have a high potential to deliver water quality improvement to downstream waterbodies remains a challenge. Within a river network, many point and diffuse pollution sources affect the water quality in a river system. The spatial allocation of remediation measures (ecosystem restoration) should be adjusted to the type and location of the different pollution sources. The main objective of this work was to establish a framework that:

1. allows for the upstream-downstream evaluation of catchment properties through a routed river network,
2. is able to evaluate ecosystem service demand and supply within a catchment context,
3. explores the possibilities for ecosystem service development / restoration for integrated catchment management.

Based on the overall framework, an evaluation system was developed and tested to assess the in-stream nitrogen concentrations (SDA) within a catchment and to identify opportunities for upstream nitrogen removal (SPA) through ecosystem restoration. The evaluation system was developed and tested for the Nete catchment in Belgium.

2. MATERIALS AND METHODS

2.1 Description of the methodology

Delivery of ecosystem services is defined by the location of both SPA and SDA, connected to each other through different flow mechanisms and SCA (Fisher et al., 2009). When we want to improve the delivery of ecosystem services, such as water quality regulation, within a catchment, potential supply locations need to be selected that are located upstream of the demand location and downstream of the sources of pollution. In order to select these locations, the developed methodology combines separate analysis of ES supply and ES demand in an overall ranking of areas for ES restoration (Figure 5-1). The following analysis steps are part of this framework:

1. <u>ES Demand:</u> Evaluation of current nitrogen concentrations:
    a. Assessment of the sample point (SP) under evaluation and identification of upstream SPs and wastewater treatment plants (WWTP).
    b. Evaluation of the four nitrogen components (N-component) and identification of the most probable source for nitrogen pollution based on changes in the N-components.
    c. Ranking of subcatchments based on the potential demand for nitrogen removal.
2. <u>ES supply:</u> Evaluation of potential nitrogen storage and removal:
    a. Development land use and groundwater maps of ES calculation.
    b. Calculation of actual and potential nitrogen storage and removal maps within the catchment.

   c. Aggregation of results on subcatchment level and ranking the subcatchment for their potential for additional ES supply.
3. Selection of areas: suitability ranking of subcatchments with opportunities for restoration of ES supply.

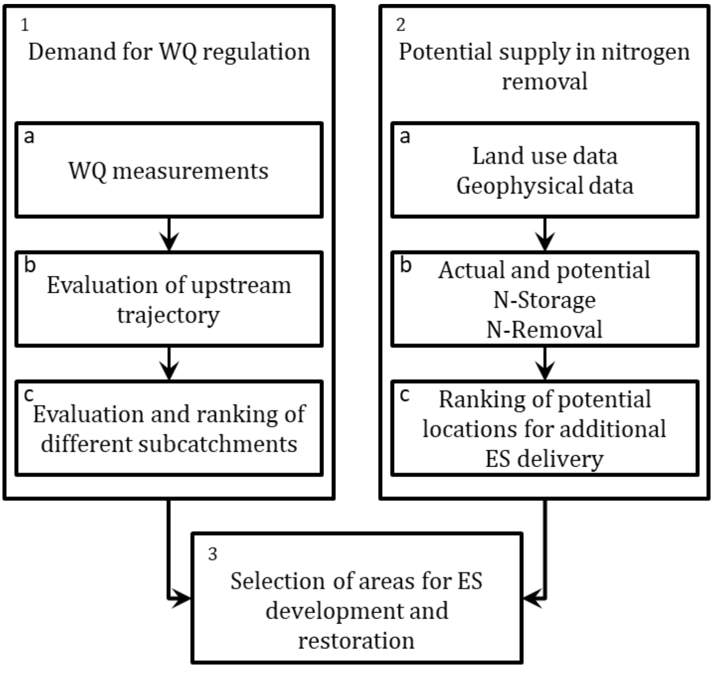


Figure 5-1: Overview of the different steps within the methodology.

2.2 Evaluation of current nitrogen concentrations

2.2.1 *Evaluation of nitrogen components*

As water quality standard, total nitrogen (Nt) within the Nete catchment is evaluated during the summer period (April to September). According to the Flemish water quality standards (B.S. 28/09/2010) mean Nt concentrations for this period (at least 4 measurements) need to be below 4 mg N/L. To select areas for ecosystem restoration however, upstream-downstream changes of four different N-components, that make up Nt, are evaluated separately (nitrate - NO_3, nitrite - NO_2, ammonium - NH_4 and the remaining fraction - $N_{Remains}$). As such, a better distinction can be made between the impacts of different tributaries and land uses, compared to evaluation of only Nt.

Each sample point (downstream sample point or SPd) is evaluated based on its own WQ data and the WQ data of the nearest upstream sample points (SPu) and WWTPs. To select the SPu and WWTPs a routed river network is used, which can search for available data along the upstream reaches (see 2.2.2). With this network the first SPs upstream of the SPd are selected and the subcatchments between those SPs are listed. From the SPd, SPu and WWTP data, an ES demand indicator is calculated for each of those subcatchments, located between the SPd and SPu. By evaluating each of the sample points is such a manner, an ES demand indicator is calculated for each of the subcatchments within the catchment.

This final demand indicator is based on a combination of indicators that are calculated for each of the N-components. For each N-component, two indicators are derived, based on the concentrations and changes in concentrations between the upstream and downstream sample points (Figure 5-2). For each of the N-components summer mean values are calculated, which are then used to calculate the indicators for one specific year:

- The first indicator: "Ratio Indicator" (I_{ratio}) describes the weight of each N-component compared to the Nt concentration at the SPd. Based on this indicator the relative importance of each component in N_t is defined. This indicator can be calculated based on the components themselves when no SPu is available or the differences in concentration between both SPd and SPu, when available (see 2.2.2).
- The second indicator: "Reach Indicator" (I_{Reach}) gives an indication whether actions should be taking place within the evaluated area or further upstream. Depending on increases or decreases in concentrations, attention will be given to purification along the main river reaches (decreasing trend) or across all evaluated subcatchments (increasing trend). When a WWTP is present within the evaluated area, additional attention can also be given to the area downstream of the WWTP. Depending on the number of upstream SPs, the indicator is calculated differently (see 2.2.2).

A third N-component indicator: "Land use indicator" (I_{LU}) is calculated by taking land uses into account, which allows for a further differentiation between the different subcatchments. This indicator is only calculated with an increasing concentration downstream, when most of the pollution is expected to come from the local subcatchments and not further upstream.

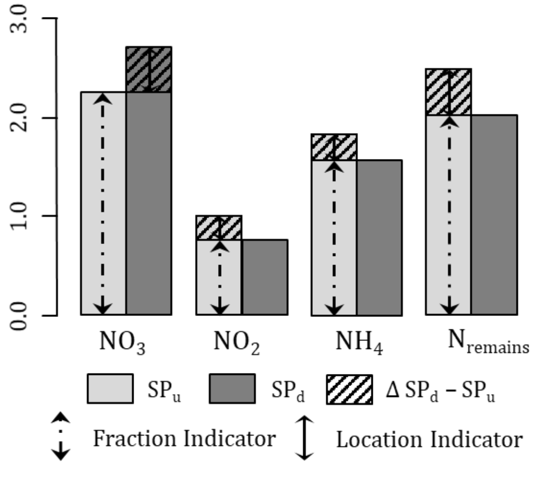


Figure 5-2: Overview of the four N-components and the information that is used in the indicator calculation. SP_d is the downstream sample point under evaluation. SP_u is one or more upstream sample points. NO_3 increases downstream, indicators focus on all subcatchments between both SPs. NO_2, NH4 and $N_{remains}$ decreases between both SPs, therefore focus will shift to the area between both sample points.

These three indicators are then combined with the initial concentrations into component indicators (I_{Comp}) which are added to one final indicator (I_{final}).

$$I_{Comp} = I_{ratio} * I_{Reach} * I_{LU} * \text{concentration N-component}$$
$$I_{Final} = I_{NO3} + I_{NO2} + I_{NH4} + I_{remains}$$

The final indicator of each subcatchment is used to create a ranking of the different subcatchments for ES demand. These indicators and rankings are calculated for different years and combined to gain more robust results (see 2.2.4)

2.2.2 *Stream reach model and upstream trajectory*

Generally, watershed tools evaluate the downstream effects of upstream activities through a range of models, often making use of river network analysis software. White et al. (1992) developed a conceptual model that partitions a watershed in subcatchment units and reaches with a coding system that defines how the reaches are connected to each other. Point and non-points-sources of pollution can be assigned to each subcatchment and the network of reaches can then be used to model the downstream impact of upstream pollution sources. This type of conceptual models has been implemented in various stream network models and watershed management tools (e.g. Benda et al., 2007; Liu et al., 2008; Strager et al., 2009). These models generally evaluate the impact of the different subcatchments on the stream properties such as water quality by calculating the downstream effects of upstream activities. But this type of network models is not capable to search for upstream solutions based on known downstream characteristics. However, the same type of digital network can be used for the development of such upstream analysis. Using such a routed river network, we developed a system that is able to search for, and evaluate upstream reach and subcatchment characteristics in reference to the downstream needs for improvement. In a first step this system evaluates the presence of upstream WQ sample points and upstream WWTPs for each WQ sample point within the Nete catchment (Figure 5-3). Based on the number of SPu and the presence of WWTPs within the evaluated area, a specific set of indicators (I_{Comp}, I_{Reach} and I_{LU}) is used. The details of the calculation methods for each indicator set can be found in Table 5-1.

Depending on the changes in concentration, reaches and subcatchments are attributed with a particular indicator set. When SPd is higher, local activities are considered to have a larger impact on the water quality than activities further upstream. Therefor the location indicator is attributed to each of the subcatchments between the sample points (Figure 5-4).

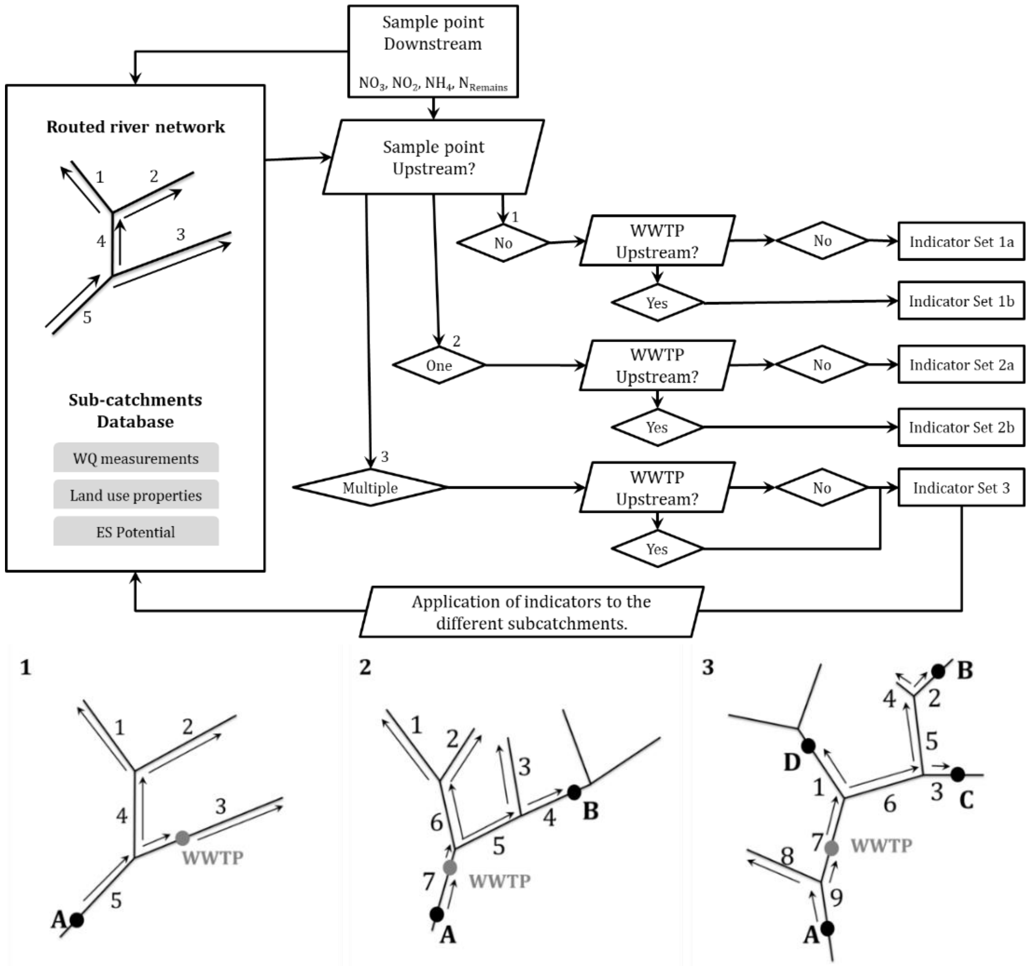
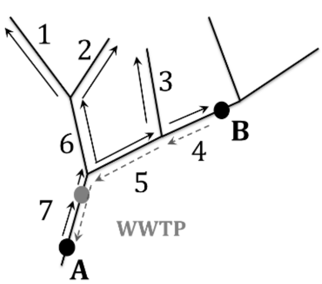


Figure 5-3: Overview of how the indicator sets are selected based on combinations of WQ sample points and WWTPs that can be found upstream.



$A_{NO_3} > B_{NO_3}$	$A_{NH_4} < B_{NH_4}$
7 6 1	4 5 7
2	
5 4	
2	

Figure 5-4: Example of how all subcatchments are evaluated when downstream concentrations are higher than upstream. However, with lower downstream concentrations only the main stream is evaluated

Table 5-1: Overview of the indicator sets that are applied in the analysis.

Indicator set	Indicator type	Condition 1	Condition 2	Formula	Location of application	I_{LU}
Indicator set 1a	I_{Comp}			$SP_{comp} / \Sigma(SP_{NO3}, SP_{NO2}, SP_{NH4}, SP_{KjN\text{-}NH4})$	All subcatchments	Yes
Indicator set 1b	I_{Comp}			$[\Delta\ comp] / \Sigma([\Delta\ NO_3], [\Delta\ NO_2], [\Delta\ NH_4], [\Delta\ KjN\text{-}NH_4])$		
	I_{Reach}	SP < WWTP		$1 + (WWTP_{comp} - SP_{comp} / WWTP_{comp})$	Downstream of WWTP	Yes
				$1 - (WWTP_{comp} - SP_{comp} / WWTP_{comp})$	All other subcatchments	Yes
		WWTP > SP		$1 - (SP_{comp} - WWTP_{comp} / SP_{comp})$	Downstream of WWTP	No
				$1 + (SP_{comp} - WWTP_{comp} / SP_{comp})$	All other subcatchments	Yes
Indicator set 2a	I_{Comp}			$SP_{comp} / \Sigma(SP_{NO3}, SP_{NO2}, SP_{NH4}, SP_{KjN\text{-}NH4})$		
	I_{Reach}	SPu < SPd		$1 + (SPd_{comp} - SPu_{comp} / SPd_{comp})$	Between SPu and SPd	Yes
				$1 - (SPd_{comp} - SPu_{comp} / SPd_{comp})$	All other subcatchments	Yes
		SPd < SPu		$1 + (SPu_{comp} - SPd_{comp} / SPu_{comp})$	Between SPu and SPd	No
				$1 + (SPu_{comp} - SPd_{comp} / SPu_{comp})$	All other subcatchments	Yes
Indicator set 2b	I_{Comp}			$SP_{comp} / \Sigma(SP_{NO3}, SP_{NO2}, SP_{NH4}, SP_{KjN\text{-}NH4})$		
	I_{Reach}	SPu < SPd	WWTP > SPd	$1 + ((WWTP_{comp} - SPu_{comp}) / WWTP_{comp})$	Downstream of WWTP	No
				$1 - ((WWTP_{comp} - SPu_{comp}) / WWTP_{comp})$	All other subcatchments	Yes
			SPu > WWTP > SPd	$1 - ((WWTP_{comp} - SPu_{comp}) / WWTP_{comp})$	Downstream of WWTP	No
				$1 + ((WWTP_{comp} - SPu_{comp}) / WWTP_{comp})$	All other subcatchments	Yes
			WWTP < SPu	1	all	
		SPd < SPu	WWTP > SPu	$1 + ((SPu_{comp} - SPd_{comp}) / WWTP_{comp})$	Upstream of WWTP	No
				$1 + ((WWTP_{comp} - SPd_{comp}) / WWTP_{comp})$	Downstream of WWTP	No
				$1 - ((SPu_{comp} - SPd_{comp}) / WWTP_{comp})$	All other subcatchments	Yes
			SPu > WWTP > SPd	$1 + ((SPu_{comp} - SPd_{comp}) / SPu_{comp})$	Upstream of WWTP	No
				$1 + ((WWTP_{comp} - SPd_{comp}) / SPu_{comp})$	Downstream of WWTP	No
				$1 - ((SPu_{comp} - SPd_{comp}) / SPu_{comp})$	All other subcatchments	Yes

		WWTP < SPu	$1 + ((SPu_{comp} - SPd_{comp}) / SPu_{comp})$	Upstream of WWTP	No
			$1 - ((WWTP_{comp} - SPd_{comp})/SPu_{comp})$	Downstream of WWTP	No
			$1 + ((SPu_{comp} - WWTP_{comp})/SPu_{comp})$	All other subcatchments	Yes
Indicator set 3	I_{Comp}		$SP_{comp} / \Sigma(SP_{NO3}, SP_{NO2}, SP_{NH4}, SP_{KjN-NH4})$		
	I_{Reach}	$SPd > SPu_i$	1	all	Yes
		$SPu_i > SPd > SPu_j$	$1 + ((SPu_i - SPd) / (\Delta SPd – SPu_{mean}))$	downstream SPu_i	No
			1	other sample points	Yes
		$SPu_i > SPd$	$1 + ((SPu_i - SPd) / (\Delta SPd – SPu_{mean}))$	downstream SPu_i	No
			$1 - ((SPu_i - SPd) / (\Delta SPd – SPu_{mean}))$	other sample points	Yes

2.2.3 *Calculation of land use indicator*

A further distinction in the impact subcatchments have on the water quality can be made based on the impact local land uses can have on the water quality. Houses that are not connected to a WWTP affect N-components such as NH4, but also NO_3 and NO_2. While impervious areas can be related to NO_3. However, the impact of agricultural land use types is less clear (Vrebos et al., 2017a; Vrebos et al., 2015). Based on these observations, a land use indicator is used to further differentiate the impact that subcatchments have on the nitrogen concentrations.

There is no linear relationship between the coverage of specific land use types and their impact on the WQ. For example, only a small part of the catchment is covered by houses that are not connected to a WWTP, but their impact on NH_4 can be large. As such, using percentage coverage, as a direct indicator would reduce the impact of not-connected houses and NH_4 in the calculation of the overall indicator, compared to, for example agricultural land. To overcome this, land use percentages are recalculated to a value between 0 and 1, where 1 is assigned to the subcatchment with the highest relative coverage of this land use type from the list of subcatchments under evaluation. The following formulas are used to adapt the indicators sets on a subcatchment level.

NO_2 I_{LU} = A (not-connected households) / max (A (not-connected households))
NO_3 I_{LU} = A (Impervious area) / max (A (impervious area))
NH_4 I_{LU} = A (not-connected households) / max (A (not-connected households))
$N_{Remains}$ I_{LU} = A (not-connected households) / max (A (not-connected households))

2.2.4 *Aggregation of yearly results*

Because of the variability in nitrogen concentrations due to weather and land use changes and changes sample sampling design in between years, indicator values and the ranking of the subcatchments can differ strongly throughout the years. To gain robust results, indicators and rankings are calculated for 10 subsequent years (2007 – 2016). The yearly rankings are then combined into one overall ranking of subcatchments with the highest demand for nitrogen removal. Yearly rankings were combined using a Cross Entropy Monte Carlo rank aggregation with a Kendall's tau distance in which after each iteration 10 samples are retained (R package "TopKLists") (Schimek Michael et al., 2015). More recent years are given an increasing weight within the rank aggregation, to ensure that the most recent results have a higher impact in the final ranking. The year 2007 was given a weight of 1, increasing to 10 for the year 2016. All statistics were implemented under R version 3.4.1 (R Core Team, 2017). This combined ranking was then combined with the results of the ES supply analysis to select subcatchments for ES development.

