




Modelling tools for cost-effective water management

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Summary

The region of Flanders, Belgium is a region facing a whole range of significant water management issues. Key issues are a bad surface water quality and groundwater quality (nutrients, chemicals), increasing flood risks (sea level rise), sediment management (dredging and processing polluted sediments), hydromorphology (a heavily modified water system) and droughts (surface water & groundwater quantity issues in specific areas). Setting up programmes of measures which are able to solve different water management issues at the lowest cost achievable is one of the key objectives of water management in this region during the upcoming decades.

Economic appraisal techniques can be important decision support tools leading to more integrated and cost-effective water management. The European Water Framework Directive (WFD) explicitly integrates economic analysis in both water management and water policy decision-making in Europe. To achieve the environmental objectives for our water systems, the Directive calls for the application of economic principles (e.g. the polluter-pays principle), economic approaches and tools (e.g. cost-effectiveness analysis) and instruments (e.g. water pricing) (WATECO, 2003). Whereas the generic principles on how to perform economic appraisal techniques are clear and well documented, the application for water management is less straightforward. It requires the use of both economic appraisal techniques and hydrological modelling tools at different spatial and time scales.

The general objective of this research is to develop and apply modelling tools that can assist policy makers to compose cost-effective programmes of measures for water management. Important for these tools is that they are on the one hand applicable for decision making on a national or regional level (macro-scale), but on the other hand also sufficiently detailed to correspond with the choices policy makers are faced with on the local project level (micro-scale). This objective is being achieved in several chapters focusing on surface water quality and flood risk management.

A first application focuses on cost-effectiveness analysis and the development of an economic optimization model for surface water quality. The research objective is to develop a model that contrary to most other existing economic models works at the same scale as hydrological models to allow for easy exchange of scenarios between hydrological and economic models, can be used for local and regional scales and includes upstream-downstream interactions. The model was specifically built to demonstrate the importance of scale and potential cost savings due to a spatial diversification of the programme of measures. To deal with the differences in scale, typical optimization algorithms are extended with the possibility to include both individual measures aimed at reducing individual sources and collective measures which are implemented at a higher scale and can reduce multiple sources simultaneously. This model set-up is applied for surface water quality in the entire Flemish region. The economic optimization model minimizes costs in function of specified emission reduction targets. The model allows to perform calculations for individual waterbodies, basins and the entire region. The impact of the reduced pressures on the surface water quality status due to different programs of measures is simulated with the water quality model Pégase. The difference in results for individual subbasins confirms the relevance of cost optimization at local scales (waterbody, subbasin). If the programme of measures, which was designed uniformely for the entire Flemish region in 2009, would be optimized at the scale of individual waterbodies, the

annual costs can be reduced with 22% to achieve similar emission reductions in every waterbody. If emission reduction targets are not restricted to the waterbody itself but can also be realized in upstream areas, annual costs can be reduced with 33% compared to the uniform approach.

A next chapter focuses on the integration of an economic optimization model and a hydrological model to assess the cost-effectiveness of measures. The objective is to determine cost-effective sets of measures to reach in-stream concentration targets instead of emission reduction targets as performed in the previous chapter. To realize this objective a modular modelling approach is applied that determines the most cost-effective set of reduction measures to reach an in-stream concentration target. The framework is based on the coupling of two models: the hydrological water quality model SWAT (reference) and an economic optimization model (Environmental Costing Model, ECM). SWAT is used to determine the relationship between the modelled in-stream concentration at the river basin outlet and the associated emission reduction. The ECM is used to determine the cost-effectiveness of measures. This model set-up can deal with smaller time scales and daily variations in hydrological conditions which is different compared to other existing applications for cost-effectiveness analysis. This allows to optimize towards other types of objectives as summer average or 90 percentile concentrations, instead of yearly averages. Daily variations in hydrological flows and different impacts of hydrological conditions on point and diffuse source emissions might influence the cost-effectiveness of measures. The results confirm the relevance of including the results from the hydrological model in the cost-effectiveness analysis. Rankings of measures to reach in-stream summer average concentration targets for N or 90 percentile concentration targets for NO_3^- are very different. Summer average concentrations are more influenced by reducing point sources (WWTP, households and industry) and peak concentrations, often experienced during winter periods, are more influenced by reducing diffuse sources (agriculture).