2.2.5 *Evaluation of demand indicators*

To evaluate the ES-demand calculation and assess the impact of the changing sampling design, calculations were done using both all available data, hereafter called "Full Analysis" as well as a test with only the SPs that were monitored throughout the 10 years, hereafter called "Limited Analysis". The Limited Analysis allowed us to test the performance of the methodology when the number of SP is fixed and only the water quality concentrations changes. To assess whether the same SPs are scored highest in between years, I_{Comp} and I_{Final} are statically compared using the Friedman test (R package "Stats"). To better assess the impact on the individual SPs on the indicator calculation, the changes in ranking were compared to the previous year and plotted in box plots.

2.3 Potential supply in nitrogen storage/removal.

2.3.1 *Ecosystem service models*

To calculate the ecosystem service maps, two models were used that were developed within the ECOPLAN project and are available within a QGIS plug-in (Vrebos et al., 2017b).

2.3.2 *Nitrogen storage*

Soils can store significant amounts of organic carbon and nitrogen stored in SOM. Typically soils with unmanaged, natural vegetation types have larger carbon and nitrogen stocks than anthropogenic land uses. Also soil hydrology plays a critical role in the creation of these stocks. Nitrogen stocks were calculated, using formulas to estimate soil organic carbon (SOC) stocks and conversion C/N ratios. Four SOC formulas describe the potential storage for land cover types and land use: grassland, forest, arable land and natural vegetation types in Flanders (Ottoy et al., 2015; Ottoy et al., 2017a; Ottoy et al., 2016; Ottoy et al., 2017b). These formulas take into account both geophysical (e.g. soil texture) and land cover characteristics and allow for the spatial explicit calculation of SOC stocks in Flanders. To calculate yearly carbon uptake into the soil, we assume that it takes approximately a hundred year to get to a new SOC stock equilibrium (Foereid et al., 2004; Freibauer et al., 2004). The SOC map was then converted to a N-storage map using different C/N ratios (Table 5-2). Higher C/N ratios in the SOC can be explained by litter production that is more difficult to decompose (high C/N, high lignin concentration).

Table 5-2: C/N ratio of SOC for several vegetation types

Vegetation type/ land-use	Upper and lower estimates	Mean value
Cropland and production grassland	8-12	10
Floristic and species rich grasslands	10-14	12
Broad leaf forest	15-25	20
Mixed forest	20-25	22
Coniferous forest	25-30	27
Heathland	25-35	30
Phragmites wetlands	25-35	30
Wetlands (sedges and tall herbs)	15-25	20
Eutrophic alluvial forests	15-20	17
Mesotrophic wetland forest	20-25	22
Oligotrophic wetland forest	25-30	27
Peat bogs	25-35	30

2.3.3 *Denitrification*

Under conditions of (temporal) waterlogging, bacterial processes enable to remove nitrogen from ground and surface water. Denitrification is a microbial facilitated process where nitrate is reduced and ultimately produces molecular nitrogen (N_2) through a series of intermediate gaseous nitrogen oxide products. Denitrification usually occurs under conditions of (temporal) waterlogging. To spatially assess denitrification, nitrate concentration in the groundwater, residence time and denitrification potential were taken into account.

Nitrate concentrations in the groundwater are calculated by estimating landscape level N-leaching to the groundwater. Therefore, atmospherical and agricultural N-deposition are combined with N-uptake by vegetation and the soil vulnerability for N-leaching, to estimate the N-load that leaches in to the groundwater. In combination with long-term average annual rain surplus, which recharges the groundwater with 0,25 m^3/m^2, an approximation is made of the average N-concentration in the subsurface groundwater.

Based on maps of mean high and low groundwater levels, the potential degree (%) of N removal through denitrification is calculated. Soils that are more than 60% water saturated have potential for denitrification. The N-removal efficiency is estimated based on the formula of Seitzinger et al. (2006). A multiple scales topographic position index is used to identify infiltration-seepage patterns (Jenness, 2006). The residence time of the groundwater is based on seepage intensity estimates that have been corrected for soil permeability. The soil permeability, which depends on the soil grain size, is based on the soil map of Flanders (Dondeyne et al., 2014). By combining these factors, a final map is produced that estimates denitrification in the Nete catchment.

2.4 Current and potential supply

To assess the potential for additional ecosystem service supply, current and potential delivery needs to be compared. Current and potential scenarios were developed and compared to each other to identify zones for improvement. For both N storage and denitrification, current supply is calculated using maps of current land use and soil hydrological conditions. Potential supply of N storage was calculated by restoring the natural vegetation that occurs within the region, an increase of the groundwater table by 10cm or 20cm and a restoration of the peat layers in the natural floodplains. For denitrification an increase in water table of 10cm or 20cm is combined with current land cover to keep the nitrogen input in the calculation the same.

To map the potential as an indicator map, current and potential supply are combined (Additional supply = potential supply - current supply / current supply). This generates an indicator map where negative values point to decreases in ES supply and low opportunities for ES development, while positive values point to potential increases and opportunities.

Both N storage and denitrification were aggregated on a subcatchment level and a ranking of the subcatchments was calculated based on the combined yearly N removal potential. This ranking was calculated for the 10cm and 20cm change in groundwater level.

2.5 Combining supply and demand.

Both rankings of demand and ES potential are combined in to a scatterplot to access which subcatchments can help to improve water quality where needed. These scatterplots are combined with a suitability classification for ES restoration: High suitability (>= 90% of both rankings), Medium (< 90 % of one ranking and >= 75% of both rankings), Low (< 75 % of one ranking and >= 50% of both rankings) and Very Low (< 50% of one ranking).

2.6 Application of the methodology

2.6.1 *Case Study*

To test the developed methodology, an evaluation of the Nete catchment, Belgium, was done over a 10-year period. The Nete catchment (approximately 1.673 km²) is situated in the central Campine region in Northern Belgium (Figure 5-5). It has a marine, temperate climate with an average precipitation of 800 mm/year. The dominant soil type is sand, with loamy sand occurring in the floodplains and a small area in the south that mainly consists of sandy loam and loamy sand. Topographic height ranges between 3 m and 82 m above sea level.

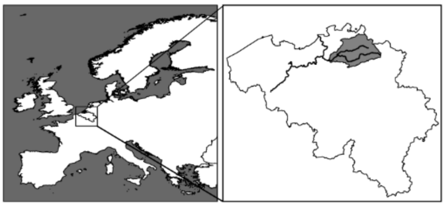


Figure 5-5: Location of the study area.

The Nete river yearly discharges on average 389 million m³ water into the Rupel. Water chemistry is highly variable, both spatial and temporal, because of anthropogenic activities like agriculture, households, industry, etc. Land use in the Nete catchment consists mainly of cropland (20%), pasture (22%), broadleaved woodland and evergreen needle leaf forests (23%) and impervious area (8%). The remaining 27% mainly consists of open water, gardens, bare land, etc. The land cover is highly fragmented with a median parcel size of 0.10 ha.

In total 55 cities and municipalities are fully or partially located within the catchment. Population densities for these ranged in 2010 between 152 inh/km² and 1530 inh/km² with an average of 420 inh/km² for the entire catchment. The urbanized areas are mainly situated around the town centers. Typical ribbon development is present in between the villages, making the development of sewer systems costly and time consuming (De Decker, 2011).

At the moment 30 WWTPs are situated within the catchment. In 2015 82% of the households was connected to a WWTP in which 78% of the received pollution load was removed. Only during the 90's and the beginning of the 21st century most of these WWTPs have been expanded with tertiary treatment systems. Wastewater treatment zones do not coincide with the natural catchments and wastewater is actively transported from one (sub) catchment to another for treatment. Although the Nete catchment is considered to be one of the most natural catchments within Flanders (northern part of Belgium), almost none of the rivers and streams within the catchment achieve the European Water Framework Directive standards and more investments to reach these standards are needed. Over a period of 10 years we large number of sample points did not meet the Nt water quality (Figure 5-6), although in the most recent years this number has decreased (Figure 1-6). A major part of the streams within the catchment have been straightened, deepened and embanked. As a result, these streams no longer follow their natural flow path. At the same time large obstructions like canals and high ways have been constructed. These constructions are bypassed by siphons, which allow water to run under obstructions in the ground.

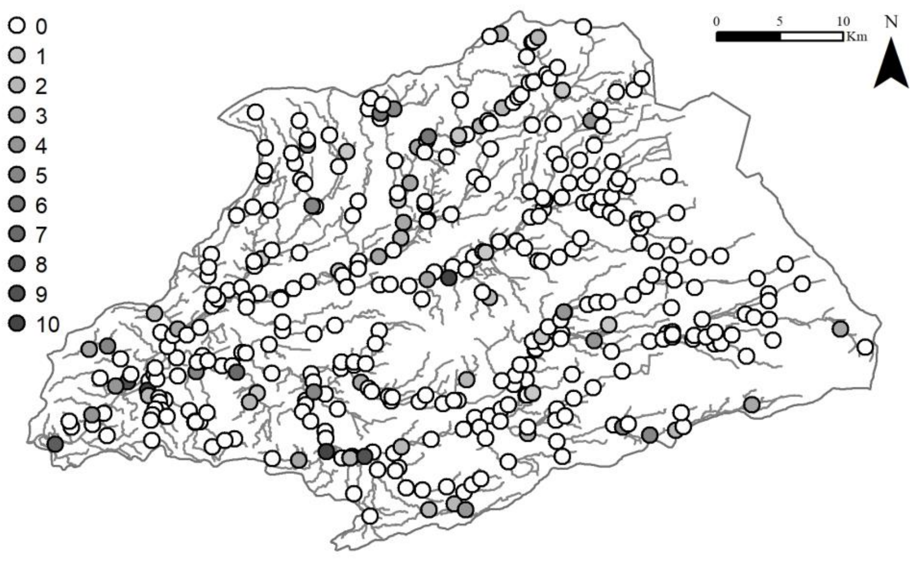


Figure 5-6: Overview of the sample points which were monitored at least ones between 2007 and 2016 and the number of times the Nt water quality standard was not met.

2.6.2 *Datasets*

Water quality data, from 2007 until the end of 2016, was obtained from the Flemish Environmental Agency (FEA). During this period, total nitrogen (Nt, mg/L), nitrate (NO_3^--N, mg/l), nitrite (NO_2^--N, mg/l), ammonium (NH_4^+-N, mg/l) and Kjeldahl nitrogen (KjN) concentrations were measured at 326 locations in the catchment. Sampling frequency differed between the sample locations. Some sample points were sampled monthly, others only once a year; some points were sample each year, others only one year.

For NO_3 18330 measurements were available for analysis, NO_2 15493 measurements, NH_4 15342 measurements, Kjeldahl nitrogen 14630 measurements and for total nitrogen 14819 samples. KjN was used to calculate the N_{remain} (KjN – NH_4). There was a decreasing trend in the number of samples that were available each year. For example, in 2007 222 sample points were sampled with in total 2261 NO_3 measurements, in 2016 this had decreased to 192 sample points and 1564 samples. Information regarding the WWTP and their effluent discharges was also obtained from the FEA. Effluent discharge information was available for 29 out of the 30 WWTP. In total 7604 NO_3^--N, 7606 NO_2^--N, 7606 NH_4^+-N, 7603 KjN and 7522 total N measurements were available for the WWTPs.

To calculate the ecosystem service maps, different datasets and models were used that were developed within the ECOPLAN project and are available within a QGIS plug-in (Vrebos et al., 2017b).

2.6.3 *Catchment delineation and routed river network development.*

The routed river network developed for the Nete catchment comprises out of 954 reaches and subcatchments. The river network is based on the Flemish hydrological atlas (FEA, 2005), which was simplified to limit the number of reaches by removing the smallest ditches. Subcatchments were delineated for each river reach from a 1:5000 digital elevation model (DEM) expressed as a 5m-raster (FEA, 2006) using a D8-runoff model (Jenson et al., 1988). The DEM was modified by lowering the elevation values based on mapped stream channels and siphons and by elevating values based on mapped dikes (NGI, 2007). This allowed us to force flow-direction maps to match existing streams. Natural subcatchments (n = 954) were calculated using the hydrology tools in ArcGIS 9.3. Areas that drain into the artificial water navigation canal system were removed from the analysis. Size of the subcatchments ranges between water quality sample points and discharge locations of the different WWTP were attributed to the correct river reaches.

Land use indicators were calculated following the procedure described in Vrebos et al. (2015), taking the effect of the sewer infrastructure on the upstream delineation into account. Land use maps (1:10.000 vector-layers) were obtained from the National Geographic Institute (NGI) and consist of 49 categories (NGI, 2007) The land use vector layer was converted to a 1m-raster and the land use categories were initially aggregated to 8 classes: woodland, cropland, pasture, buildings, paved area, water, greenhouses and others. A distinction was made between buildings (area buildings) and other impervious areas (roads, concrete areas, etc.) because buildings can be an important source of wastewater, while other impervious areas mainly collect rainwater. All GIS-calculations were performed in ArcGIS 9.3. Where needed areas connected to a sewer system were designated to the subcatchments the sewer system drains into.

3. RESULTS

3.1 Evaluation nitrogen concentrations

3.1.1 *Example of results*

For each of the subcatchments I_{final} is calculated for each year, using the available concentration data and indicator sets. As an example of the analysis, the calculation in 2008 is given for a cluster of subcatchments located between two SPs. Concentrations between the SPu and SPd increase for NO_2 (0.076 mg N/L to 0.106 mg N/L), NO_3 (1.106 mg N/L – 1.272 mg N/L) and N_{Remain} (1.362mg N/L 1.438 mg N/L) and decrease for NH_4 (1.030 mg N/L – 0.52 mg N/L).

As NO_3 concentration increases, I_{reach} for NO3 are the same for all subcatchments and equal to 1 (Figure 5-7A). The same I_{reach} is obtained for NO_2 and N_{Remain}. Because of the decrease in concentration, the I_{reach} for NH_4 focusses on N removal along the reaches between both SPs (Figure 5-7B). As the number of subcatchments along the trajectory is the same as the number of tributary subcatchments, I_{Reach} is equal to 2.

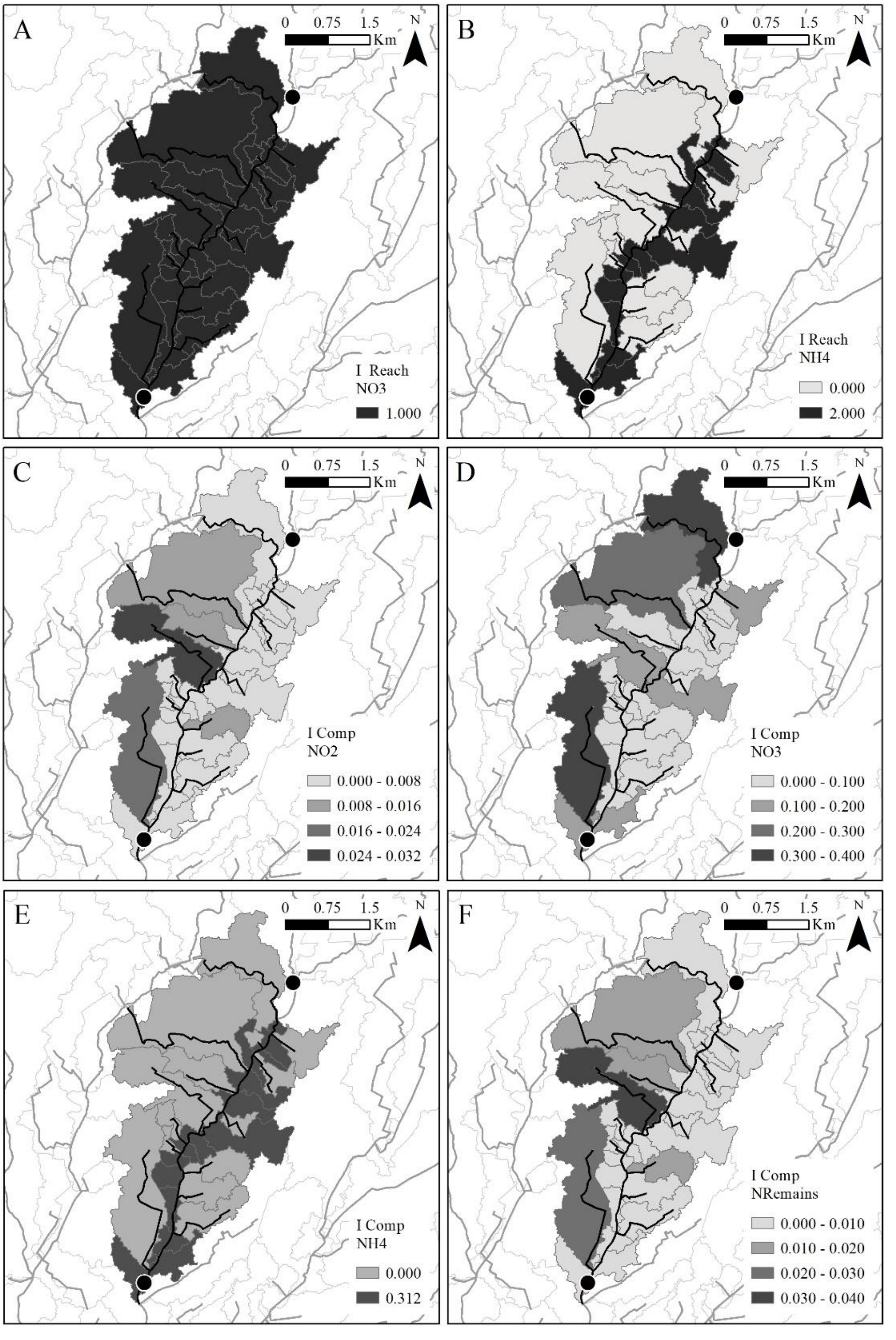


Figure 5-7: Overview of some of the intermediate results for one area under consideration in 2008. Analysis for these catchments is based on 2 SP and calculated with indicator set 2A. A) Reach indicator for NO_3, B) Reach indicator for N_{Remain} C) I_{Comp} for NO_2. D) I_{Comp} for NO_3. E) I_{Comp} for NH_4. F) I_{Comp} for $N_{remains}$.

Further differentiation in I_{Comp} is made by the LU indicator for NO_2, NO_3 and N_{Remain} and the I_{ratio} of the different components (Figure 5-7C, D, and F). Spatial distribution of I_{Comp} for NO_2 and N_{Remain} is the same as both have an I_{Reach} of 1 for all subcatchments and make use of the same I_{LU}, only the I_{ratio} of both N-components differ (Figure 5-7C and F). NO_3 makes use of another I_{LU} resulting in a different I_{Comp} distribution (Figure 5-7D). I_{Reach} to I_{Comp} for NH_4 is only changed by the I_{ratio} (Figure 5-7E). The I_{final} gives the combined effect of the N-components taking the concentrations at the downstream SP into account, giving a broad differentiation in indicator values between the different subcatchments. (Figure 5-8).

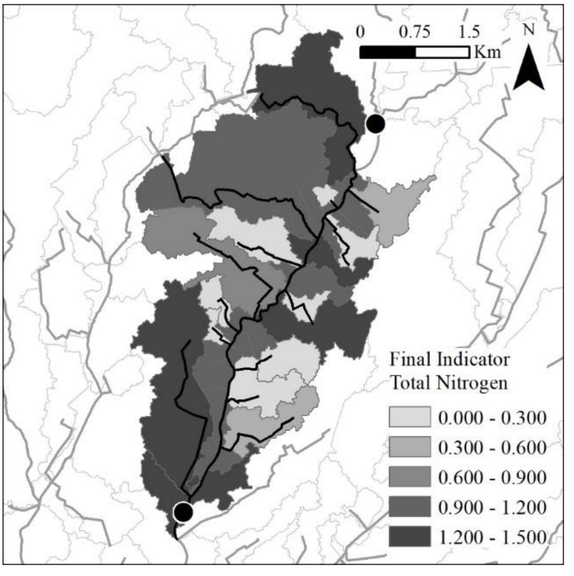


Figure 5-8: Final indicator combining the results of Figure 8 (C-F) and the nitrogen concentration of the downstream SP.