A cost-benefit analysis to determine spatially diverse flood risk management plans for the tidal Scheldt estuary is a third application. This means costs of programmes are no longer minimized towards reaching predefined target levels but risk based approaches are applied whereby specifically for different locations costs of measures are balanced with the benefits (flood risk reduction and other environmental impacts) they achieve. Scenario development is performed by iterative feedback loops between flood risk simulation models and economic analysis. Spatial interdependencies are dealt with by a stepwise scenario development. The study area is subdivided in 5 subzones and the optimal scenario in subzone 1 is the starting point for subzone 2 and etc. The results indicate the added value of modelling tools, both economic appraisal techniques and hydrological models, for integrated water management. Especially a localized optimization of measures based on actual flood risks instead of fixed safety targets can realize much higher net societal benefits compared to a large scale implementation of a storm surge barrier or dyke heightening. Besides effects on flood protection, the impact on other ecosystem services (water quality, sediment management, recreation and climate change regulation) are also valued as part of the cost-benefit analysis. The valuation of these additional services demonstrates the potential added value of constructing reduced tidal areas instead of traditional flood control areas. In reduced tidal areas water flows in and out the area during normal tidal cycles through well designed culverts. This also means the area is used as a nature conservation area. This has a positive impact on water quality, sediment management, recreation and climate change regulation.

Further refinements on the valuation of ecosystem services are performed in a next chapter. The objective is to develop a valuation tool that is able to deal with a range of ecosystem services that can be created by land and water management. To use this framework for location specific optimization of measures, it is crucial to take into account the major characteristics of project sites influencing the value of the services they deliver. In the previous chapter unit value Benefits Transfer techniques are applied. More specifically the value of an ecosystem service at the project site is estimated by multiplying a mean unit value estimated at another study site with the size of the area. This is sufficiently detailed to demonstrate the added value of reduced tidal areas compared to traditional flood control areas, but is insufficiently detailed to determine where the construction of a reduced tidal area would generate the highest benefits. In this chapter value function transfer methods are applied. This means that more characteristics of the project site (e.g. soil, surrounding land use, population living nearby, sociodemographic factors of the beneficiaries) are taken into account to transfer values from other study sites. The results of three case studies demonstrate the large differences in the value of ecosystem services created and the added value of value function transfer methods.

A last chapter describes how data and results from hydrological models and economic appraisal techniques can be integrated and made available for end users in decision support. The objective is to provide a synthesis of information from monitoring, field studies and models on different scales and water aspects. This allows end users to consult information from monitoring campaigns and modelling tools as developed in the previous chapters. To realize this objective, a web-based decision support tool was developed to provide the necessary data to assess costs, effects, benefits and affordability of packages of measures. Information about status, pressures, costs and effects of measures can be retrieved and simulation results can be generated on different scales, from individual waterbodies to regional level. End users can build up draft packages of measures (scenarios), assess their costs and effects and share these scenarios with other users (e.g. users building scenarios for other aspects or for other waterbodies). This tool is currently being used to perform a desktop screening of waterbodies, to prioritize which waterbodies to target first and to perform a disproportionate costs analysis for the second generation river basin management plans.

In general, this research confirms that tools can be developed to determine cost-effective programmes of measures in Flanders. Results of these tools confirm the added value and potential cost savings which can be realized by more locally targeted water management.

Finally some issues for further scientific research were identified in this study: (1) predicting the long term impact of measures and the consequences for cost-effectiveness analysis (2) the importance of assessing cost-effectiveness across water aspects and (3) assessing and communicating uncertainty to end users.

Samenvatting

Vlaanderen als regio wordt geconfronteerd met een hele reeks van waterbeheerskwesties. Belangrijke kwesties zijn o.a. oppervlaktewater- en grondwaterkwaliteit (verontreiniging door nutriënten, prioritaire stoffen), een toenemend risico op overstromingen (mede onder invloed van de zeespiegelstijging), sedimentbeheer (baggeren en verwerken van verontreinigde waterbodems), droogte (tekort aan oppervlaktewater of grondwater op specifieke locaties) en hydromorfologie (herstellen van natuurlijke condities in een sterk veranderd systeem). Het opstellen van kosten-effectieve maatregelenprogramma's die in staat zijn om die kwesties geheel of gedeeltelijk op te lossen is hierbij een belangrijke stap.

Economische beoordelingskaders kunnen belangrijke beslissingsondersteunende instrumenten zijn die bijdragen aan een meer integraal en kosteneffectief waterbeleid. De Europese kaderrichtlijn water (WFD) beschouwt expliciet economische analyses als een essentieel onderdeel in waterbeleid in de hele Europese Unie. Om de milieudoelstellingen te bereiken voor onze watersystemen, schrijft de richtlijn voor om economische principes toe te passen zoals het "vervuiler betaalt"-principe, economische analysekaders te gebruiken zoals een kosten-effectiviteitsanalyse en economische instrumenten te verbeteren zoals de prijszetting van water. Hoewel de generieke methodes voor deze analysekaders wel gekend zijn, blijft de toepassing minder voor de hand liggend. Het gebruik van economische afwegingskaders en hydrologische modellen op verschillende tijd- en ruimteschalen is hiervoor noodzakelijk.

De algemene doelstelling van deze thesis is modellen en instrumenten te ontwikkelen en toe te passen om beleidsmakers te ondersteunen bij het samenstellen van kosten-effectieve maatregelenprogramma's voor waterbeleid. Belangrijk hierbij is dat de modellen enerzijds geschikt zijn om besluitvorming te ondersteunen op nationale of regionale schaal (macro-schaal), maar anderzijds ook gedetailleerd genoeg zijn om inzichten te geven op het lokale project-niveau (micro-schaal). Integraal waterbeleid vereist immers dat tools inzetbaar zijn op verschillende schalen.

Een eerste toepassing om deze doelstelling te realiseren is gericht op kosten-effectiviteitsanalyse voor oppervlaktewaterkwaliteit. Een gedetailleerd economisch optimalisatiemodel werd uitgewerkt op dezelfde schaal als hydrologische modellen. Dit heeft het voordeel dat eenvoudig scenario's tussen hydrologische en economische modellen kunnen uitgewisseld worden. Dit model werd toegepast voor alle individuele waterlichamen in Vlaanderen. Het model is specifiek ontwikkeld om de kosten-effectiviteit te vergeljiken van maatregelen die worden geïmplementeerd op verschillende schalen. Specifiek werden naast individuele maatregelen, die worden geïmplementeerd op één specifieke bron, ook collectieve maatregelen beschouwd, die van toepassing zijn op meerdere bronnen gelijktijdig. Besluitvorming voor dit soort collectieve maatregelen gebeurt op een hogere schaal. Het economische optimalisatiemodel minimaliseert de jaarlijkse kosten in functie van opgelegde emissiereductiedoelstellingen. Het model laat toe om berekeningen uit te voeren op verschillende schalen gaande van individuele waterlichamen, tot bekkens en de hele regio. De impact van maatregelen op de oppervlaktewaterkwaliteit is gesimuleerd met het Pégase model. De resultaten van het economisch optimalisatiemodel bevestigen de toegevoegde waarde van kostenoptimalisatie op de lokale schaal. Als het maatregelenprogramma, dat voor heel Vlaanderen uniform werd vastgelegd in 2009, zou geoptimaliseerd worden voor individuele waterlichamen, kunnen de jaarlijkse kosten om dezelfde emissiereducties te

realiseren in ieder waterlichaam met 22% gereduceerd worden. Als dezelfde emissiereducties ook kunnen gerealiseerd worden in bovenstroomse gebieden, dalen de jaarlijkse kosten verder tot 33% t.o.v. de uniform Vlaamse aanpak.