3.1.2 *Indicator sets*

The indicator sets were applied to the catchment for 10 subsequent years. Changes in the sampling network throughout the years affect the calculation method of the final indicators, as the application of the indicator sets depends on the number of sample points available and their location. Of the 954 subcatchments available within the network, only 454 have the same downstream sample point throughout the years. Some of the subcatchments (n = 25) are evaluated by 5 different downstream sample points over these 10 years. Selection of upstream sample points and WWTPs for indicator calculation are affected in a similar fashion. As a result, the information and calculation method used to evaluate a subcatchment can differ strongly throughout the years. Due to the recent decrease in available sample points, the number of downstream sample points that are evaluated using indicator set 2 has diminished significantly (Figure 5-9A).

Of the 326 SPs within the catchment, 164 were at least used once within the evaluation and 59 were used each year (Figure 5-9B). Most of those 59 SPs are located along the main tributaries of the Nete river. WQ data for the upper reaches and headwaters are generally only available for a few years, making a detailed evaluation of these areas irregular.

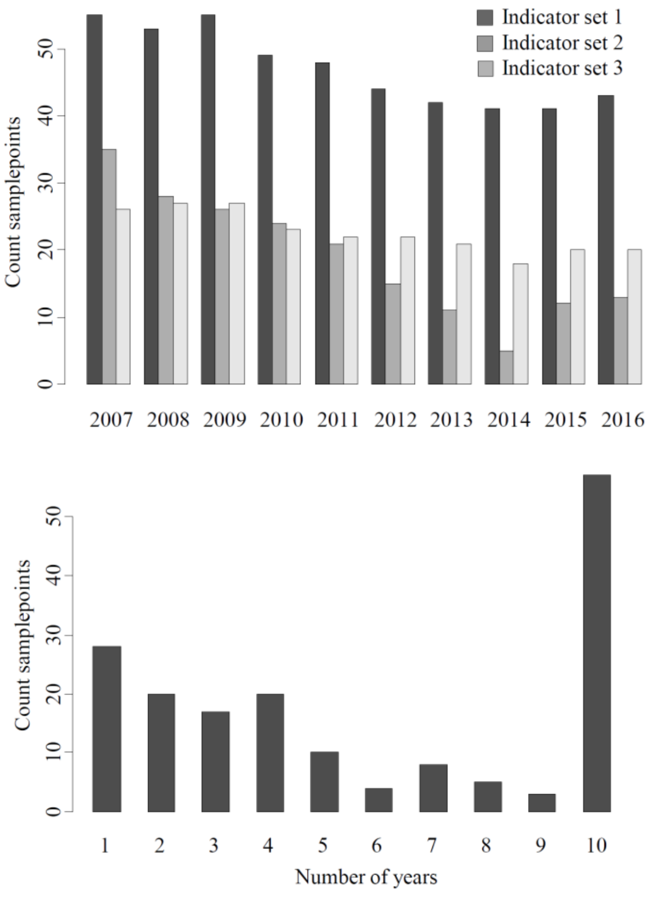


Figure 5-9: A) Overview of the number of times the different calculation types were used each year. B) Number of times each sample point (n = 172) is used in the analysis.

3.1.3 *Comparison Full Analysis – Limited Analysis.*

Changes in sampling designs and in water quality between years can affect the calculation of the indicators. To assess this effect rankings and changes in ranking were evaluated for both the full analysis and limited analysis. Friedman test for both analyses were found significant ($p < 0.005$) for all I_{Comp} and the I_{Total}, indicating that overall SPs are consistently ranked higher or lower throughout the years.

However, large changes in ranking do occur for some SPs between years. Ranking of SPs change in between years for both the full and limited analysis (Figure 5-10A and 9B). With many of the SPs changing at least once strongly in ranking from one year to another. Changes in ranking decrease in more recent years for the full analysis, while these changes remain more stable for the limited analysis. When comparing both analysis to each other, ranking of the subcatchments differ strongly (Figure 5-10C).

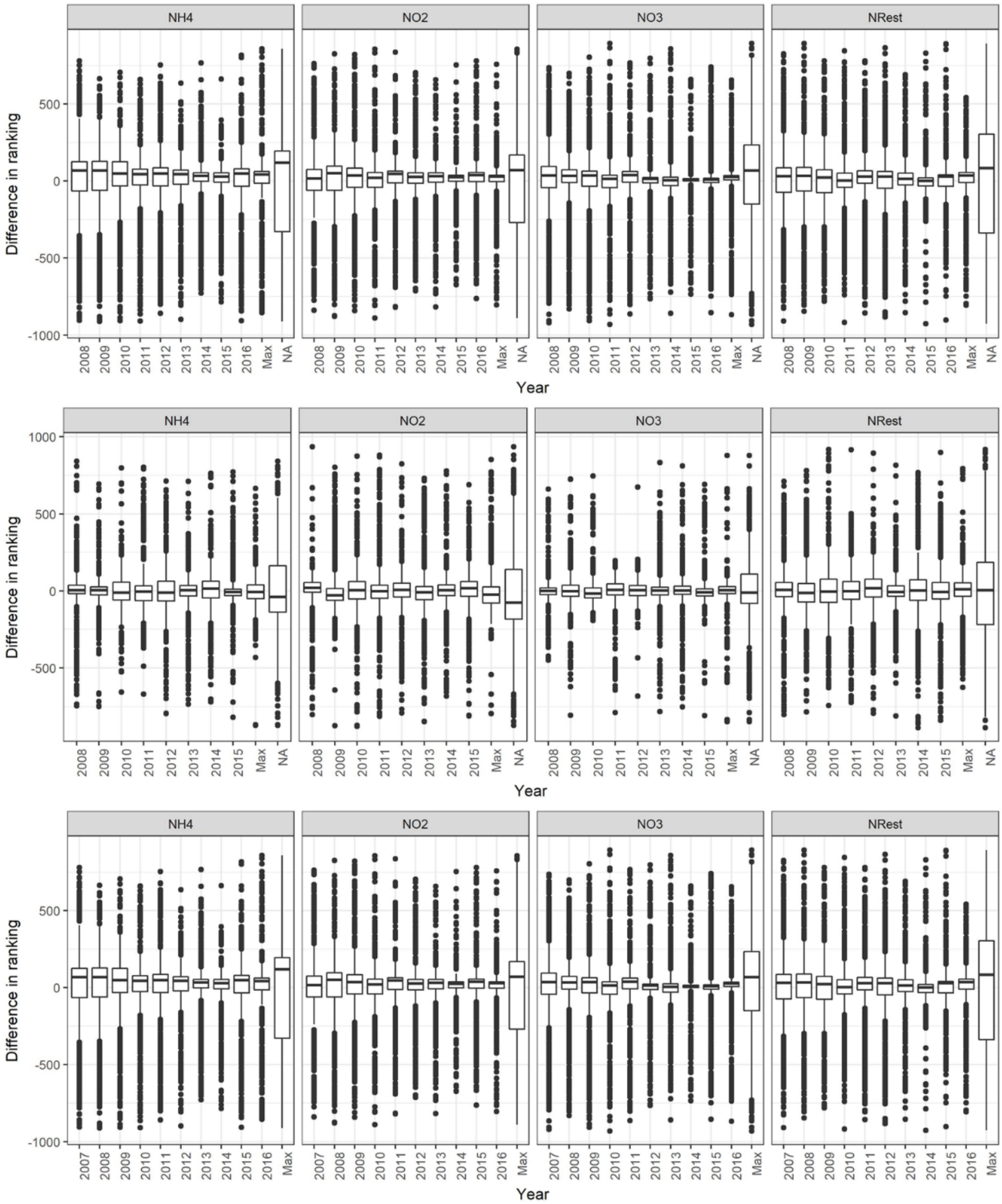


Figure 5-10: Boxplots of the changes in ranking for the different indicators. A) changes compared to the ranking of the previous year when all available SP are used, B) changes compared to the ranking of the previous year when all available SP are used for the 57 SPs that are available throughout the years C) changes in ranking comparing both methods with all SP and the selection of 57 SPs. Max. gives the highest change in ranking for each SP over the 10 years. Boxes present median values, 25th and 75th percentile (interquartile range). Whiskers on the boxes give the largest values within 1.5 times interquartile range above 75th percentile range and the smallest value within 1.5 times interquartile range below the 75th percentile range.

3.2 ES Supply maps

Two ecosystem services were calculated, which can decrease total nitrogen in the river reaches. Nitrogen storage in the soil can both increase (up to 32.72 ton/ha*year) as well as decrease (up to 0.54 ton/ha*year) compared to the current situation. Decreases in storage are located in agricultural land which have a high carbon and nitrogen storage due to manure input. Converting them into specific forest types results in reduced input and (short term) storage. Denitrification can range between 0 and 0.397 ton/ha*year (10 cm increase) or 1.188 ton/ha*year (20cm increase). While changes in nitrogen storage can take place across the catchment, denitrification is limited to areas with higher groundwater tables, often located along the reaches (Figure 5-11).

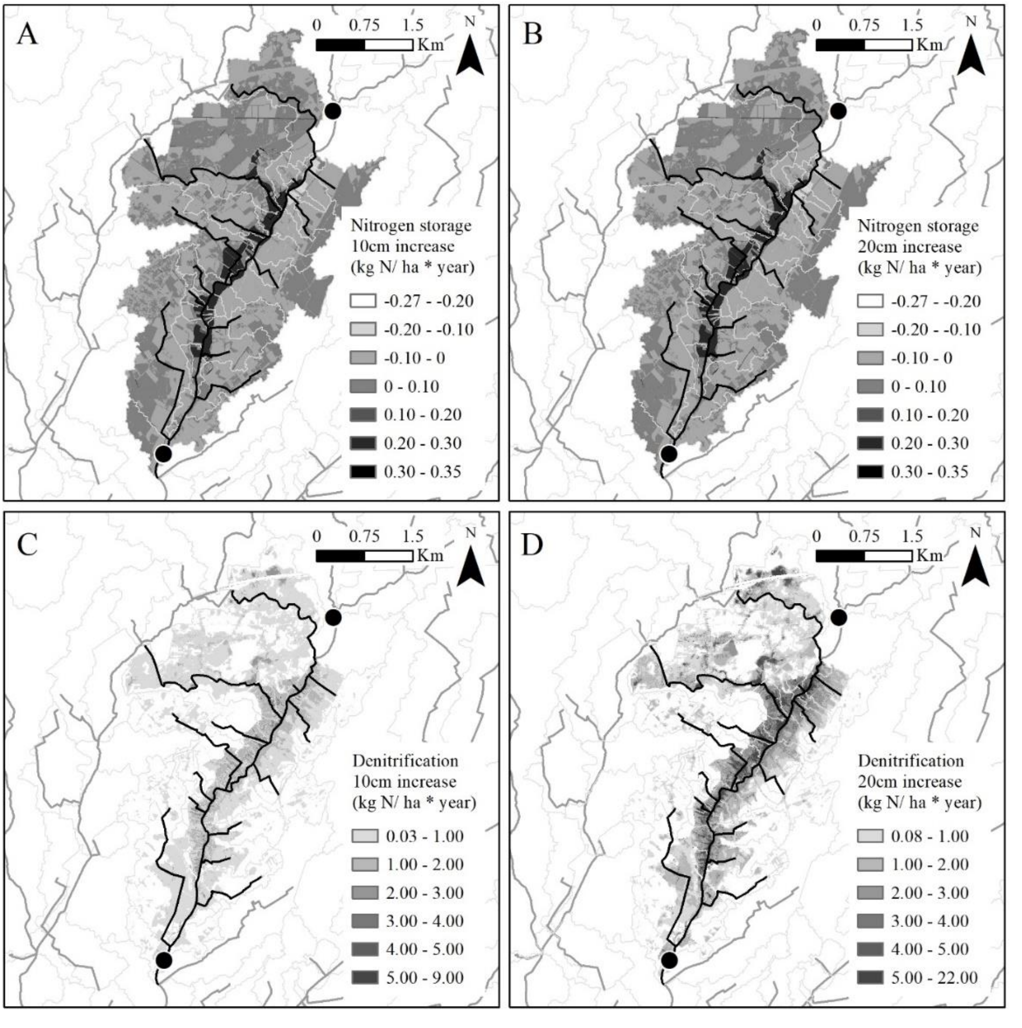


Figure 5-11: Detail of the changes in ecosystem service delivery calculated for the land cover and groundwater scenarios compared to the current situation. Changes in nitrogen storage under natural vegetation C) Denitrification under current land use and an increase of groundwater by A) 10cm. ad B) 20cm compared to the current situation (kg N/ha * year). Color legends of denitrification are non-linear to better illustrate the spatial variation.

Increasing the groundwater level with 20cm instead of 10cm has a non-linear effect throughout the catchment. Some of the subcatchments are effected more strongly, then others. As a result, rankings of the subcatchments based on a 10cm or 20 cm increase of the groundwater level differ from each other. With some subcatchments becoming more suitable compared to the rest of the subcatchments then others.

3.3 Combining supply and demand

In order to select subcatchments with a high demand for nitrogen removal and opportunities for ES development, scatterplots were made of both the demand and supply rankings (Figure 5-12). No trend was available between both rankings. Subcatchments with a high demand for nitrogen removal can have a very high potential for ES development or a very low potential. Combining both rankings results in a classification of the subcatchment with 4 suitability classes. The subcatchments with a high and medium suitability are located throughout the catchment and are located along both the headwaters as well as the lower reaches (Figure 5-13).

Because of the difference in ranking between both 10cm and 20cm groundwater increase, the suitability ranking also differs. For example, under a 10cm increase of the groundwater level, 13 subcatchments are located within the 90-percentile of both the demand as well as the supply ranking. With an increase of 20cm of the groundwater level 9 subcatchments are located within both 90-percentiles. Most of the subcatchments that have a medium to very high classification are located along the river continuum. Although some headwaters are selected as well.

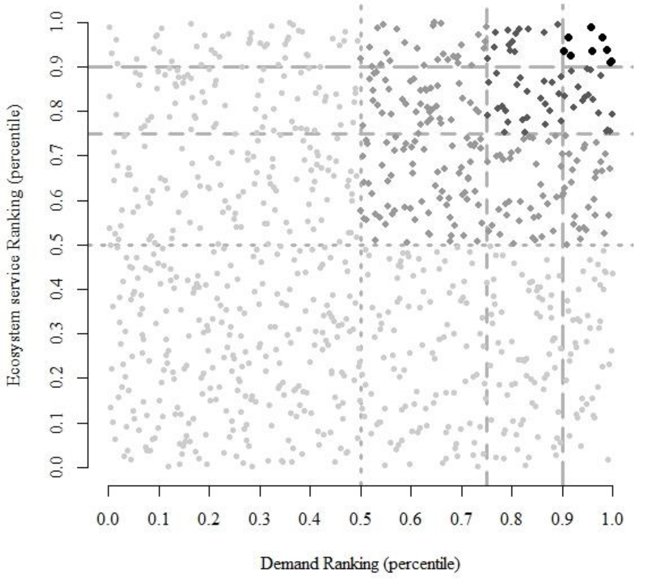


Figure 5-12: Scatterplot of the WQ demand and ecosystem service supply ranking (in percentile: 0 = low: 1 = high) with a groundwater increase of 20cm. Subcatchments located within the 90-percentile range of both rankings provide the greatest opportunities for ecosystem service restoration. Colors within the graph indicate the classification of suitability (Very High = Black, High = Dark grey, Medium = Grey and Low = Light grey) used in Figure 5-13.

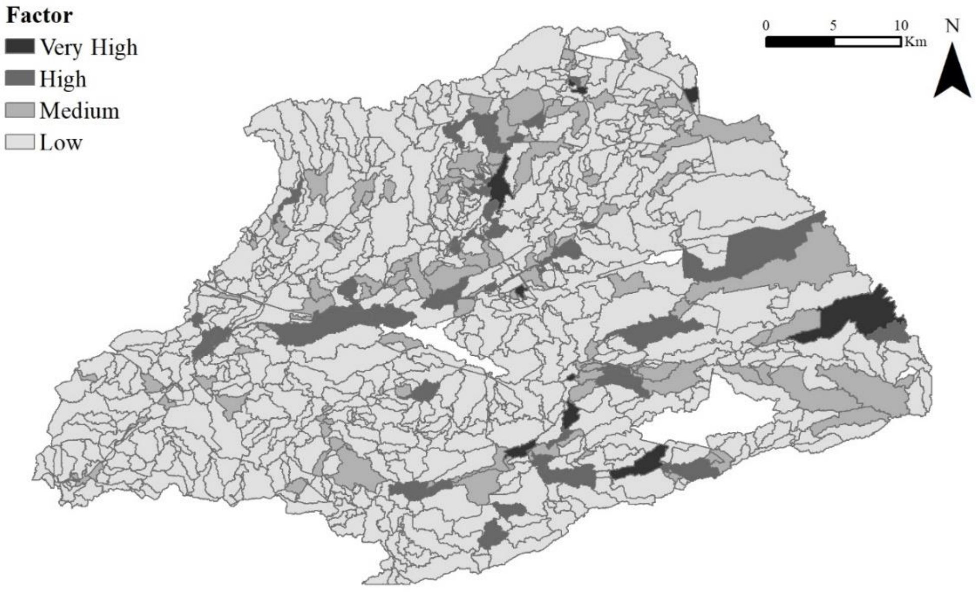


Figure 5-13: Location of the subcatchments and their suitability for ES development. Subcatchments with a classification of "Very High", have a high demand for nitrogen removal and potential for ES development. The classification of the subcatchments is based on the scatterplot classification in Figure 5-12.

4. DISCUSSION

Within this study, a methodology was developed to evaluate ES demand and supply for water quality management within a catchment context. It specifically aims to integrate upstream-downstream flows in the analysis, moving further in the full implementation of the ICM aims. It allows for the selection of subcatchments that have a high potential for nitrogen removal through rewetting on locations where water quality improvement is needed. By taking into account changes in upstream and downstream concentrations of the four N components, a detailed assessment can be made of different pollution sources and their combined effect on the total nitrogen concentration. The presented indicator calculation allows an analysis of the WQ on a larger scale then possible with more complex models.

To develop and apply the presented methodology large amounts of data are needed with sufficient temporal and spatial detail. Although the Nete catchment is a well-studied catchment, the methodology still struggles with a relative lack of data, making it difficult to calculate reliable indicators for each of the subcatchments. The accuracy to which the indicators can select the subcatchments with the highest potential is difficult to assess. However, our analysis over a longer period of time shows that the indicator values of many of the subcatchments can differ significantly between years. To better access the reliability of the subcatchment selection a cross-validation on a smaller scale with for example a SWAT model, might be helpful.

The presented methodology makes use of the actual measured water quality data and assesses the impact of local and upstream activities along the river continuum. As with most WQ standards, total nitrogen (N_t), is measured and evaluated in concentrations. Although concentrations are relevant in a monitoring context, they are difficult to use in a nitrogen balance analysis. Ideally upstream – downstream analysis on a catchment level integrates WQ concentrations and river flow data in to one evaluation of pollution loads as mass balances, as for example can be done in SWAT models (Chen et al., 2014; Cools et al., 2011). This allows to access more complex processes, such as in-stream conversions and removals. but requires accurate flow data (Jin et al., 2016). However, measuring or modelling water flows at a scale, needed for a detailed analysis at catchment level appears to be impossible due to the spatial and temporal complexity of the river system in Flanders and limitations of available hydrological models. Current hydrological models can only be applied to parts of the Nete catchment and to a limited number of streams (e.g. Vansteenkiste et al., 2013; Vrebos et al., 2014). Even on this smaller scale, hydrological models struggle to provide reliable flow data across a catchment (Vrebos et al., 2014).

Our methodology tries to compensate for this lack in hydrological data by using only available concentration measurements. As such, our assessment incorporates as good as possible the actual state of the catchment with all its (recorded) changes, even over small scales. Previous studies have shown how WQ can change in the Nete catchment over a range of distances (Vrebos et al., 2017a). As models depend on input data regarding, they can only model impacts of well-known, if possible monitored, pollution sources and processes. Especially on larger scales models cannot incorporate all processes and pollution sources and take local variability into account. For example, local agricultural practices, specific overflows and illegal discharges are difficult, if not impossible to incorporate in a large scale model. This make them less reliable in predicting changes due to unexpected events.