Een volgend hoofdstuk gaat dieper in op de integratie van het economisch optimalisatiemodel met een hydrologisch model om kosten-effectiviteit te beoordelen. De doelstelling is om kosten-effectieve maatregelenprogramma's te bepalen om waterkwaliteitsdoelstellingen (concentraties) te realiseren i.p.v. emissiereductiedoelstellingen. De methode is gebaseerd op de koppeling van twee modellen: het waterkwaliteitsmodel SWAT en het economisch optimalisatiemodel zoals hiervoor besproken. Dit is toegepast voor de Grote Nete. SWAT wordt gebruikt om de relatie te bepalen tussen de gemodelleerde waterkwaliteit en de geassocieerde emissiereducties. Het economisch optimalisatiemodel wordt gebruikt om de kosteneffectiviteit van maatregelen te bepalen in functie van emissiereducties. De koppeling heeft als voordeel dat rekening kan gehouden worden met kleinere tijdschalen en dagelijkse variaties in hydrologische condities. Dit laat toe om te optimaliseren voor specifieke parameters en toetswijzes zoals zomergemiddelde of 90-percentiel concentraties i.p.v. jaargemiddelde concentraties. De resultaten bevestigen de relevantie om resultaten van het hydrologische model mee te nemen in de kosten-effectiviteitsanalyse. De rangschikking van maatregelen om specifieke concentratiedoelstellingen te bereiken voor de zomergemiddelde concentraties van N of de 90-percentiel concentraties voor NO_3^- zijn zeer verschillend. Zomergemiddelde concentraties worden meer beïnvloed door de reductie van puntbronnen (RWZI, huishoudens en industrie), terwijl piek concentraties vooral voorkomen tijdens de winter en meer beïnvloed worden door diffuse bronnen (landbouw). Dit heeft een grote invloed op de kosten-effectiviteit.

Een kosten-batenanalyse gaat nog een stap verder dan een kosten-effectiviteitsanalyse. Er wordt niet langer uitgegaan van voorgedefinieerde doelstellingen, maar er wordt gezocht naar de meest efficiënte maatregelenprogramma's of de programma's die de hoogste netto baten genereren. Deze aanpak is toegepast voor de preventie van overstromingen op het Schelde estuarium. Een risico-gebaseerde benadering waarbij dus niet meer vertrokken wordt van uniforme doelstellingen, maar wordt gezocht waar bijkomende maatregelen de meeste netto baten genereren, laat toe om meer maatschappelijke baten te realiseren tegen dezelfde kosten. Scenario-ontwikkeling is toegepast door stelselmatig en iteratief resultaten van de overstromingsmodellen met de economische analyse uit te wisselen. Ruimtelijke afhankelijkheden worden aangepakt in een stapsgewijze aanpak, waarbij het studiegebied wordt onderverdeeld in 5 deelgebieden. Het optimale scenario voor deelgebied 1 is het startpunt voor verdere optimalisatie in gebied 2 enzovoorts. De resultaten van de modellen tonen de toegevoegde waarde aan van dit analysekader. Een risico-gebaseerde aanpak die toelaat om ruimtelijk divers beschermingsniveaus tegen overstromingen te bereiken door de gerichte aanleg van overstromingsgebieden is in staat om gelijkaardige maatschappelijke baten te realiseren tegen veel lagere kosten in vergelijking met de grootschalige aanleg van een stormvloedkering of dijkverhogingen. Naast de impact op overstromingsrisico's wordt ook gekeken naar de waarde voor andere ecosysteemdiensten. Met name werd geschat hoe de aanleg van gereduceerde getijdegebieden bijkomende baten realiseert voor waterkwaliteit, sedimentafvang, klimaatregulatie en recreatie in vergelijking met traditionele overstromingsgebieden.

Verdere verfijningen van de waardering van ecosysteemdiensten worden toegepast in een volgend hoofdstuk. De doelstelling is om een instrument te ontwikkelen dat in staat is om een brede range van milieu-effecten te waarderen die kunnen worden gerealiseerd door land- en waterbeheer. Om een waardering van ecosysteemdiensten te kunnen gebruiken voor locatie-specifiek beleid, is het noodzakelijk dat de belangrijkste gebiedsspecifieke karakteristieken die een invloed uitoefenen op de creatie van ecosysteemdiensten mee in rekening worden gebracht. In het vorige hoofdstuk werden waardes berekend op basis van eenheidswaardes per oppervlakte-eenheid (zogenaamde Unit Value Benefits Transfer). Meer concreet werden geobserveerde waardes van een studie-site getransfereerd naar een project-site op basis van de oppervlakte. Dit is voldoende gedetailleerd om aan te tonen dat een gereduceerd getijdegebied een meerwaarde kan creëeren t.o.v. een gecontroleerd overstromingsgebied, maar is onvoldoende om aan te tonen waar precies de aanleg van een gereduceerd getijdegebied de meeste meerwaarde kan creëren. Om deze vraag te beantwoorden, wordt in dit hoofdstuk gewerkt met waardefuncties i.p.v. eenheidswaardes (zogenaamde Value Function Benefits Transfer). Dit betekent dat veel meer gebiedsspecifieke kenmerken van de project-site zoals bodemkenmerken, landgebruik en bevolkingsdichtheid in de omgeving en socio-demografische kenmerken van die bevolking worden meegenomen om geobserveerde waardes te transfereren van de studie-site. De resultaten van drie gevalstudies tonen inderdaad aan dat er grote verschillen kunnen zijn voor de waarde van ecosysteemdiensten en dat het werken met waarderingsfuncties een toegevoegde waarde heeft.

Een laatste hoofdstuk beschrijft hoe data en resultaten van hydrologische modellen en economische afwegingskaders kunnen geïntegreerd worden en beschikbaar gemaakt worden voor eindgebruikers in een online beslissingsondersteunend systeem. De doelstelling van dit systeem is om een synthese van relevante informatie voor verschillende schalen en wateraspecten te geven die gebaseerd is op monitoringgegevens, veldstudies en modelresultaten. Meer concreet geeft dit systeem informatie over de status, druk (bronnen), kosten en effecten van maatregelen. Ook kunnen voor scenarios berekeningen uitgevoerd worden van de kosten, effecten en kosten-effectiviteit. Dit systeem is uitgewerkt voor oppervlaktewaterkwaliteit, hydromorfologie en sedimenten. Dit instrument wordt op dit moment toegepast om een screening uit te voeren van individuele waterlichamen, de selectie van pilootgebieden te onderbouwen en de disproportionaliteitsanalyse uit te voeren voor de tweede generatie stroomgebiedbeheerplannen.

In het algemeen bevestigt dit onderzoek dat er instrumenten kunnen ontwikkeld worden om kosten-effectiviteit van maatregelenprogramma's te beoordelen. De resultaten van deze instrumenten bevestigen de toegevoegde waarde en potentiële kostenbesparingen die Vlaanderen kan realiseren door een meer locatie-specifiek beleid te voeren.

Tot slot worden ook nog een aantal aandachtspunten aangehaald voor verder onderzoek. Het dynamisch voorspellen van de impact van maatregelen op langere termijn heeft mogelijk een belangrijke impact op de kosten-effectiviteit van maatregelen gericht op het reduceren van diffuse emissies. Het zal ook belangrijk zijn om methodes verder te ontwikkelen om kosten-effectiviteit te beoordelen over water aspecten heen. Voorts zal ook beter moeten worden uitgewerkt hoe omgegaan wordt met onzekerheden, zowel voor de berekeningen als voor de communicatie ervan naar eindgebruikers.

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BAT	Best Available Technologies
BATNEEC	Best Available Technologies Not Entailing Excessive Costs
CBA	Cost-benefit Analysis
CEA	Cost-effectiveness Analysis
CIW	Coordinatiecommissie Integraal Waterbeheer
CRT	Controlled reduced tidal area
CSO	Combined sewer overflow
CVM	Contingent Valuation Method
DSS	Decision Support System
EC	European Commission
ECM	Environmental Costing Model
ES	Ecosystem Services
EU	European Union
FCA	Flood control area
IE	Inhabitant Equivalent
LNE	Flemish Environment Administration
MKM	MilieuKostenModel (Environmental Costing Model)
OECD	Organisation for Economic Co-operation and Development
RBMP	River Basin Management Plan
VMM	Vlaamse MilieuMaatschappij (Flemish Environment Agency)
WFD	Water Framework Directive
WTP	Willingness To Pay
WWTP	Wastewater Treatment Plant
UWWD	Urban Wastewater Directive