However, the spatial complexity of the catchments river system, the variability in available monitoring data and the lack of suitable flow data made the design of reliable, understandable indicators challenging. To make the analysis throughout the catchment possible, different types of indicator sets had to be developed, depending on the location of the SP and the amount of available upstream data. These indicator sets make use of a differing number of formulas and data. Ensuring that the information that is contained within one indicator set does not differ from the others and a specific value in each indicator has the same meaning, posed a problem. Although these indicator sets are designed to give the same information within the same indicator boundaries, some differences between the indicator sets remain.

The current calculation of the indicators does not reflect all processes available within the river system. For example, it partly neglects the impact of instream processes during which N-components are converted to one another. Dominant instream processes can partly be assessed by decreases between upstream and downstream SPs. However, these decreases can also be due to dilution with less polluted water from other reaches or groundwater seepage. In these cases, the indicator set will promote a further increase of the in stream processes between both sample points. When the pollution source is located on adjacent subcatchments and instream processes significantly impact the concentrations, high indicators values might be assigned to the wrong subcatchments.

The data, used in this study, are primarily collected for water quality monitoring. Every year the FEA changes part of its monitoring network to get a better overview of the spatial variation in WQ within the catchment. However, this results in a yearly, changing list of sample points available for evaluation. As a result, the same subcatchment can be evaluated from a different downstream SP on or with changing upstream SPs and another indicator set. Our results suggest that these changes in monitoring design and the decrease in available SPs affect the results of the evaluation and the yearly ranking of the subcatchments. At the same time, changes in climatic conditions between years impacts the ranking throughout the years. The opposition between a reduction in reliability due to a lower number of SPs and the up-to-date information of the most recent sampling years, makes it difficult to access the reliability of the methodology. Therefore, the evaluation was done over a large 10-year period and a final ranking was statistically obtained. By combining the different results and giving additional weight to the most recent years, a ranking was created that better reflects the overall conditions of the current subcatchments. But whether this time period and the statistical analysis can fully compensate for the observed variability is unclear. To make further use of these monitoring networks within such analysis, more research is needed on the impact of the sampling locations on the overall results.

To estimate the potential ES delivery, different groundwater and land use scenarios were implemented in two ES models. Both the models as the scenarios are a simplification of reality and do not reflect the actual in N-storage and denitrification that can take place. For example, an overall increase in groundwater depth is not realistic and in reality groundwater depths will change more variable within a subcatchment when the local hydrology is altered. However, the results of these scenarios give an indication of the potential of the different subcatchments. Subcatchments that are ranked high regarding storage and denitrification will in reality be more suitable for these services. The results can therefore be used as suitable indicators within this analysis.

Although we are able to select subcatchments for ES development, it is unclear to what extent ES development within a subcatchment can impact the local water quality. Translating the yearly removed N, due to storage and denitrification, into a decrease in N concentration within the reach, requires more local information regarding flow regime and N loads. To understand how the selected sites, need to be designed for optimal ES restoration and how much area is needed, requires a specific study for each of the selected subcatchments with specific attention for the flow interactions and the actual current ES delivery. The used scenarios for the ES calculation are fairly simple, but do reflect potential changes within the catchment. Further elaborated scenarios, with for example modelled groundwater tables, might affect the overall ranking of the subcatchments and make the results more reliable.

Considering the time and effort that is needed to apply the methodology, the information that can be derived from the analysis is restricted and its reliability uncertain. As such the applicability of the methodology in the actual decision process of ICM appears to be limited. In a catchment with fragmented land use and many potential pollution sources, an overall catchment analysis of the supply and demand for N-removal appears to be almost impossible. Incorporating upstream-downstream calculations in the methodology complicates the analysis to a point where results become difficult to interpret.

A complete ICM assessment is also more complex then only water quality management and encompasses the management of flow dynamics and many water related activities within a catchment setting. The presented methodology is limited to only one part of the overall water quality management. To further expand these upstream-downstream analyses to other catchment properties and activities, more research is needed regarding the movement of indicators along the river continuum and how these indicators can be combined within a branched network, such as a catchment. But considering the reliability of the current methodology it is doubtful if such a system can be applied for a complete ICM assessment. Whether the required data to develop these systems is available at a sufficient spatial detail remains also an open question. The implementation in the Nete catchment illustrates the relative potential of such analysis, as well as the many remaining challenges to develop a robust, reliable methodology.

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Chapter 6 - Mapping ecosystem service flows with land cover scoring maps for data-scarce regions.

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Abstract. Natural resource management requires spatially explicit tools to assess the current state of landscapes, to analyse trends and to develop suitable management strategies and interventions. The concept of ecosystem services can help in understanding the importance of natural resources for different stakeholders and at different spatial and temporal scales. Simple methods to map ecosystem services using scoring of land cover types are particularly useful in data scarce regions, but do not reflect the dynamics of supply and demand. Within this study, GIS scripts were developed to represent and assess several different modes of ecosystem service flows between supply and demand, using ecosystem services scoring tables. By integrating the flows, the ecosystem services can be better evaluated. The outcomes do not give quantitative information on whether supply meets demand, but indicate the spatial distributions of both supply and delivery and where ecosystem services are under threat because of changes in ecosystem or flow mechanisms. The scripts allow us to identify sites that are vulnerable to ecosystem service loss and to evaluate possible management scenarios.

1. INTRODUCTION

Africa and other developing regions are often highly dependent of natural resources for livelihoods and development (Reardon et al., 1995). The degradation and decline of ecosystems and related ecosystem services (the benefits people obtain from ecosystems) can have a large impact on local livelihoods especially of the poor. In the long term this degradation can threaten sustainable development (Scherr, 2000). Integrated natural resource management (INRM) aims to provide a management framework for sustainable use of natural resources and ecosystem services and to prevent further degradation. One requirement for INRM to work in practice is the availability of tools for spatial analysis to map and understand the spatial relationships between ecosystems and the socio-economic system (Frost et al., 2006). Spatial distribution of natural resources and major processes need to be analysed at an appropriate scale (Lovell et al., 2002), to allow researchers, managers and authorities to identify opportunities for and threats to sustainable use, and to define spatially differentiated management plans.

Ecosystem services have gained much attention in recent years (de Groot et al., 2012; MEA, 2005; TEEB, 2010) and are often considered to be helpful in natural resource management (e.g. Tallis et al., 2009; Wainger et al., 2010). The concept can bridge the gap between research and management (Sitas et al., 2014) and provide a common language for managers, researchers and different stakeholders (Granek et al., 2010). But its place within management as a whole (Norgaard, 2010) and specific management and impact assessment frameworks is still disputed (Baker et al., 2013; Cook et al., 2012). Where and how ecosystem services can be integrated in natural resource management remains therefore a topic of discussion.

Operationalization of the ecosystem services framework requires different aspects and dimensions to be taken into account (Seppelt et al., 2011; Seppelt et al., 2012). One of these aspects is how ecosystem service supply and demand can be separated in time and space: ecosystem services are often used at different locations and scales from where they are produced (Luck et al., 2009). Different types of flow mechanisms can deliver the service to the demanding areas. These flows can be mediated by both natural (e.g. water flow) and human induced processes (e.g. movement of people.). The flow mechanism can differ between ecosystem services, the ecosystems considered and the spatial dimension of the analysis (Blaschke, 2006). Assessment of ecosystem service flows is sporadic and usually remains conceptual (Serna-Chavez et al., 2014). Therefore, ecosystem service flows are currently difficult to incorporate in ecosystem service assessments.

In recent years a large variety of mapping methods and models has been developed within the ecosystem services framework that can be used in parts of the INRM framework (e.g. Johnson et al., 2010; Nelson et al., 2009) e.g. to improve stakeholder interaction (Brown et al., 2012) or for scenario analysis (Wang et al., 2009). Each of these mapping methods and models was developed with a specific goal in mind, incorporating different parts of the ecosystem services concept and addressing different levels of system as well as methodologic complexity (Pagella et al., 2014). Methodologies vary from simple land use/land cover based assessments (Burkhard et al., 2009) to highly complex systems that model ecosystem service flows in a detailed manner (Bagstad et al., 2013; Tallis et al., 2009) and incorporate economic, ecological and also social values (Bryan et al., 2010). One of these relatively simple methods has attracted much attention both inside and outside the scientific community. The ‘table scoring’ method developed by Burkhard et al. (2009) uses land cover or land use maps as proxies for ecosystem service supply and demand (Burkhard et al., 2012). The methodology assumes ecosystems to be the main unit for ecosystem service supply. Based on expert and local knowledge, the land use types can be scored against different values and criteria (quantities, monetary, etc.). Therefore, the methodology requires only a limited amount of data and technology (e.g. computer power), making it an easy to use, popular mapping and evaluation method (e.g. Kaiser et al., 2013; Nedkov et al., 2012; van Oudenhoven et al., 2012). The method has also been used to assess changes in ecosystem service delivery over time (Kroll et al., 2012; Lautenbach et al., 2011). However, the use of land use proxies for ecosystem service assessments has also been criticized as being a poor fit for the actual ecosystem service provision due to uncertainties in the scoring system and discrepancies between the land use classes and the ecosystem functions that provide the ecosystem service (Eigenbrod et al., 2010; Hou et al., 2013).

Previous studies in Africa on ecosystem services have made use of both complex models e.g. Swallow et al., 2009) and landscape proxies (e.g. O’Farrell et al., 2012; Willemen et al., 2013) O’Farrell et al., 2012; Willemen et al., 2013), but mapping of ecosystem services is potentially limited, especially in data scarce regions as Africa, by the available data (Egoh et al., 2012). In the absence of local data, primary data from regional or global studies (e.g. de Groot et al., 2012) are often used (e.g. Leh et al., 2013). However previous ecosystem service studies have demonstrated that data and value transfer between case-studies should be handled with care (Plummer, 2009). Correspondence between the case-study and data source sites can be limited and/or difficult to assess. As INRM encompasses all natural resources, all relevant ecosystem services need to be addressed and the risk of missing data increases. As an alternative, INRM considers local knowledge to be highly relevant for good management as it can improve analysis and increase local support for management actions (Frost et al., 2006). Therefore, the integration of local knowledge in a qualitative assessment in combination with land cover proxies might be more appropriate than relying on data transfer for quantitative assessments in data scarce case studies and regions.

In this study we developed a methodology to take different flow types into account while using expert scoring tables for ecosystem service mapping and evaluation. The methodology was developed and tested within the Lake George catchment in western Uganda as part of a larger INRM research project, Afromaison, which incorporates ecosystem services into different steps of the INRM framework. Our approach was designed to integrate ecosystem services into INRM for implementation in data-scarce regions. The presented analysis was developed as a communication tool for stakeholders. The tool facilitates communication on how management actions can have an impact on the ecosystem service distribution within the region. The study focusses on local, semi-subsistence livelihoods but does not consider ecosystem service supply, demand and flows on larger scales. Different GIS scripts were designed to reflect the different ecosystem service flows that are relevant to the local communities. To evaluate the impact of the flows, the outputs of these scripts were compared with the original maps from expert scoring. To evaluate the delivery of ecosystem services in the region ant test different INRM strategies, several land use change scenarios were used to test the effect of these different interventions on the delivery of provisioning and cultural ecosystem services.

2. MATERIAL AND METHODS

2.1 Study area

Lake George catchment, located in Western Uganda (Figure 6-1), consists of a series of rivers that are situated around and drain into Lake George. The catchment is part of the Great Rift Valley and is characterized by diverse geophysical and ecological systems: high mountains, tropical high forest, savannah and papyrus wetlands. It includes also several important protected areas: Queen Elisabeth Park, Kibale National Park and Rwenzori Mountains National Park (Figure 6-2).

Figure 6-1: Location of the Lake George Catchment within Uganda.

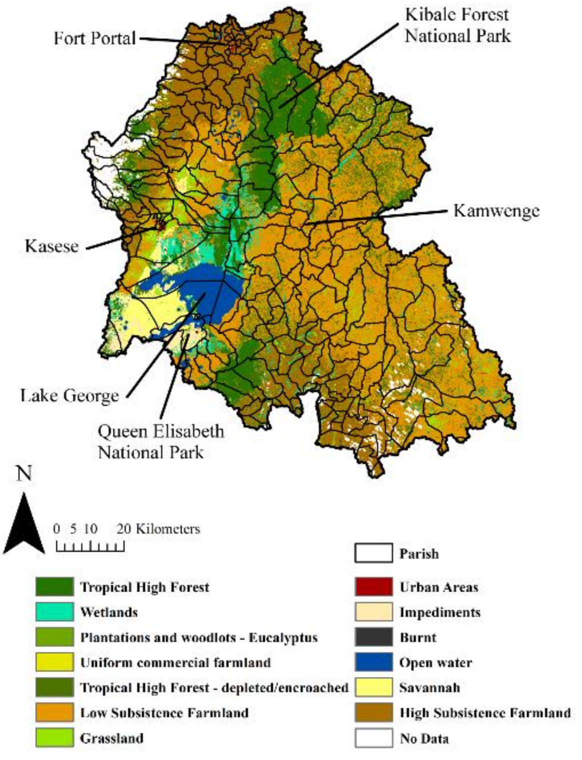

Figure 6-2: Overview of the Lake George Catchment.

Land and natural resources are essential for rural livelihoods in Sub-Saharan Africa (Abalu et al., 1998). In Uganda 88% of the population lives in agrarian areas, making them vulnerable to natural resource degradation. Population densities in Western Uganda are high and were reported in 2002 to be 110 inhabitant/km². At the same time the region is confronted with a high birth rate (245 births/1000 inh. in 2011) and immigration from Congo. This results in high population growth in Uganda (3.4% per year) and rapidly increasing population densities (Uganda Bureau of Statistics, 2005). For example at the eastern border of Kibale National Park population density is now estimated to have reached more than 335 inh./km² (Hartter et al., 2009).

High population densities and ongoing growth, combined with low agricultural efficiency, have resulted in an ever increasing demand for natural resources (e.g. wood, water, etc.) and additional agricultural land in Uganda (Nkonya et al., 2008). These pressures have resulted in increasing deforestation and land degradation. Conversion of natural ecosystems to agricultural land is associated in Uganda with deforestation, wetland degradation and soil erosion (Hartter et al., 2009; Laurance et al., 2014; Mutekanga et al., 2010; Nakakaawa et al., 2011). This also results in an increasing pressure on the remaining natural ecosystems for resources, leading to additional overexploitation and loss of biodiversity (Hartter et al., 2011). The high rate of ecosystem degradation in the region threatens the long term delivery of many ecosystem services and the sustainable development of the region (Zhen et al., 2014).

In the past decades Uganda has undergone a process of decentralization in which rights and responsibilities were transferred to the local government. However, at a local level there is confusion on the right of access and use of natural areas. Nevertheless, natural resource management on a local level is the only viable option for effective management in Uganda (Hartter, 2010), as it is the only scale at which actions for management can be effectively communicated to those who need to implement them, the local communities and people. Therefore, methods are needed to help in the development of this local natural resource management.

2.2 Mapping ecosystem service supply

For this study, we used the ecosystem service classification system of The Economics of Ecosystems and Biodiversity report (TEEB, 2010). This classification system was found to be the best suited system at the beginning of the project. However, some changes were made by a team of local researchers, familiar with the region, to better represent the local situation. For example, ‘raw materials’ from the TEEB study was split into ‘Timber and Fuelwood’ and ‘Fodder’, because the high significance of both to the region. Other services were removed as they were considered to be less important. To assess ecosystem services, scoring tables for ecosystem service supply were developed. The same team of researchers was responsible for scoring the different tables, incorporating knowledge they gathered during several stakeholder meetings in the period 2012 – 2013. In these stakeholder meetings discussions on INRM took place between local representatives from government, management agencies, relief and development organisations and industry. Scorings were assigned between 0 (no supply) and 5 (high supply). Tables were scored for each land cover type present in the region. The land cover typology was based on a national biomass study (Drichi, 2002). Initially the tables were used to create ecosystem provisioning maps following the methodology developed by Burkhard et al. (2009). No detailed land cover map of Uganda existed that represented the current situation in Uganda, so a land cover map was developed based on Landsat 5 TM images from January 2010 and other local data sources, resulting in a map with a 30m resolution. The region is cloudy throughout the year and no data was available for some areas (Fig. 1).

2.3 Mapping ecosystem service demand

Beneficiaries of ecosystem services can be situated at different scales (local, regional, global). In this study, ecosystem services were only assessed in relation to local livelihoods at meso-scale. Therefore, ecosystem services demand was evaluated on a local level using population density data, disregarding potential demand on higher geographic levels (e.g. national or international demand). Ugandan census data from 2002 (Uganda Bureau of Statistics, 2005) were recalculated to population densities per parish, to adjust for the differences in parish sizes. Parishes are the smallest administrative level in Uganda and the level on which census data are collected, but parish size can differ strongly and ranges within the study area between 102ha and 26.334ha. To integrate the demand data into the qualitative ecosystem service supply maps, the range of population densities were reclassified to an indicator between 0 (no population/no demand) and 5 (high population density/high demand). Population densities above 0 were reclassified using an ‘equal area’ algorithm, so that indicator values between 1 and 5 approximately cover the same proportion of area within the catchment. This reclassification method was selected, instead of a linear classification, to better represent the spatial distribution of the demand across the catchment.

2.4 Mapping ecosystem service flows

Fisher et al. (2009) identified four different flow mechanisms between ecosystem services supply and ecosystem services demand areas. This classification was selected for this study as it encompasses the most important ecosystem service flows and it is easily to communicate. In some cases, e.g. water supply, these flow mechanisms have to be combined to encompass the entire flow of the ecosystem service. Therefore, an extra flow mechanism, ‘combination of flows’, was developed that combines both gravitational and omni-directional flow. Scripts were developed to incorporate each of these flow mechanisms into the ecosystem service scoring evaluation (Figure 6-3). For each ecosystem service, the associated flow mechanisms are set out in Table 6-1. The scripts were developed in Python incorporating tools available in ArcGIS 10.2 (ESRI Inc., 2010). An overview of the data processing steps is given for each flow mechanism in Table 6-2.

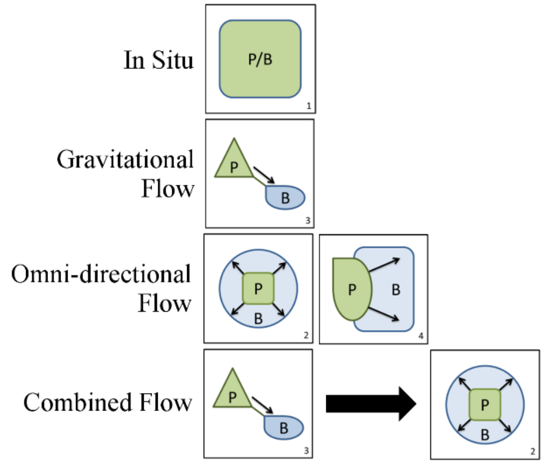

Figure 6-3: Overview of the different flow mechanisms between provisioning (P) and benefiting areas (B) evaluated within this study (Symbols after Fischer et al., 2009).

Table 6-1: Overview of ecosystem services and associated flow mechanisms.