CHAPTER 1. Scope, objectives and outline

1.1. Introduction and problem statement

Water is an important resource for humanity and the rest of the living world. Our rivers, lakes, coastal and marine waters as well as our ground waters are valuable resources to protect (European Commission, 2010). The OECD Environmental Outlook to 2050 identifies different factors as increasingly rapid urbanisation, population growth and economic dynamics that are expected to increase pressure on water resources in the upcoming decades. Water demand worldwide is projected to grow by some 55% due to growing demand from manufacturing (+400%), thermal electricity generation (+140%) and domestic use (+130%). Water pollution from point sources (urban wastewater) and “diffuse sources” (mainly from agriculture) is projected to worsen in most regions, resulting in intensified eutrophication and damaged aquatic biodiversity. The combined effects of these pressures can lead to water shortages and pollution that hinder the growth of economic activities (OECD, 2012a). The water system itself is also particularly vulnerable to climate change. During the coming century, climate change will lead to reduced access to safe drinking water, as glaciers melt away and drought becomes more frequent in areas like the Mediterranean. This will diminish the supply of water for irrigation and food production. More frequent flooding will increase damage to homes, infrastructure and energy supply (OECD, 2012b).

An improved governance of our waters is required to better protect this valuable resource in the future. However, water cuts across all social, economic and environmental activities. Its governance requires cooperation and coordination across diverse stakeholders and sectoral ‘jurisdictions’. Integrated water management can be defined as setting up a holistic management system that brings different water management issues into one framework. This holistic approach is aimed to protect the whole body of water, its source, tributaries, delta and river mouth. Ideally, it tackles pressures and risks addressing the whole life cycle of water policy across the different policy spheres through a co-ordinated strategy, involving all the interested parties in decision-making (European Commission, 2010). Such an overall strategic approach can deliver more effective, efficient and sustainable policies (OECD, 2012b).

Economic appraisal techniques can be key elements leading to more integrated water management. An important stepping stone leading to an increased application of economic appraisal techniques for integrated water management is the International Conference on Water and the Environment in Dublin in 1992 and the proclamation that water should be treated as an economic good. “Water has an economic value in all its competing uses and should be recognized as an economic good. Within this principle, it is vital to recognize first the basic right of all human beings to have access to clean water and sanitation at an affordable price. Past failure to recognize the economic value of water has led to wasteful and environmentally damaging uses of the resource. Managing water as an economic good is an important way of achieving efficient and equitable use, and of encouraging conservation and protection of water resources.” (ICWE, 1992) Inspired by the proclamation of water as economic good, the European Water Framework Directive (WFD) (European Commission, 2000) explicitly integrates economic analysis in both water management and water policy decision-making in Europe. The main objective of the Directive is to meet good status of all waters. To ensure that this goal will be met, member states have to assess the current state of all waters, existing pressures, identify significant water management issues and publish river basin management plans to tackle these issues. To achieve its environmental objectives and promote integrated river basin management, the Directive calls for the application of economic principles (for example, the polluter-pays principle), economic approaches and tools (e.g. cost-effectiveness analysis) and instruments (e.g. water pricing) in order to achieve its environmental objectives and promote river basin management (WATECO, 2003). The Water Framework Directive is the first EU environmental directive that explicitly integrates economic principles, tools and instruments into legislation. Whereas the generic principles on how to perform these appraisal techniques are clear and well documented, the application is less straightforward. It requires the use of both economic appraisal techniques and hydrological modelling tools at different spatial and time scales.

This thesis is focused on the development and application of tools for the region of Flanders, Belgium. This region is facing a whole range of significant water management issues. Flanders is a highly urbanized and densely populated region with a surface of 13,521 km² and a population of more than 6 million inhabitants (population density 462 inhabitants/km²). The region is part of two international river basin districts, the Scheldt and the Meuse. The water system mainly consists of lowland rivers with wide valleys and slow flow velocities. Highly industrialized areas are the ports of Antwerp and Ghent. Agriculture is mainly intensive and cultivated land occupies 45% of the area. The assessment of the current status in 2009 (Coordinatiecommissie Integraal Waterbeleid, 2009) indicated that a very small amount of surface and groundwaterbodies are currently in a good status. Key issues preventing to achieve this good status are surface water quality and groundwater quality (nutrients, chemicals), flooding (sea level rise), sediments (dredging and processing polluted sediments), hydromorphology and droughts (surface water or groundwater quantity issues in specific areas). Due to a variety of reasons it is very difficult to significantly improve the status of all waterbodies. From a technical point of view it is especially difficult to restore rivers in a highly urbanized area and to tackle diffuse pollution and historic pollution stocks present in groundwater and sediments in a short term. From an economic point of view, reaching good water status is very expensive. Large shares (60%) of the environmental expenditures by the government are already going to water policy (VMM-MIRA, 2012). Additionally, the financial burden for the different sectors (households, industry and agriculture) related to water increased significantly in the last decade. The drinking water price for households has for example tripled between 2000 and 2011 (VMM, 2012). Setting up programmes of measures which are able to solve different water management issues at the lowest cost achievable is therefore one of the key objectives of water management in this region during the upcoming decades.

1.2. Conceptual framework

1.2.1. Overview

The conceptual framework developed and applied in this PhD research is represented in Figure 1 and describes how modelling tools can be used to determine a cost-effective program of measures for water management. A combination of economic appraisal techniques and hydrological models is used to support decision making in integrated water management. Results of economic appraisal techniques are applied to compose scenarios and to estimate how pressures are reduced with these scenarios. The impact of these pressure reductions on the water status is predicted by hydrological models. The impact of scenarios on status can again be used as input for economic appraisal techniques to indicate if additional measures are required to reach the environmental targets.

Both economic appraisal techniques and hydrological models require extensive databases. Databases are required on pressures, state and measures. Data on pressures indicate the contribution of the different sources to an environmental issue. The pressure database is used both by the hydrological models and the economic appraisal techniques. Information on measures consists of costs and effects. Effects are expressed as the effectiveness of reducing pressures from a specific source. Costs are investment and operational costs for installing a certain measure. The state database contains information on the existing state of the water system. State information is used to set up and calibrate hydrological models and important to determine how far the existing status is still removed from the target level.

More details of this conceptual framework are given in the next paragraphs.

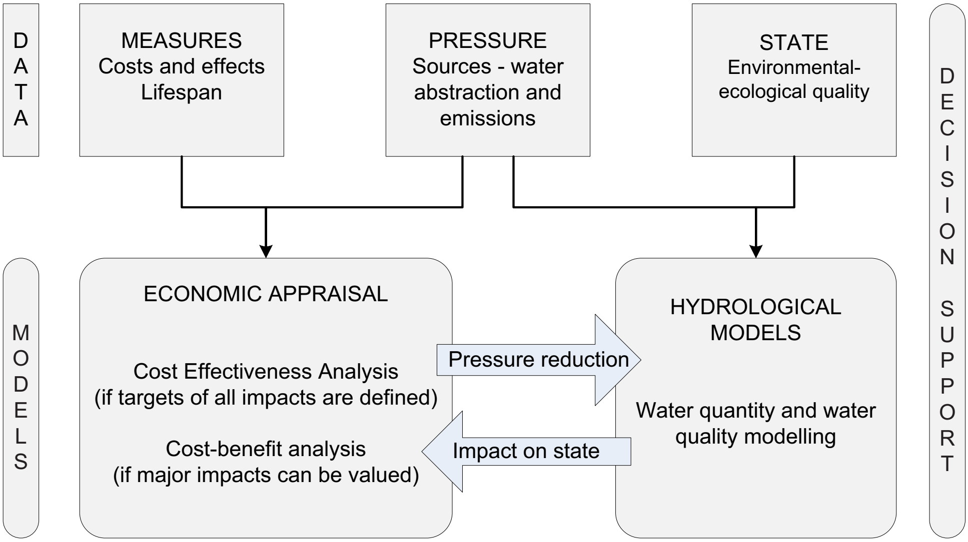

Figure 1: Conceptual framework on integration of economic appraisal techniques and hydrological models developed in this thesis