Provisioning Services	
Water supply	Combination of flows
Timber and Fuelwood	Omni-directional flow
Capture and Collection	Omni-directional flow
Crops	In situ
Fodder	Omni-directional flow
Regulating Services	
Water Quality Regulation	Gravitational flow
Water Flow Regulation	Gravitational flow
Soil Maintenance	In situ
Erosion Control	In situ
Flood Control	Gravitational flow
Carbon Storage	In situ
Habitat services	
Habitat	In situ
Maintenance of Biodiversity	In situ
Cultural Services	
Ecotourism	Omni-directional flow
Cultural Significance	Omni-directional flow

Table 6-2: Overview of the analysis steps used to map supply and demand.

a) In situ	Input	Ecosystem service score map
	Steps	No special calculations are needed.
	Output	The original ecosystem service score map
b) Gravitational flow	Input	Ecosystem service score map
	Steps	1. Calculate total upstream area for each river reach. 2. Calculate total ecosystem service score based on the upstream area for each river reach. 3. A mean ecosystem score for each reach is calculated by dividing the total ecosystem score from step 2 with the total upstream area of step 1. 4. Reclassify mean values into 5 classes/scores. Each class contains an equal number of reaches.
	Output	Ecosystem service supply score map with values for the different river reaches
c) Omni-directional flow	Input	Ecosystem service score map
	Steps	1. Create separate maps for each score occurring in the Ecosystem service score map. 2. Calculate Euclidean distance to data pixels for each map (score), with a threshold distance of 10 kilometers. The score will decrease with increasing distance from the score value to zero. 3. Recombine these Euclidean distance maps by selecting the highest value for each raster cell. 4. The combined "maximum score" map is reclassified in to 5 classes/scores using an equal area algorithm. Each score will cover the same amount of area within the Lake George catchment.
	Output	Ecosystem service supply score map for the entire region
d) Combined flow	Input	Ecosystem service score map
	Steps	1. The different steps of the gravitational flow method are calculated 2. The different steps of the omni-directional flow method are calculated. As input, the map with ecosystem service scores of the reaches is used.
	Output	Ecosystem service supply score map for the entire region
e) Demand	Input	Population density map on parish level
	Steps	1. Reclassification with equal area algorithm
	Output	Ecosystem service demand score (0-5) for each parish

2.4.1 *In situ*

For only a few ecosystem services, demand spatially coincides with the ecosystems that generate/supply these services (= In situ). Most of the ecosystem services require some kind of movement or flow between provisioning area and beneficiary. Agricultural provisioning is considered to be used locally within the region, since most of the population in the Lake George basin depends for their food provisioning on the harvest of their gardens, which surround their houses. Part of the harvest is traded on local markets in the surrounding parishes and along the main roads that cross the basin. Some specific agricultural products (e.g. tea) are traded nationally or even internationally, but these flows are not taken into account due to the focus on local livelihoods and a lack of data.

For some ecosystem services, especially regulating services, the demand takes mainly place on a higher spatial and/or temporal scale (e.g. carbon storage for climate mitigation). Although the impact of these services can be large over time on local livelihoods, we consider demand being equal across the entire catchment. Therefore, no specific flow needs to be integrated. Erosion control has a local effect as well as an impact downstream. In this study we only consider the local effect.

In these cases, no special calculation is needed and the supply scoring tables can be used directly.

2.4.2 *Gravitational flow*

Several ecosystem services – particularly those related to water flows - are mediated by a (partially) gravitational flow between areas supplying ecosystem services and beneficiaries. Water is transported downstream to users, while the amount and quality of the water is influenced by the upstream land uses, activities and management. Although a local ecosystem score can be relatively good, the upstream areas can severely degrade the actual supply of the service. Therefore, the impact of the entire upstream areas must be evaluated to assess the ecosystem services along a river.

Incorporating gravitational flow in the analysis required the following steps. First, the ecosystem scores of upstream land uses were combined to give an overall score for the upstream areas. Secondly, a river network needed to be defined. The river network and subcatchments of the Lake George catchment were delineated from the Aster II digital elevation model using the hydrological tools in ArcGIS 10. Then, both the subcatchments and the river network were then used to calculate mean ecosystem services scores for the different river reaches (= part of the river between two confluences). For each river reach a total ecosystem service score was calculated by multiplying the ecosystem services table scores of the land cover types with the associated land cover areas within the upstream catchment. This total score was then divided by the total upstream area to get a mean ecosystem service score for the river reach. Finally, the mean scores were finally reclassified using an equal area algorithm to reduce the values to integer scores (1 – 5). The equal area algorithm ensured that each score was attributed to almost the same number of reaches within the catchment. The ecosystem services that are evaluated with the gravitational flow have an impact on the river itself but no or only a limited impact on the surrounding region. Therefore, scores are only attributed to the reaches and not to the surrounding land. The scores allowed for further analysis (see 2.5 and 2.6).

Ecosystem services that are evaluated with the gravitational flow script are water bound (water quality regulation, water flow regulation and flood protection) and have no or limited impact on the surrounding region. Flood protection does have a significant impact on the flood areas adjacent to rivers, but it is difficult to assess the size of the area affected and it is difficult to translate this in to maps. Therefore, values are assigned to segments of the river system itself and not to the surrounding land. As a result, ecosystem services scores are limited to the river reaches and cannot be evaluated on a parish level. Instead a comparison is made between the original mean ecosystem services scores on subcatchment level and the output of the flow script for the related river segments within the sub-catchment.

2.4.3 *Omni-directional flow*

The most common flow mechanism for ecosystem services is the movement of people to the supplying areas for local use or extraction of the service. Service provisioning depends on both the intensity of the supply (supply score) and the distance that needs to be covered to the provisioning area (distance decay effect). Because the focus of this study was on the role of ecosystem services in semi-subsistence livelihoods, the actual service delivery is limited by the ability of people to walk to the supplying areas. We hereby disregard other means of transportation. Maximum walking distance was established at 10 kilometres, based on local knowledge.

A Euclidean distance algorithm was used to calculate a separate distance map for each of the six ecosystem services scores (0-5). Therefor the original supply map was split into 6 different maps, where each map comprises the areas with the same score and for which the Euclidean distance was calculated. The Euclidean area algorithm results in maps with decreasing ecosystem services scores as distance from the supplying areas increases. At a distance of 10 kilometres the ecosystem services score will be zero. The 6 maps were then integrated into one by selecting the highest value for each cell. The assumption here is that people will make a trade-off between the amount and quality of the provided ecosystem service and the distance they have to walk to reach these supply areas. The integrated map was finally reclassified, using the previously described equal area algorithm.

2.4.4 *Combination of flows*

For many ecosystem services, multiple flow mechanisms contribute to the flow of ecosystem services to the beneficiaries. Therefore, a combination of flow mechanisms is needed to mimic delivery. As an example, we take the case of water provisioning, where water first flows downstream through the rivers (gravitational ecosystem services flow mechanism), and beneficiaries walk to the stream or lake to collect the water (omni-directional ecosystem services flow). Water provisioning is evaluated based on both supply from the upstream catchment and the maximum distance people can walk to get to a water supply. Because the catchment has many crater lakes, these sources also have to be taken into account as potential supply. This is achieved by integrating the procedures for gravitational and omni-directional ecosystem services flows. First a score is calculated for the different river reaches. Subsequently, the omni-directional flow is calculated for both the river reaches scores and the crater lakes that are extracted out of the original ecosystem service maps.

2.5 Evaluation of flow effects

The effects of accounting for ecosystem service flow mechanisms on the spatial configuration of the scoring maps were evaluated by comparing the resulting maps with the original ecosystem service scoring maps. Analysis was done on a level that allowed us to compare the different maps and assess changes in spatial distribution of the ecosystem service provisioning. Therefore, mean ecosystem scores were calculated on a parish (omni-directional flow and combination of flows) or subcatchment (Gravitational flow) level. Ecosystem scores were evaluated as a value per square meter, summed for the entire parish or subcatchment and divided by total area of the parish or subcatchment. The derived mean values were then reclassified to an integer score between 1 and 5 (no zero values were present on parish or subcatchment level) using an equal area algorithm. As a result, indicator scoring between 1 and 5 covered almost equal areas within the catchment for both the original as well as flow maps. By ruling out the effect of changes in value distribution (applying the equal area algorithm), the effect of the flow calculations on the spatial configuration of ecosystem services could be analysed.

2.6 Evaluation of ecosystem services delivery

Parishes are the smallest scale for which demand maps can be made and management can take place. As such they can be regarded as service supply/demand units. Analysis of supply/demand mismatch at the parish level is only useful for provisioning and cultural services because the demand for regulating and habitat services is independent of population densities and/or takes place on higher spatial levels. The maps of the selected ecosystem services supply and demand on parish level were combined and evaluated with a conversion table (Table 6-3), which translates each of the supply and demand combinations in to a new value class between 0 and 5 (Figure 6-4), representing ecosystem services delivery. High values imply a combination of a high demand and a low supply, indicating an important shortcoming in delivery. Low values indicate a sufficient availability of the ecosystem services or a low demand.

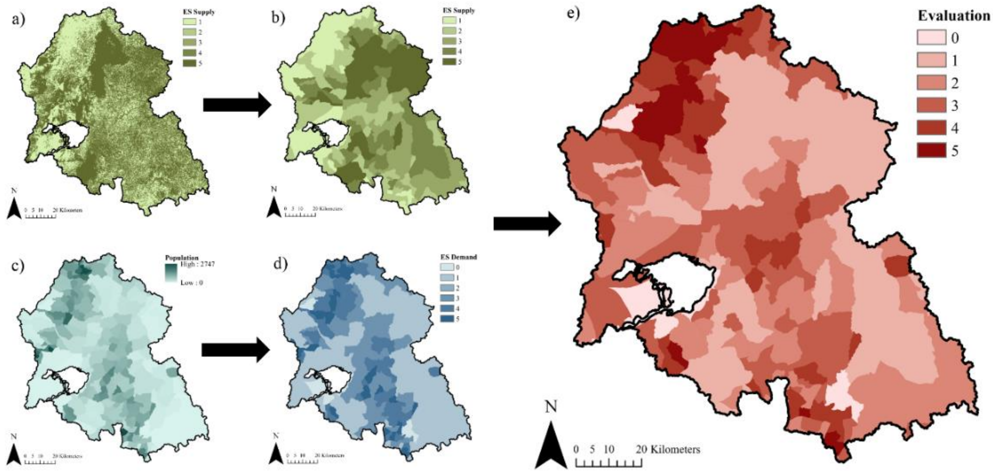

Figure 6-4: Overview of how both ecosystem services supply (A) and ecosystem services demand (B) are translated to qualitative indicators on parish level (B and D) and are combined to assess the actual delivery of ecosystem services (E).

Table 6-3: Table to assess delivery of ecosystem services based on the supply and demand map. High values indicate a potential deficiency in ecosystem service supply compared to the demand.

		Supply					
	score	**0**	**1**	**2**	**3**	**4**	**5**
Demand	**0**	0	4	4	5	5	5
	1	0	3	4	5	5	5
	2	0	2	3	4	4	4
	3	0	2	3	3	3	3
	4	0	1	2	2	2	2
	5	0	1	1	1	1	1

2.7 Scenario analysis

Three land use change scenarios were developed to simulate potential trends in development and implementation of specific natural resource management actions. Each of the scenarios targets a specific type of land use and associated range of ecosystem services. But improving specific land-uses and ecosystem services typically also affects other ecosystem services. The evaluation of these scenarios gives an indication on how ecosystem service delivery is affected across the research area. This information allows better informed choices in land development to improve local livelihoods.

- **Deforestation:** For this scenario we consider a complete deforestation of the region (including national parks) in five time steps. This represents a worst case scenario for deforestation in the catchment. Due to a lack of reliable data we did not define a specific time-step for the deforestation. Therefore, we opted for the following approach. In each of the five time steps one fifth of the remaining forests is converted to agricultural land. The scenario is developed to first deforest the smaller, fragmented areas. Only in the later stages the large forests will fully disappear. The goal of this scenario is to see how the different ecosystem services are impacted by the ongoing deforestation and which areas will be affected the most.

- **Agroforestry:** 25% of the most erosion-vulnerable areas of the region that are now used for agriculture are converted to agroforestry. The scenario targets improvement in both water related services and wood production.
- **Plantations:** Parts of the agricultural land are converted to plantations for wood production. The areas for conversion are selected based on population densities and erosion vulnerability. The higher the population density the more of the agricultural land (maximum 25%) will be converted to plantations for wood production. This scenario targets wood production at the expense of agricultural production.

In order to make a comparison between the current land use and scenario outcomes, we used the following protocol. For the actual land-use, the results from the flow scripts were reclassified using an equal area reclassification for each ES. The break values of this equal area reclassification were then used to reclassify the ecosystem service scores for all scenarios. Applying a new equal area algorithm for each scenario and service would make a comparison between the scenarios impossible. This would in fact change the meaning of each ecosystem score and shifts in values would be masked, because it would result different break values for each scenario and service. The deforestation scenario would for example result in a significant decrease in timber and fuelwood provisioning. However, the equal area algorithm would still give one fifth of the case-study a score of 5, making a comparison between scenarios meaningless. Instead the break values used in the equal area algorithm to reclassify current land use scores after calculating the different flows were applied to the scenario outcomes. As a result, the areas with a certain ecosystem services score would increase or decrease compared to the current land use analysis.

Ecosystem services demand was considered static, although we are aware that also demand is affected by land-use change. The high population growth in Uganda will result in increases in population densities and eventually also affect population distributions within the catchment. Local increase in population densities will for example coincide with conversion from forest to agricultural land and vice versa. The scenarios will therefore also result in local changes in ecosystem service demand. We were however not able to develop reliable population scenarios that can objectively be linked to the scenarios. Therefore, the land use scenarios were evaluated with the existing population data.

3. RESULTS

3.1 Original scoring table

Together, local experts scored the ecosystem services for each land use class, incorporating knowledge they gained from several stakeholder workshops (Table 6-4). Natural vegetation types generally provide more ecosystem services and generate higher scores than anthropogenic land uses. Local knowledge was especially valuable for the evaluation of some provisioning and cultural services. Most of the scorings are in line with expert scoring from other studies. However, some results are case-study specific. The water supply score of plantation and woodlots is lower than farmlands, since plantations and woodlots generally consist of plants (e.g. eucalyptus) that have a higher water use than crops, which results in lower water availability.

Grasslands also have a high cultural significance for some of the ethnic groups in the research area.

Table 6-4: Overview of the ecosystem service supply scores for the different ecosystem services and land uses.

	Provisioning Services	Water supply	Timber and Fuelwood	Capture and Collection	Crops	Fodder	Regulating Services	Water Quality Regulation	Water Flow Regulation	Soil Maintenance	Erosion Control	Flood Control	Carbon Storage	Habitat services	Habitat	Maintenance of Biodiversity	Cultural Services	Ecotourism	Cultural Significance
Tropical High Forest		5	5	5	0	2		4	5	4	4	4	5		5	5		4	5
Wetlands		5	1	5	2	5		5	5	0	0	5	4		5	5		2	1
Plantations and woodlots - Eucalyptus		0	4	0	0	0		0	0	4	4	0	0		1	1		0	0
Grassland		2	0	2	0	5		2	2	2	4	2	1		2	2		0	5
Tropical High Forest - depleted/encroached		2	5	2	1	2		2	2	2	2	2	3		3	3		1	3
Uniform commercial farmland (lowland)		1	0	0	5	0		1	0	2	2	0	1		1	1		0	0
Woodland		2	5	4	0	2		4	3	4	4	3	3		5	5		1	3
Urban		0	0	0	0	0		0	0	0	0	0	0		0	0		0	0
Savannah		3	2	3	1	1		3	3	3	3	2	4		2	3		2	3
Burnt		0	0	0	0	0		0	0	0	0	0	0		0	0		0	0
Open water		5	0	5	0	0		5	5	0	0	5	0		5	5		4	5
Impediments		0	0	0	0	0		0	0	0	0	0	0		1	1		1	1
Subsistence Farmland (highland)		2	2	2	5	2		2	2	3	3	2	1		2	2		1	5
Agroforestry (lowland)		3	5	2	4	4		3	2	4	4	3	4		2	3		2	1
Agroforestry (highland)		3	5	2	4	4		3	2	4	4	3	4		2	3		2	1

3.2 Flow effects on scores

Each of the ecosystem services flow calculations has a different impact on the evaluation of the service, changing how scores are distributed within the region (Figure 6-5).

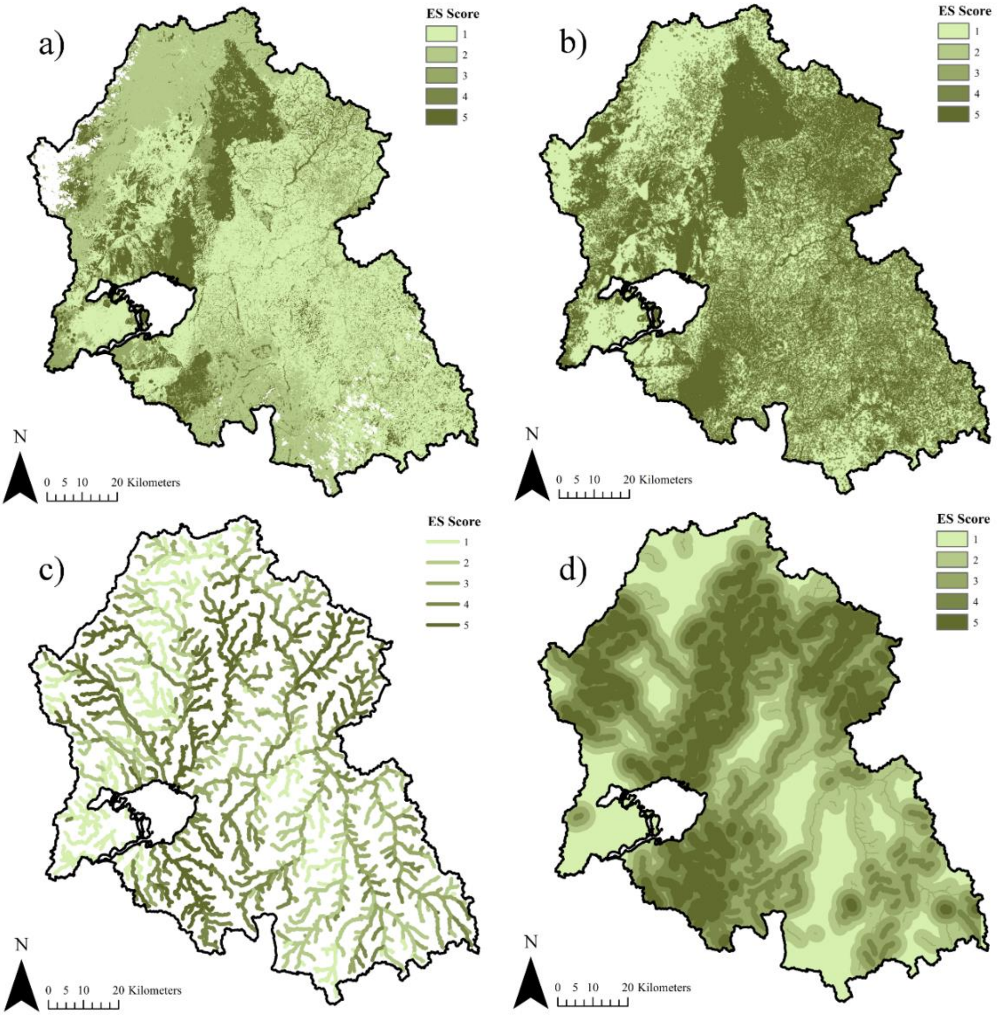

Figure 6-5: Output examples of the different flow scripts a) In Situ (biodiversity), b) Omni-directional flow (Timber and Fuelwood), c) Gravitational Flow (Water Quality Regulation) and d) Combined Flows (Water Provisioning).

Integration of the omni-directional flow in the ecosystem services score calculation has a clear impact on the spatial configuration of the ecosystem services scores (Figure 6-6). When comparing the mean ecosystem services score on parish level there are clear changes in values. The impact for many parishes is relatively small, some parishes are heavily affected; and the impact differs between the different services. Fodder production scores for example changed drastically, completely altering the spatial configuration of the service. This indicates that provisioning areas for fodder production are unevenly distributed within the region and located outside of the main living areas.

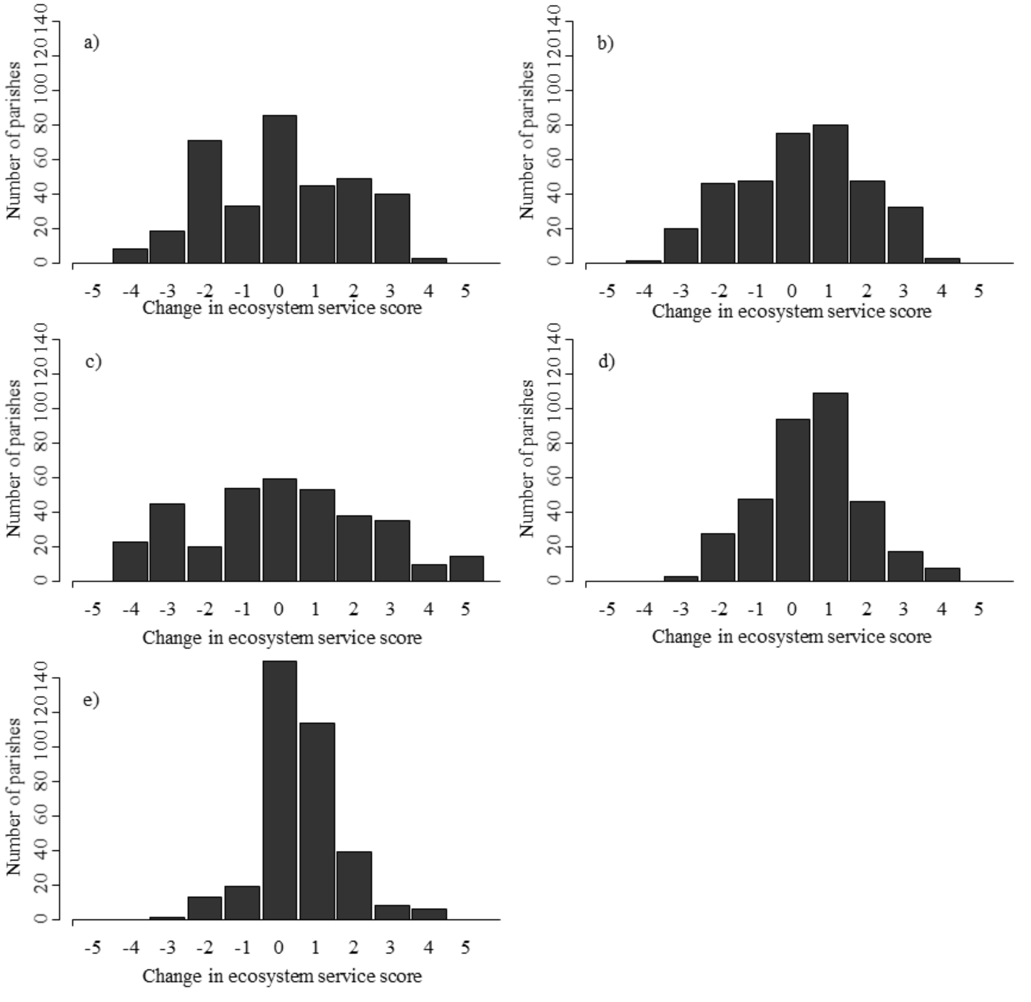

Figure 6-6: Changes in ecosystem score on parish level between original calculation and after taking the ecosystem services flows (Omni-directional) into account a) Timber and Fuelwood, b) Capture and Collection, c) Fodder, d) Cultural Significance, e) Ecotourism.

Incorporating gravitational flow results in a different type of output compared to the other three flow mechanisms. Values are assigned to the different river segments of the river system itself and not to the surrounding land (Figure 6-7). Again the different ecosystem services are impacted to a different extent by the flow script. For example, some mountain sites of the Rwenzori Mountains are heavily impacted by deforestation and the development of agriculture. Streams downstream of these areas have a poorer water quality compared to the more pristine rivers coming from the mountains. These differences are reflected by the flow calculation. Water provisioning scores were calculated and evaluated on a parish level by combining both gravitational and omni-directional flow scripts. The results after incorporating the combination of flows correspond more closely with the actual situation than the scores mapped without accounting for ecosystem services flows. For example, water provisioning downstream of Kibale Forest is much better due to water quality improvement and buffering of flow, compared to the reaches upstream of the forest, which are polluted by the city Fort Portal. Areas located nearby or downstream of forests and wetlands sites have the highest ecosystem services scores as, the upstream forests ensure a better provisioning of water to areas where it is not expected in first instance. This effect is also visible along downstream river reaches that run through agricultural areas. The effect of the upstream forest weakens further downstream as agriculture and habitation progressively degrade the water provision. The changes in ecosystem service scores on parish level are depicted in Figure 6-8.

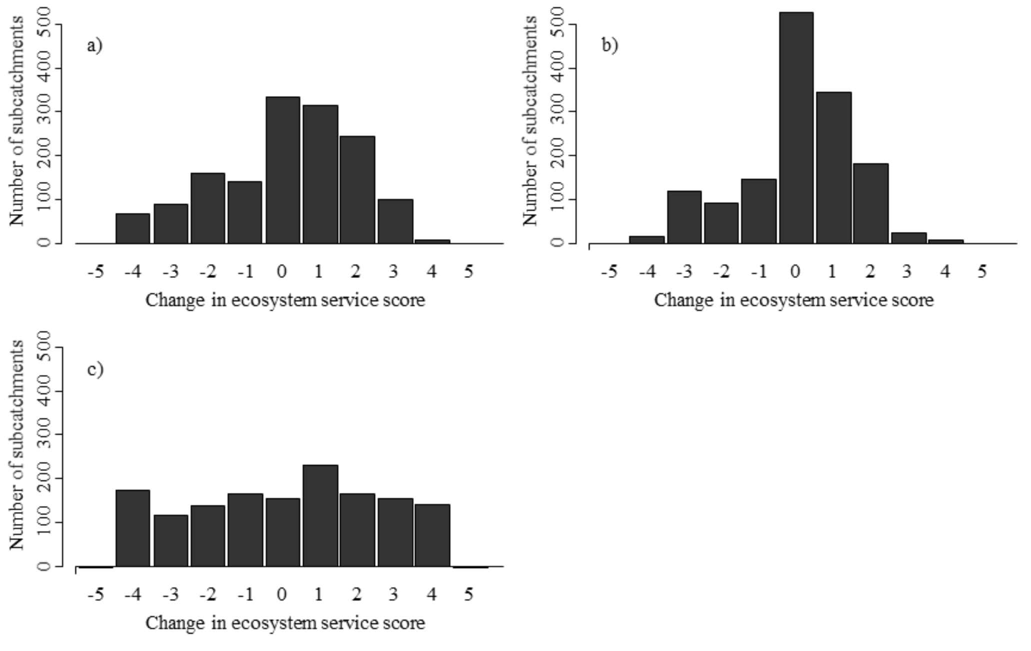

Figure 6-7: Changes in ecosystem score on subcatchment/river segment level between the original calculation and after taking ecosystem services flows (Gravitational) into account: a) Water Quality Regulation, b) Water Quantity Regulation, c) Flood Regulation.

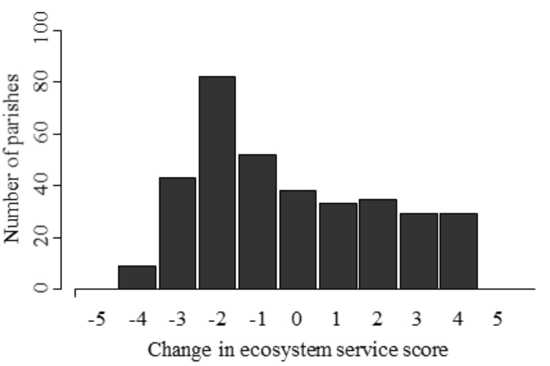

Figure 6-8: Changes in ecosystem score on parish level between the original calculation and after taking the ecosystem services flow (Combined) into account for Water Provisioning.

3.3 Scenario analysis

The scenarios were analyzed, taking the different flows into account, and assessed on their impact on both ecosystem services scores and ecosystem services delivery. Deforestation has a strong negative impact on many of the ecosystem services, for example water quality regulation (Figure 6-9). The first step in the stepwise deforestation scenario has by far the strongest impact on the water quality regulation. This is because the scenario first deforests the smaller, fragmented areas. Therefore, many of the smaller, upstream subcatchments are strongly impacted as they lose their last forest remnants. The larger forest areas, that are situated more downstream, are less vulnerable to this first step of deforestation. As a result, the downstream reaches of the river network are only strongly impacted in a later stage of the deforestation.

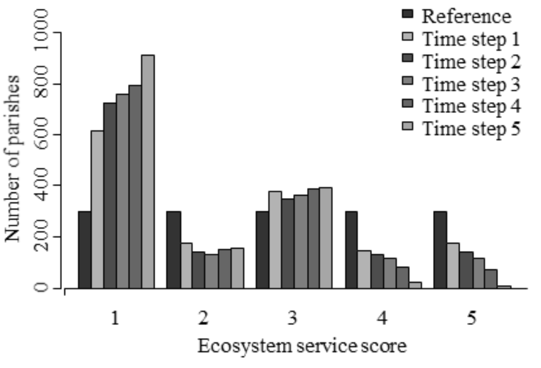

Figure 6-9: Changes in Water Quality Regulation on river segment level for the reference scenario and the 5 deforestation scenarios.

In addition to this worst case scenario of stepwise deforestation, two alternative management scenarios were developed to improve natural resources and ecosystem services delivery, namely agroforestry and plantations. When evaluating both management scenarios for the ecosystem service 'timber and fuelwood production', strong differences in impact were found (Figure 6-10). Deforestation has clearly a negative impact on the provisioning of wood. The number of parishes that have scores between 2 and 5 all decrease, while the number of parishes with score 1 increases. At the same time the discrepancy between demand and supply for timber/wood increases. When evaluating the management scenarios, agroforestry has by far the largest impact on ecosystem services supply. Most of the parishes reach an ecosystem score of 5 and a discrepancy in delivery of 1. The impact of the plantations scenario on ecosystem services delivery is less pronounced because the experts attributed lower scores to the plantations (4 instead of 5). As a result, the positive changes in ecosystem service supply have a spatial mismatch with the demand for ES.

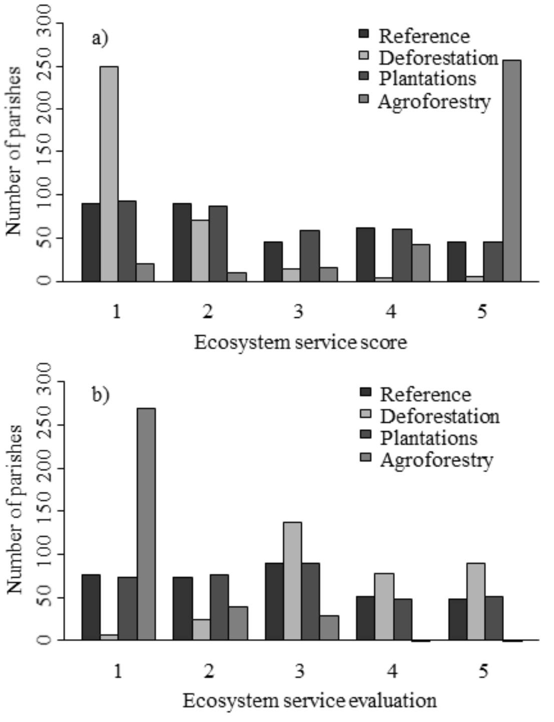

Figure 6-10: Changes in ecosystem service score (a) and delivery evaluation for Timber and Fuelwood production under the different scenarios.

4. DISCUSSION

4.1 Integration of flows

A methodology was developed to integrate flows into ecosystem services scoring maps and assessments. To do so, scripts were developed to recalculate the qualitative scorings, taking spatial relationships into account. As there are distinct differences in characteristics between omni-directional and gravitational flow, the scripts reassess ecosystem scores in different ways. The resulting maps give different weights to the ecosystem scores, compared to the original scoring maps.

Taking flow mechanisms into account changed and improved the results, as it incorporates a crucial aspect of the ecosystem services concept, namely the spatial connectivity between ecosystems and their beneficiaries (Bagstad et al., 2013). Ecosystem supply and demand scores are typically evaluated on the level of administrative units, for which socio-economic data (ecosystem service demand) is available. However, in the classic approach discrepancies between supply and demand are difficult to assess on this administrative level, as effects of the ecosystem service flow over administrative boundaries can have a profound effect. For example, large forests located in the adjacent parishes can have a positive impact on the delivery of the ecosystem service; but will be completely disregarded in the classic approach. By integrating flows, the full potential of the service providing areas can be taken into account, making the ecosystem service evaluation more reliable.

However, not all flows could be incorporated into the analysis. Other means of transportation like motorbikes and trucks increase/improve ecosystem service delivery in some parts of the region, but these flows are more complex and difficult to predict. Although anthropogenic infrastructure and other investments do facilitate certain flows of ecosystem services, they were less relevant within the context of this study. The aim of this study was to analyse and provide information to improve the livelihoods of the local, poor people in the region. Ecosystem services flows at higher spatial scales are less relevant as transportation is most often unavailable to these local people. Further research may be imperative, as such transport mediated ES-flows become important in developing countries and compete with local ES-flows.

Our results clearly illustrate the significance of incorporating flows in the ecosystem service analysis. However, the importance is dependent on the scale and objectives of the study. For some ES, such as climate regulation, flows were not considered, because they encompass a much higher administrative or management level. The selection of the relevant ecosystem services and flows thus depends on the scale and objectives of the study.

The demand for an ecosystem service has, compared to the supply, an equal weight in the final result, when evaluating the ecosystem service provisioning. Therefore, the mapping of the demand and spatial representation in the analysis is of equal importance and needs to be well considered. To evaluate and integrate the demand for ecosystem services in the region, we used population densities as proxies. Demand for the ecosystem services that were tested for delivery (provisioning and cultural services) is directly related to number of people using these services e.g. amount of drinking water needed or number of people who use an area for cultural practices. Although we initially tested land scoring tables for mapping demand, we considered population density to be a more reliable, indicator for many, (but not all) ecosystem services than the land cover map.

The results obtained from the analysis cannot give us information as to the extent that this supply meets the actual demand. Qualitative maps can therefore not be used to analyse whether ecosystems are used in a sustainable way, as the qualitative data do not contain the correct information to make these conclusions. They can however indicate where in the region the discrepancies between supply and demand are the highest, overexploitation is most likely, management actions are mostly needed and where they will have the biggest impact.

Due to the data scarcity and the qualitative nature or the output, validation of the model results remains a challenge and a research topic for the future. To what extent the qualitative model outputs represent the actual situation is difficult to assess. Further stakeholder consultation might help to improve the maps in the future and evaluate their weaknesses. But communication of the results has to be well considered and a consistent evaluation methodology needs to be developed.

4.2 Use of the methodology

Incorporating ecosystem services is not yet common practice for INRM-assessments, and how they can be integrated in INRM is still a topic of discussion. Where in the INRM framework can ecosystem services have an added value? Can quantitative and qualitative INRM and ecosystem service methods be compatible? How do we cope with different levels of accuracy? Can we integrate the different ecosystem services taking different types of flow into account?...

INRM aims to evaluate all relevant natural resources and is benefited by methods that can analyse this full range of natural resources. Qualitative assessments of ecosystem services provide an opportunity to comprehensively analyse all relevant ecosystem services. Exclusion or misrepresentation of important ecosystem services in INRM can result in misleading outcomes and undermine the reliability of proposed management actions. Qualitative assessments are less limited by data availability than quantitative approaches (Busch et al., 2012), making them highly applicable in data scarce regions.

Inclusion of local stakeholder knowledge is also an important factor in INRM, which emphasises integration of local knowledge in developing management strategies and building local support for management (Douthwaite et al., 2005). Subjective qualitative assessments by multiple stakeholder groups can help to assess the diversity of stakeholder views and perspectives relating to ecosystem services delivery, and improve stakeholder interaction.

In both cases, data scarcity and stakeholder incorporation, qualitative assessments can contribute in the development of ‘good’ INRM plans.

In many studies “hot spots” that provide many different ecosystem services are mapped (e.g. Naidoo et al., 2008; Raymond et al., 2009), since these are considered to be highly valuable and require protection (Egoh et al., 2008). Although this information would benefit INRM, using the results from the different scripts for the analysis and identification of ecosystem services hot spots is not recommended. Combining the final ecosystem services maps has no meaning, because of the different types of flow mechanisms and outcome of the scripts. Combining maps of ecosystem services supply is appropriate for the mapping of supply hotspots, but usually does not incorporate use or flows to beneficiaries. Incorporating use and flow aspects would require scripts that can trace back ecosystem services from beneficiaries to the supply sites. These trace-back methods would also require a quantitative approach to ecosystem service flows and will not work with the qualitative scoring.

Due to the calculation procedures, ecosystem services scores will not have the same weight for each ecosystem service. The interpretation of the values is service and case-study dependent and the interpretation exercise is of as much importance as the development of the ecosystem scoring tables. In addition, not every ecosystem service is of the same importance to the case study or has the same weight within a study. Simply combining the different output results would lead to unreliable outcomes. The results should therefore be used to evaluate the ecosystem services individually, within a spatial context.

In data scarce regions it is difficult to assess whether data and knowledge transfer from other study sites is appropriate and useful. When considering INRM and livelihoods, local ecological and socio-economic characteristics can differ significantly even over short distances, and data transfer can therefore provide unreliable results. However, when good quantitative data are available or can be compiled, the ecosystem services scoring method should be avoided and instead quantitative methodologies or qualitative derivatives of the quantitative data should be considered.

5. CONCLUSION

The integration of ecosystem services in INRM has only just started and is still a topic of discussion. How and where it is integrated within the INRM framework can depend on the goal of the study and the characteristics of the study site. In data-scarce regions qualitative ecosystem services assessments provide a possibility to overcome some of the data issues and incorporate local knowledge in the analysis. However, in order to obtain reliable and relevant results for INRM, different aspects of the ecosystem service framework need to be addressed.

Ecosystem service flows are a fundamental aspect of the existing ecosystem service concept, although not often addressed, especially in qualitative ecosystem services assessments, which lack a biophysical modelling component. In many studies, ecosystem services with a strong flow component between supply area and beneficiary are excluded from the analysis. In other studies, these spatial flows are not explicitly dealt with, which results in less reliable data when supply and demand are compared at the local scale.

Within the framework of a larger INRM project we developed scripts to mimic ecosystem services flow mechanisms between ecosystem services supply sites and the beneficiaries. We used conventional expert scoring methods for ecosystem services supply in the Lake George catchment, Uganda. The ecosystem services flow scripts were applied on these datasets to calculate ecosystem services delivery maps (combination of demand and supply).

Integrating flow mechanisms into the qualitative assessment of ecosystem services provided added value, as results better represent the actual situation. In this paper we demonstrate that the comparison of supply and demand of ecosystem service scores after incorporation of the spatial flow mechanisms differed significantly from the static analysis that is more commonly applied, but does not reckon with flows across administrative or physical boundaries. The incorporation of flows makes the evaluation of ecosystem services supply and demand more relevant and reliable for applications within an INRM context. However, the use of the mapping method should be carefully considered. These qualitative methods are useful for evaluation of management in data scarce regions, but do not replace more complex and accurate, models and methodologies where data are available to support them.

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Chapter 7 - Synthesis and general discussion

1. CATCHMENTS AS COMPLEX SYSTEMS

1.1 Determinative flow pathways

Understanding flow pathways throughout a catchment is the first, most important step to a system understanding and sound catchment management. Without a comprehensive overview of these pathways it is impossible to make reliable estimations of how management measures would affect the river system. In natural, undisturbed catchments different flow pathways can be identified (Figure 7-1A). Self-organization of the system leads to a high level of water retention and nutrient cycling. As such flow pathways generally buffer rain events and release limited amounts of substances in to the streams. Inter-catchment transfers are limited and only occur through groundwater flows, feeding river baseflow, or snow movement. Urbanization and agricultural intensification significantly diverge these flow pathways. In chapter 2 to 4 we explored several of these changes and their impact on hydrology and water quality (WQ) within the Nete catchment.

Urban development and wastewater treatment infrastructure creates entirely new flow pathways, altering both the rivers hydrology and water chemistry. During dry periods (Figure 7-1B) two distinct flow pathways can be identified related to urban development. The main flow pathway consists out of the sewer infrastructure which collects wastewater within its wastewater collection region and transports this towards the treatment plant (WWTP). As wastewater is produced continually throughout the day and season, WWTPs generate a consistent anthropogenic baseflow. This baseflow can be an important part of the overall river flow, especially during dry periods in summer (Chapter 2). The infrastructure is not bound by the natural catchment topography and wastewater is transported across and between different catchments. As sewers are often leaky systems, leaking but also collecting groundwater (Dirckx et al., 2009). Therefore, they can also create important inter-catchment transfers of groundwater. Smaller hamlets and individual houses are often not connected to collective sewer infrastructure and discharge their wastewater directly into the nearest stream. On these locations no inter-catchment transfers will take place. However, these houses can collectively generate smaller, but significant baseflow in the smaller streams.

During rain events (Figure 7-1C) flow pathways in urban areas will alter drastically compared to dry periods. Most villages and cities encompass large zones of impervious areas, where rainwater cannot infiltrate. This rainwater is most often collected and diverted to combined sewage systems or rain sewers. Sewer infrastructure cannot manage these amounts of water and sewer overflow devices will release rainwater and untreated wastewater at different locations throughout the catchment. In general, sewers create an enhanced hydrological connectivity towards the stream network, reducing travel time and increasing peak flows. But also impervious areas that are not connected to the sewage system create an increased flow response as rainwater is not able to infiltrate. Runoff is captured directly in nearby ditches and streams, reducing travel time. As a result, peak flows will increase, infiltration and groundwater recharge will lessen and subsurface flow toward riparian zones and stream will decrease.

Rainfed agriculture can also impact the catchments flow pathways. But it is less disruptive compared to irrigated agriculture and sewer infrastructure. In rainfed agriculture, soil compaction, drainage ditches and gullies can increase longitudinal, surface flows during rain events. In general, these flow pathways coincide with the natural, downstream flow directions. Although in some cases ditches are dug to divert water towards other streams to improve drainage capacity. These ditches can transport runoff water, during and after, rain events. But the deeper drainage ditches can also capture subsurface flows, lowering the groundwater table and reducing the lateral, subsurface flow towards riparian zones and rivers showers (Figure 7-1B - 4). Although these effects are known to be present within the catchment, their impact was not assessed within our analysis.

Infrastructure and agriculture not only change the magnitude and variability of the flows; they also generate alternative pathways to the stream network. These anthropogenic flow pathways affect the amount of rain that can infiltrate into the soil, reducing vertical (deep groundwater) and lateral (subsurface groundwater) flow. In response, the longitudinal flow (surface flow in reaches and rivers) increases significantly and their hydrological response to rain events becomes more flashiness and unpredictable (Chapter 2). Although groundwater flows are not a core topic of this thesis, results from Chapter 4 suggest that groundwater flows do impact the river's hydro-chemical signature over large and small distances.

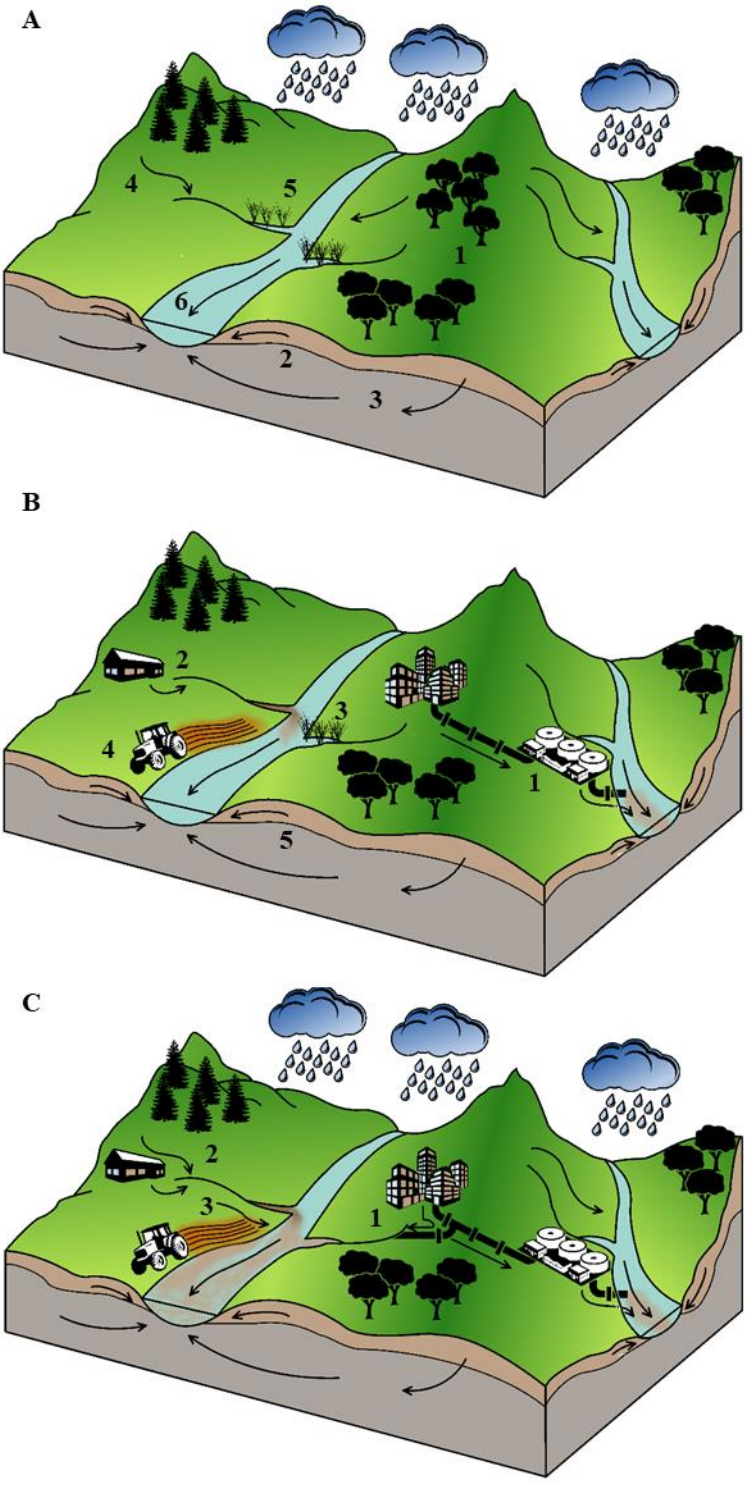
The combined effects of altered flow paths and the often abrupt changes in system behavior between dry and wet period makes urban and peri-urban catchments highly complex. The interaction between different anthropogenic impacts results in river systems that are rarely hydrologically or hydro-chemically connected to all land areas in their watersheds at all times. Nor do all areas of the landscape contribute equally to the river or stream (Seitz et al., 2011). These combined effects make these catchment systems and their flow pathways difficult to analyze, understand and manage.

Figure 7-1: Overview of the main flow pathways and their impact on the water quality (Original drawing).

A. Natural catchment under dry/wet conditions: 1. Rainwater will infiltrate into the groundwater, transporting nutrients and other substances 2. During dry and wet periods, subsurface groundwater seeps into the riparian zones and streams. 3. Depending on soil layers, part of this groundwater will infiltrate into deeper layers, flowing over longer distances between catchments. 4. When rain exceeds infiltration capacity, runoff carries sediments and other substances directly into nearby streams. 5. In-stream processes capture and transform sediments and substances 6. Eventually rivers carry water and substances further downstream, impacting downstream areas.

B. Catchment under human procures during dry periods: 1. Wastewater is collected through sewer systems and transported towards WWTPs in and outside the natural catchment. WWTPs will remove part of the nutrients and contaminants, but will still release significant loads of these substances. 2. Houses, not connected, to a sewer system will discharge untreated wastewater directly into the nearest stream. 3. In-stream processes convert and store anthropogenic substances. 4. Drainage ditches along fields can drain part of the subsurface flow, changing it to longitudinal, surface flow. 5. Seepage of groundwater will release nutrients and other substances into the stream providing its baseflow.

C. Catchment under human procures during wet periods. 1. Sewer overflow devices become active during heavy rain events, shortcut flow pathways and release untreated wastewater in other, often smaller, streams. 2. Impervious areas, not connected to sewer systems enables run-off of contaminants to the lower positioned streams. 3. On arable land, part of the nutrients infiltrates into the groundwater. However, on waterlogged soils or soils with low infiltration capacity, run-off will carry nutrients, sediments and other loads directly to ditches and streams.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1.2 Catchment boundaries

Changes in flow pathways have resulted in a hydrological system with blurred boundaries, which is difficult to comprehend and manage (Lookingbill et al., 2009). Yet for a system to be distinguishable, it inevitably requires the drawing of boundaries (Cilliers, 2001). Traditionally a watershed or catchment is an area of land that captures water in any form, such as rain, snow, or dew, and drains it to a common water body, e.g., stream, river, or lake. (DeBarry, 2004). Usually models that derive catchment boundaries assume that all rainwater runs downstream and no alternative pathways are present (Jenson et al., 1988; O'Callaghan et al., 1984). These models fail to cope with natural lateral flow pathways related to groundwater flows and the observed anthropogenic pathways within heavily altered systems such as the Nete catchment. To delineate these catchment boundaries accurately, alternative calculations are required (Hammond et al., 2006; Jankowfsky et al., 2012). Current scientific research still focusses on these effects in heavily urbanized systems (e.g. Kayembe et al., 2018), while the results in chapter 2 signify also its importance in peri-urban catchments. As peri-urban area might become the dominant urban land cover in the future (Ravetz et al., 2013), more research is needed regarding the impact of this land cover and associated sewer systems on catchment delineation and dynamics.

In heavily adapted systems, such as the Nete catchment, the integration of the sewer system required the manual integration of its flow pathways (Chapter 2, 3 and 4). In our analysis we made use of existing spatial data to identify the location of areas connected to sewer systems. Our results show that the evaluation and integration of these flow pathways can significantly change the extent of the catchments boundaries and the upstream area at different locations along the river systems. Sewer systems not only transport wastewater within the catchment, but also import and export water depending on the location of the wastewater plants. As a result, catchments are connected with satellite catchments, which are located outside its natural boundaries (Figure 7-2). In these satellite catchments wastewater, but also rainwater from impervious areas, is collected and transported to WWTPs located outside the natural catchment.

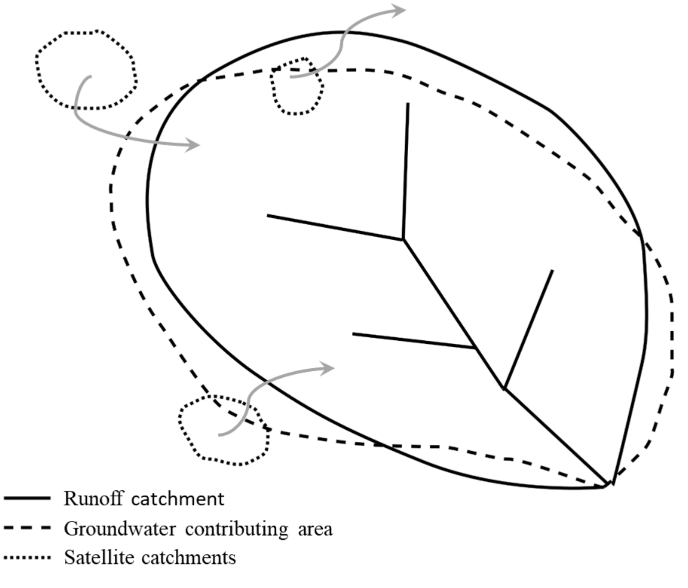

Figure 7-2: Figure of the natural, runoff boundaries of a catchment, its groundwater contributing area and different satellite catchments which import and export water towards and from the catchment (original drawing).

While natural catchments have relatively stable catchment boundaries, the presence of artificial flow pathways creates a more dynamic catchment delineation. Connectivity between the natural catchment and its satellite catchments can change, depending on, for example rain events. Sewer systems are constantly expanded due to urban sprawl, but also renovated by removing leaky pipes or changing combined sewers in to separate rain and wastewater systems. When these changes take place in cross-boundary wastewater collection regions, they can impact the catchment delineation, flow dynamics and other catchment processes by displacing rain and wastewater. Although these changes are usually small, the continuous expansion and renovation of the sewer systems might have distinct effects on how the catchment functions.

As illustrated in Chapter 2, connections to canals can also remove significant parts from the natural river system and reshape the catchment. On a smaller scale, drainage ditches and small streams are often connected to the sewer system. This practice is especially common for many urbanized areas, where streams have been piped and buried (Elmore et al., 2008). In our calculations we did not take these into account, but it is evident that these many artificial connections lead to a catchment with virtual black holes (where precipitation disappears) and small satellite catchments that bring in water flows from outside the natural catchment. To what extent these satellite catchments are present within the Nete catchment is unclear and requires further research.

1.3 Upstream indicators and land use effects

In the Nete catchment land uses affect both hydrology (Chapter 2) and river geochemistry (Chapter 3 - 4). Of all the upstream land use indicators assessed within the different analysis, the impact of houses and related sewers is best evidenced by its impact on hydrology as well as a wide range of WQ parameters.

The sewer system causes both hydrological and biogeochemical changes to the water system. But such interactions are also affected by other seasonal processes and land uses. In summer, during low flow periods, the relative contribution of the sewer system towards the river flow increases (Chapter 2). This coincides with an observed stronger relationship between the connected buildings and NO_3 in summer (Chapter 3 - rho 0.2 in winter and rho 0.5 in summer). These results do not relate to higher concentrations per se, but to a stronger relationship between explaining indicator and the variable. These higher rho-values are probably a combined effect of higher relative contributions from the sewer system for both flow and loads, compared to the overall catchment budget.

In winter however, rho values for NO_2 are higher compared to summer. In summer these lower values are probably due to in-stream processes which transform NO_2 to NO_3. However, these transformations do not occur equally throughout the system. In general, we can expect the impact of the sewer system to increase in summer due to higher relative contributions in both flow and loads. However, some of these effects are obscured due to in-stream processes and other seasonal patterns which occur patchy throughout the catchment.

The absence of significance or lower explanatory value of grasslands and cropland towards the different WQ variables was unexpected (Chapter 3 - 4). In many previous studies the impact of these land uses was clearly demonstrated (e.g. Dodds et al., 2008; Meynendonckx et al., 2006)). Our results can be an effect of both insufficient indicators as well as additional ecological processes. We made use of comparable upstream land use indicators, and applied this to a 10-yr dataset. Agricultural use can differ between years since arable land is frequently converted to grassland and vice versa. This can affect the accuracy of the indicator. The poor performance of the agricultural indicators might also be a result of changes in land management throughout the analyzed period and legacy effects. Certain management practices (e.g. manure) may still affect river processes and/or spatial differences in various ecological processes. Denitrification and N-storage maps were developed in Chapter 5. They illustrate how the intensity of such processes can have strong spatial differentiation throughout a catchment.

Understanding to which extent the indicator development is responsible for these results and what the actual impact of these land uses are on a catchment scale requires additional research.

Calculating interannual changes in upstream land use indicators can provide valuable information to understand changes in the river system. As our results show, indicators should not only encompass land use changes, but also changes in flow pathways. In Chapter 1 we found significant decreases in upstream area and effective impervious area (EIA) for many of the subcatchments, due to sewer system mediated water transfers. This may also signify that there are profound changes in groundwater flows and related biotic and abiotic processes. Decreases in EIA can signify decreases in infiltration, seepage and river flow. By calculating these indicators over a longer time period, systemic changes of the catchment might become apparent. A better integration of these issues in future studies can significantly improve the reliability of these results.

1.4 Spatial scales and complexity

Both hydrology and WQ vary along different spatial scales within the Nete catchment. These spatial scales were defined in Chapter 4 by the actual distance between locations (multi-directional) as well as the flow distance (directional) along the river network. Considering the upstream-downstream connectivity of a river network, the latter was expected to be more influential. Nevertheless, spatial structures selected in Chapter 4 were all multi-directional.

Why these spatial scales are found to be multi-directional and not upstream-downstream is, considering our other results, difficult to explain. The size of the wastewater collection regions (Chapter 2) signifies the large scale at which anthropogenic flow pathways can operate. Changes in upstream area and urban infrastructure are found in both Chapter 2 and 3 and can be related to a wide range of WQ parameters in Chapter 3 and 4. These changes where expected to have important downstream effects. Nevertheless, we were not able to find any of the directional structures to be significant. One of the possible explanations is that many of the processes taking place and pollution sources located along the river continuum are highly intermittent and could not be represented by the structures generated for the analysis. For example, Kaushal et al. (2014) have shown how in-stream retention and release (0–100 %) of watershed C and N loads can change over the scale of kilometers. These small scale processes can strongly impact the water quality within specific reaches.

The structures that were found to be statistically significant, illustrate how water chemistry can vary over small, medium and large distances. This suggests that also natural lateral or vertical connectivity through groundwater impacts the river system over small and larger distances. Groundwater discharges can provide a more-or-less identical input of water and nutrients along stretches of the river system and nearby river systems. As these signals can be sampled before they are changed through instream processes, our statistical analysis could find significant structures. The results point out that lateral flows take place over a wide range of distances and add an additional level of complexity to water quality patterns.

Ecosystem complexity is defined by heterogeneity of the system, connectivity, and history (Cadenasso et al., 2006). Understanding these complex relationships in a catchment perspective is not only a problem within the Nete catchment, but remains a challenge globally (Lintern et al., 2018). Our results illustrate how both heterogeneity (e.g. different land uses) and connectivity (e.g. changes in flows) can lead to a more complex system that changes over different scales. Although our analyses do not incorporate an historical investigation, it is expected that legacy effects (e.g. nitrate leaching to groundwater) will also have an impact on the investigated WQ patterns within the Nete catchment. A better understanding of these different aspects can help us to better predict and manage riverine water quality.

1.5 Temporal variation

Chapters 2 to 5 evaluate both hydrology and water quality over different time periods, with a main focus on interseasonal and interannual changes. The presented results illustrate how different processes can effect these river characteristics differently in-between seasons. But also how they can change over the years as results of differences in weather patterns and changes in land use. Therefore, to understand which processes and relationships are predominant, research needs to be done over longer periods of time.

Although most of the presented research focusses on seasonal and interannual effects, both hydrology and WQ can change over much shorter time periods. Different water quality parameters such as nutrients vary throughout the day as natural in-stream processes change with moving daily light and temperature patterns (Nimick et al., 2011). These dial patterns of increased activity can be combined with daily patterns from different pollution sources such as wastewater plants or not-connected houses (e.g. Gammons et al., 2011).

In our research we were not able to assess the impact of these dial patterns as no information is available regarding the sampling time of the WQ data. Monitoring data used in Chapter 3-5 are sampled monthly over several days following fixed sampling routes. As a result, some of the sample points are consistently sampled in the same period of the day, incorporating these dial patterns within the datasets. Patterns and processes that are present in the catchment might be, partly, obscured by this. As for example two locations downstream of forests, sampled one in morning and one in the afternoon, can give different WQ patterns.

Each year other sample points are assessed within the Nete catchment, changing sampling routes and the moment each location is sampled. Many of the relationships found in Chapter 3 are consistent throughout the years, signifying their importance, despite these dial variations and changes in sampling routes. Nevertheless, we probably missed some land use – WQ relationships in Chapter 3 or spatial scales in Chapter 4, that are strongly defined by this dial variation. In order to better understand these effects, more detailed sampling data is needed with several samples for each SP throughout the day.

1.6 Implications for concepts

River systems have been described through a wide range of theoretical concepts, such as the River Continuum Concept of Vannote et al. (1980), which emphasize upstream-downstream connectivity. In Chapter 4 no statistical evidence was found for clear upstream-downstream relationships throughout the Nete catchment. These concepts, describing many natural river processes, seem to be outdone through anthropogenic changes, making them useless within the Nete catchment regarding the WQ.

Kaushal et al. (2012) described the Urban Watershed Continuum concept, by which different catchment characteristics are being related to urban development, such as specific changes in hydrological connectivity and flow interactions. The Urban Watershed Continuum (UWC) applies the 4 dimensions from Ward (1989) (Figure 7-3A) to an urbanized system (Kaushal et al., 2012) (Figure 7-3B). The different aspects put forth by both Ward (1989) and Kaushal et al. (2012) can be identified in the Nete catchment. Although the UWC was in first instance developed for strongly urbanized catchments, it seems also to be applicable to peri-urban catchments, such as the Nete catchment, as well. However, the interaction between the natural and urban hydrological system, makes these concepts separately inadequate in representing an actual peri-urban catchment. In reality any location along the river reaches is impacted by a combination of both types of longitudinal, lateral and vertical dimensions (Figure 7-4). As such both concepts need to be integrated. The interaction between, and importance of ,these flows changes over time both seasonally (Figure 7-3 C and D) as well as over longer time periods due changes in landscape composition, vegetation transition and changing urbanization. The relative weight of the lotic longitudinal dimension increases further downstream. However, this dimension incorporates flow dynamics and loads from the other, upstream located dimensions.

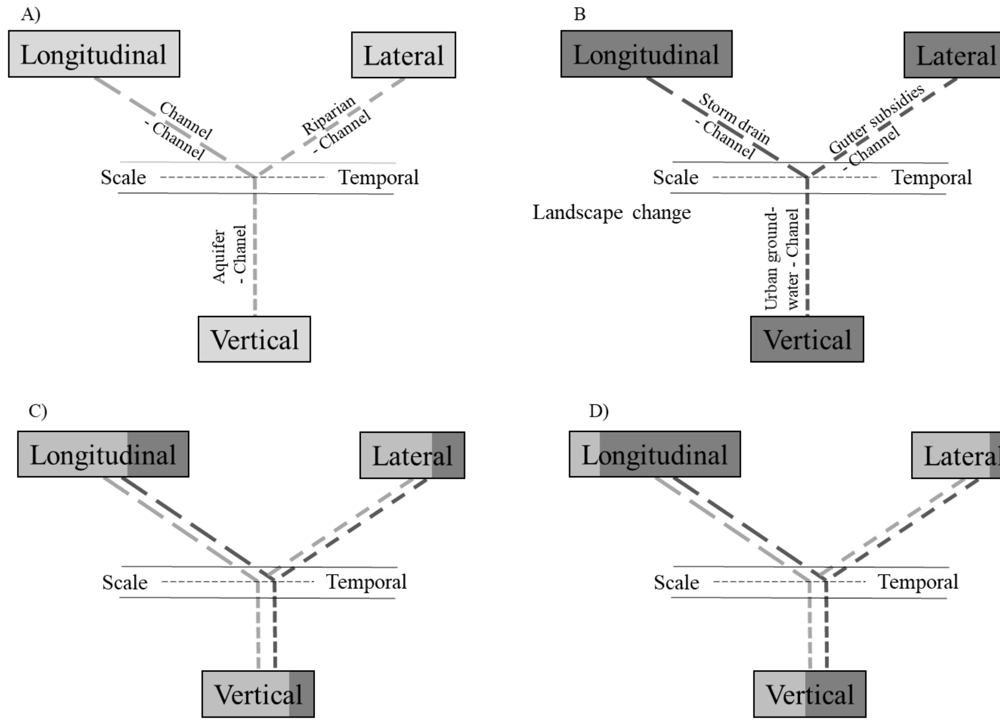

Figure 7-3: A) Conceptualization of the four-dimensional nature of lotic ecosystems (adapted from Ward (1989)). B) Conceptualization and interpretation of the four-dimensions described by Kaushal et al. (2012) in the Urban Watershed Continuum. Both concepts can be integrated into one scheme, where the relative importance of natural (light grey) and urban system (dark grey) can be given in color shading. As such changes in these relative importance due to e.g. dry or wet periods can be illustrated (C and D).

The 4D scheme can be integrated along the river continuum where the relative importance of the different dimensions is given for each of the different reaches (Figure 7-4).

Much of the literature on hierarchical stream classifications describes stream networks as completely nested hierarchies, wherein all river reaches, pools and segments are equivalently nested within a broader catchment area (Melles et al., 2012). However, our results illustrate how in human disturbed catchments nesting and hierarchy is changed due to urban hydrological infrastructure and how hydrological interactions can change between dry and wet seasons. As a result, many of the existing stream classifications cannot directly be applied to heavily altered, complex catchments. Within geomorphology, one of the most basic stream classification was developed by Strahler (1957). While this classification method can work in pristine catchments, urbanization and other changes can significantly change the Strahler classification throughout the catchment (Figure 7-5). While Chapter 4 has shown how these upstream-downstream concepts are outdone in a peri-urban catchment, this simple example illustrates on a basic level the importance of a good incorporation of anthropogenic developments in any applied stream classification method.

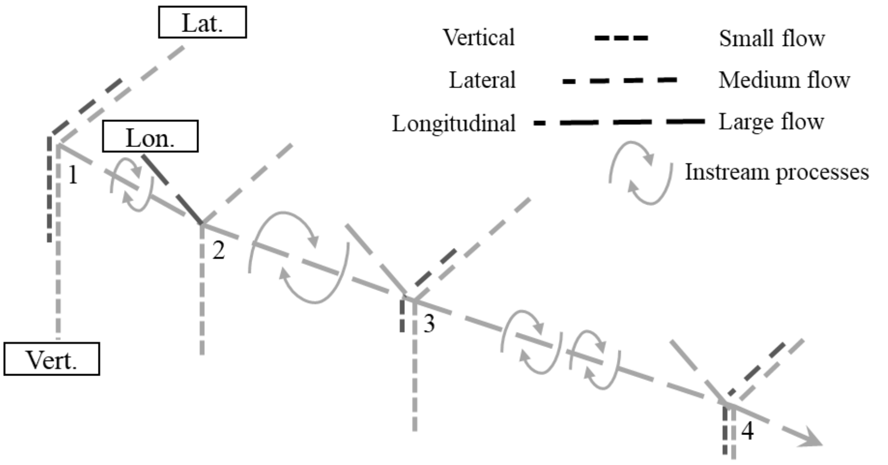

Figure 7-4: Along the river continuum the relative importance of the different dimensions can change significantly. In headwaters (1) the hydrological regime is mostly determined by the vertical and lateral dimension, while further downstream (4) the longitudinal dimensions will become dominant as a mixture of upstream flows. Both natural (light grey) and urban (dark grey) dimensions impact the system. Not all dimensions need to be present in all reaches: e.g. WWTP (3) or (4). Instream processes impact and change this mixture of flows/dimensions throughout the catchment in different ways (original drawing).

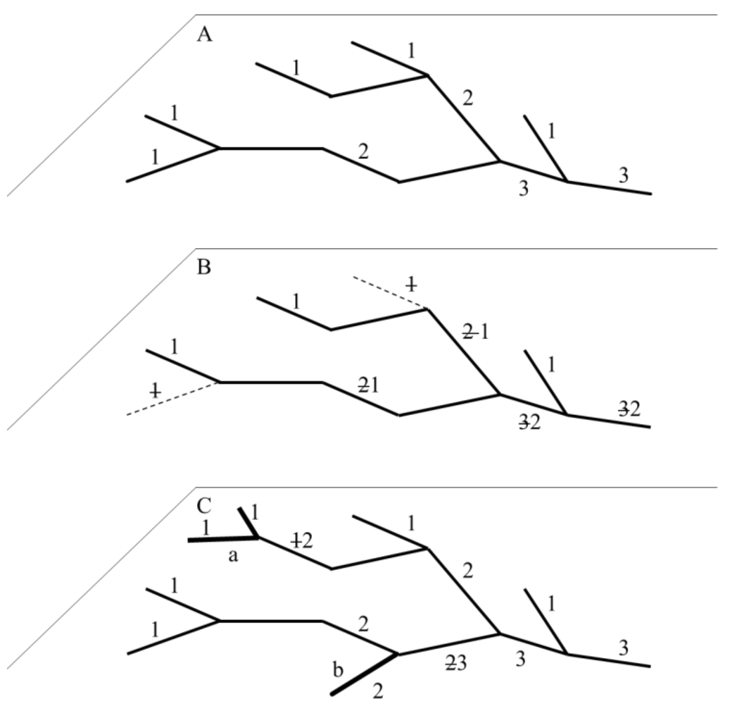

Figure 7-5: A) A classic Strahler calculation for a natural stream network. B) Incorporation of headwaters into the sewer system decreases the Strahler order throughout the catchment. C) a. Additional drainage ditches in agricultural areas can increase Strahler values. b. Discharges from WWTPs are not recognizable as reaches but its impact can have the size of a sizeable stream – higher Strahler reach.

1.7 Implications for research methodologies

The outcome of a model or statistical analysis of a catchment depends on the correct integration of flow pathways. Therefor careful consideration on how these flow pathways are incorporated within the study is of utmost importance. In chapter 2 we applied a type of hydrological model that is often used to assess land use change scenarios. These models have however limitations to how sewer systems and related land uses can be incorporated. A realistic representation of the flow pathways by the model is a constraint to derive reliable results that can be used for hypothesis and scenario testing. The same conclusion can be made on the application of the upstream land use indicators. They are often presented as easy to use indicators for river health. However, in complex catchments the correct integration of the flow pathways in the indicator calculation is not straightforward and time consuming. Nevertheless, these investments in time are needed to come to accurate and reliable indicators that can be used in research and management.

The scale of application also determines the suitability of a methodology. While management takes place on the catchment level of the Nete, many studies are difficult to apply on such a scale. The hydrological model used in Chapter 2, cannot be applied on a level of the Nete without making compromises towards the spatial and temporal resolution. These changes in their turn undermine the reliability of the results, making the model ineffective. Other hydrological models do exist for the Nete catchment, but are also limited in spatial extent (Dams et al., 2008; Vansteenkiste et al., 2014). Therefore, the hydrological functioning in chapter 2 was evaluated on only 20% of the Nete catchment. In Chapter 3 and 4 we make use of water quality datasets from existing monitoring networks of the Flemish Environmental Agency. However, in Chapter 4 the sample point density and number of WQ parameters was increased by taking additional samples on key locations and analyzing all samples, both from the Flemish Environmental Agency and our own, on additional parameters (n = 10). To make this additional sampling feasible, the sampling campaign targeted only part of the Nete Catchment.

Throughout the chapters, we had to make pragmatic choices, concerning research questions, study sites and applied methodologies. Despite the complementarity of each chapter, it was impossible to perform each analysis on a full scale catchment level. Even though the results on a Nete catchment level would be more suitable for management application.

2. ECOSYSTEM SERVICES AND FLOWS

2.1 Supply and demand

Ecosystem services (ES) are the used contributions of ecosystem structures and functions to human well-being (Daily, 1997; MEA, 2005). The ES concept offers a valuable methodology to link society to nature and reinforces arguments to conserve and restore ecosystems, while providing a common language for managers, researchers and different stakeholders (Granek et al., 2010). Together with the development of the theoretical framework, quantification and mapping of both ES demand and supply have become important topics within the ES research field.

Over the past years several classifications have been developed which try to categorize the wide range of ecosystem services. In chapter 6 we used an international classification system, while we made use of ES definitions which are tailored to the Flemish situation in chapter 5. Many authors argue for the use of one international system to improve the scientific comparability of the different studies such as CICES (Haines-Young et al., 2013). However, from our experience these systems do not always fit the local situation and adapted ES definitions can improve its applicability. Therefore, depending on the research area, goal of the analysis and available knowledge different types of classification should be used.

These definitions need to be aligned with how ES demand and supply are defined. In chapter 5 demand is looked at from an institutional, legislative perspective, while in chapter 6 demand this is done from a local stakeholder perspective. As a result, the ES type that is considered and how the demand for the ES is defined differs considerably. When ES are to be evaluated the goal of the study, target audience etc. need to be clearly delineated, before ES classification, demand and supply are considered.

In chapter 6 we explored the demand and supply for a wide range of ecosystem services in a data-scarce region in Uganda. A scoring system developed by Burkhard et al. (2009) was used as a basis to assess the ecosystem supply and demand flows within the region. The scoring system allowed us to assess a range of ecosystem services despite lacking local data. However, the reliability and applicability of the methodology remains uncertain. In first instance, the aim of the results is to illustrate the importance of natural resources to the local communities. But communication of these results to locals remains challenging considering the poor development of the region. How the results can support actual local decision making is yet unclear.

When good quantitative data are available or can be compiled, the ecosystem services scoring method should be avoided. Instead, quantitative methodologies or qualitative derivatives of the quantitative data should be considered. In Chapter 5 we quantified the additional demand for water quality regulation along the services flow paths. To map both supply, additional supply and demand we made use of data with a high spatial and temporal resolution. But validation of both supply and additional demand indicators remains challenging. The complex spatial setup of the area for which the indicators were developed counteracts with the large amount of available data. Therefore, additional research and field sampling is needed to verify the accuracy of the developed indicators. Quantification of ecosystem services does not necessarily ensure reliable results and the obtained indicators should be handled with care. A good balance between data quantity and quality and spatial and systemic complexity (spatial and temporal) of the analyzed system is needed to gain sound results.

2.2 Flow pathways and connectivity

The presence of both demand and supply of ES within an area does not necessarily ensure the delivery and use of these ES when both areas are spatially separated from each other. The actual delivery of ES strongly depends on the connectivity between and by the availability and type of different ES flows and pathways. Understanding how these pathways work and whether they are present is of utmost importance for a reliable ES assessment.

Besides understanding which flows are of importance, we should also be able to incorporate them realistically within a modelling environment. In chapters 2 to 4, the significance of the different water flow pathways for water quality regulation within in a peri-urban catchment was demonstrated. These flow paths were only partly integrated in the ES demand analysis in chapter 5. But even then the integration of these pathways within the ES analysis and indicator development was challenging. The development of a system which allowed for the calculation and movement of indicators along the river network happened to be time consuming due to the complexity of the river network with its weirs, culverts, channels, etc. The development of the indicators themselves also proved to be challenging. The development of the indicators itself, to make them applicable to different situations such as the presence of a WWTP, also proved to be difficult, hampering their reliability and the reducing the comprehensibility of the results.

In Chapter 6 we implemented different ES flow pathways as proposed by Fisher et al. (2009) to evaluate a wide range of ecosystem services in a data-scarce region. Our methodology allowed for the routing of simple indicators along different potential flow paths. By evaluating the connectivity between the different supply and demand areas, the spatial discrepancy between both could be made visible. Although significant simplifications of the actual situation were made, the results can help to illustrate and communicate the dependence of local residents to the remaining natural areas, that can be located nearby or further away.

In both Chapter 5 and 6 different methodologies were used to incorporate flows in an ecosystem service assessment. Goal and location of the study; communication, management, …, define the type of methodology and the level of detail that needs to incorporated. Understanding which level is adequate and whether the main flows are well represented in the calculations is critical for a reliable assessment.

3. INTEGRATED MANAGEMENT

3.1 Manage upstream-downstream impact

As shown in Chapter 4 (and also discussed in section 1.4), the spatial structures found for the Nete catchment's water quality are multi-directional. Nevertheless, upstream-downstream relationships in WQ variation are undeniably present in the Nete catchment. The river network still brings polluted water downstream. But changes in WQ are highly variable and take place in a chaotic, non-structured way along the river continuum. As such, it would be more likely that the methodology presented in Chapter 5 is able to deal with the WQ variability within the Nete catchment, than large scale, more general WQ models. Despite this observation, it is unclear to what extent the obtained results are suitable for the planning of actual management actions. Different methodological challenges clearly remain as the resulting indicators can differ strongly between years and the amount of information required is very high. Even if the presented methodology can be made more robust, additional analysis on a smaller, subcatchment scale remains indispensable to confirm the validity of the results. These would be needed to underpin implementation of actual management actions, such as ES restoration.

Despite these considerations, it is clear that the upstream downstream analysis from Chapter 5 is partly inconsistent with the results of Chapter 4. The analysis in Chapter 5 only takes place along the river network focusing on the longitudinal connectivity. Our results of the spatial structures in the Nete catchment evidence the importance of a lateral connectivity which is probably related to groundwater flows that vary over different spatial-temporal scales. To improve upstream-downstream management of the catchment, disentangling the relationships of the different flow-paths and understanding their relative importance is key. To improve overall river conditions, management actions should be taken on locations that related to the relevant pollution sources and flow paths. Especially the effects of diffuse pollution sources, which are often difficult to manage, should be better understood.

Chapter 6 demonstrates how the lack of available information puts restrictions on how an upstream-downstream analysis can be done. Although upstream-downstream connectivity is analyzed, the results obtained from the analysis have little explanatory value. Data availability and the spatial complexity of the river system can both limit the applicability of upstream-downstream analysis and their relevance towards catchment management. The usability from a management perspective is restricted due to the relatively low spatial and temporal resolution of data in respect to the spatial and temporal complexity of the system. Nevertheless, it remains relevant as a communication tool to illustrate the importance of upstream-downstream interactions in integrated management assessments.

3.2 Decision support system

Many different decision support systems (DSS) have been developed to help in the implementation of the Water Framework Directive (WFD) (de Kok et al., 2009; Giupponi, 2007; Holzkämper et al., 2012; Maurel et al., 2007). But one of the problems is that many of the DSS's used for the WFD have been developed for specific aspects and do not cover the broadness of the WFD. These systems are very difficult to adapt to changing conditions. Tool development and catchment planning should be brought much closer together (Gourbesville, 2008). Although the initial aim of Chapter 5, was to develop a broad DSS that would address these issues and make broad upstream-downstream analysis possible, the eventual methodology is limited to the part of the WQ management within the catchment.

The absence of reliable flow data for sub-catchments hampers the development of a reliable DSS. In our analysis indicators were developed that can be applied on the river network, without the need for flow data. However, the reliability of these results is questionable. To develop indicators that can incorporate information for other catchment functions, accurate flow data seem to be indispensable. As indicators are propagated upstream in the river network, information is partitioned across the many upstream tributaries. This partitioning is almost impossible without flow data as it is unclear how much each of the tributaries contributes to the receiving stream.

However as demonstrated in chapter 2, hydrological modelling faces its own challenges. Although our knowledge regarding the impact of urbanization on the rivers hydrology continues to improve (e.g. Bhaskar et al., 2016; Diem et al., 2018), applying hydrological models in an urbanizing environment remains challenging. To get reliable hydrological data, natural and engineered water cycles need to be integrated into these hydrological models, which require huge amounts of monitoring data. Nevertheless, steps are taken to gradually overcome these issues (e.g. Hutchins et al., 2017) and to better integrate impervious areas and sewer systems in hydrological models (Gires et al., 2017).

Despite this progress, we can wonder whether reliable flow data will ever become available at the required spatial and temporal resolution. With the current knowledge and data available, full-scale ICM analyses seem to be impossible in catchments with a complex interplay of natural and anthropogenic land uses, processes and hydrological pathways. Serious technical hurdles should first be solved before this kind of overarching analysis can become available. Therefore, current practices of case specific problem solving seems to be the most effective.

3.3 Monitoring design and data availability

Data availability promotes scientific research and improves our understanding of catchment dynamics. All data used for the analysis in the Nete catchment are today freely available. This allows researchers to combine datasets from different providers and to analyze catchments on a scale and time horizon that would be impossible with datasets originating from standard scientific research programs. However, data availability still determines the scale and detail in which hypothesis can be tested. Current sampling designs, set up by government agencies, are mainly designed for monitoring, management and reporting, which makes the datasets (partially) unsuitable for statistical or model in debt analysis of the system.

Current sampling design is mainly used to monitor specific parts of the river system or pollution sources and are not designed to get an in-depth understanding of the system at different scales. As such these networks hamper integrated management as (parts of) the monitoring system breaks the catchment down into small pieces which are then management individually based on the acquired data. To move to genuine integrated management, monitoring systems should be designed towards this goal. As such sampling design should not only focus on problem solving, but also on system understanding. This would require a sampling design that monitors the catchment at different scales and time intervals. Sample points should not only be located near pollution sources or along polluted streams, but also on natural reaches to better understand the natural dynamics of the system throughout the years. As mentioned before in 0 sampling should also acknowledge the time scales at which the WQ parameters change. Therefore, sampling frequency should be in line with the most important time scales, being able to capture all relevant variation.

Compared to the analysis on the Nete catchment the lack of data in Uganda (Chapter 6) provided an enormous challenge. Although local knowledge can provide useful insights, which often cannot be captured by measurements, many other processes over larger scales are not well understood or are unknown to local residents and managers. Although some datasets do exist in Uganda, almost none of them were available. Compared to the freely available data used in the other chapters, this clearly restricted the options for analysis. Even in data-scarce regions, a minimum of data is required to make the implementation of integrated management feasible.

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