1.2.2. Economic appraisal techniques

1.2.2.1. Cost-effectiveness analysis

Cost-effectiveness analysis (CEA) is a technique for identifying the least-cost option for meeting a specific physical objective/outcome (Balana et al., 2011). In the context of water management, the purpose of a cost-effectiveness analysis is to find out how predetermined targets, for example pollutant loads in a waterbody, river basin or estuary, can be achieved at minimal costs (Lise and van der Veeren, 2002). It can be used as an appraisal technique for assessing and ranking the relative performance of different measures or combination of measures on the basis of their costs and their effectiveness. Targets can be defined in two ways: if the objective is the achievement of a given level of environmental effectiveness (e.g. specific standards of water quality), the most cost-effective measure is the one with the lowest cost. If the objective is such that the cost of the proposed measure should not exceed a given budget, the most cost-effective measure is the one with the highest environmental effectiveness achieved within this budget (Zanou, 2003). Cost-effectiveness indicators typically measure the value for money and are best applicable when environmental effects are homogenous across alternative solutions or projects and can be comprehended in single effect indicators (OECD, 2007). Individual or collective wastewater treatment for both industry and households are examples of measures with homogenous effects, influencing similar water aspects and parameters. They can be easily compared in a cost-effectiveness analysis. Wetland restoration and wastewater treatment have less homogenous effects as wetlands also have a potential impact on flood risk prevention, biodiversity and recreation, which is less the case for wastewater treatment. Comparing these last two examples in a cost-effectiveness analysis focusing on reducing nutrient losses would lead to an unbalanced comparison.

Most of the existing studies dealing with cost-effective water management focus on surface water quality and more specifically nutrient losses (Fröschl et al., 2008; Gren et al., 1997; Lise and van der Veeren, 2002; Panagopoulos et al., 2011; Schleich and White, 1997; Wang and Cresser, 2007; van der Veeren and Tol, 2001; Mewes, 2012). Typically, economic optimization models are used to achieve multiple emission reduction targets for multiple locations at the lowest achievable cost. Whereas Schleich and White, 1997 discussed the optimization of individual sources in a smaller catchment (waterbody), the other authors investigated the impact of implementing measures on a much larger scale in international river catchments. To date, the application of this type of models in decision making processes for integrated water management still remains limited as they experience difficulties in grasping the complexity of integrated water management and making balanced comparisons between different types of measures at different scales. Measures which are considered in river basin management plans can be very diverse in different aspects. Measures are not limited to technical measures as wastewater treatment, specifically focused on reducing specific types of pollution. They also include land based measures as bufferstrips and restoration of hydro-morphological conditions which are mostly not very cost-effective in tackling a specific type of pollution but have wider positive impacts on other water aspects as flood risks and sediments. The spatial scale of measures can be diverse ranging from more strict nutrient legislation defined at the national level to implementing wastewater treatment at the individual household or company level. Also the relevant time scale can be very diverse. Measures reducing annual emissions cost-effectively, which is typically determined in a CEA, might be less suited to improve water quality in specific time periods. Water quality targets are often defined as seasonal averages or peak concentration levels and not necessarily in terms of annual average concentration levels.

1.2.2.2. Cost-benefit analysis

Cost-benefit analysis (CBA) is a technique that is used to estimate and sum up (in present value terms) the future flows of benefits and costs of society's resource allocation decisions or policy alternatives to establish the worthiness of undertaking the stipulated activity or alternative, and inform the decision maker about economic efficiency. CBA addresses the question of whether the objective (or action) is economically worthwhile and finding the efficient level of emissions: do the benefits exceed the costs and are net benefits maximized (Balana et al., 2011)? A CBA goes further than a CEA, in a sense that we do not start from a pre-defined target level of environmental pollution. Instead, the objective is to estimate the efficient level of pollution or emissions. In a CEA the objective is fixed exogenously whereas in a CBA it is endogenously determined in the optimization. The idea of economic efficiency is that there should be a balance between the value of what is produced and the value of what is used to produce it (Field and Field, 2008).

Cost-benefit analysis is most often applied in water management to compare alternative scenarios for flood risk prevention (Bouma et al., 2005; Brouwer et al., 2004; Dutta et al., 2003; Kourgialas and Karatzas, 2012; Penning-Rowsell and Chatterton, 1977; Turner et al., 2007). Effects in this case are not expressed as the relative contribution to reach a predefined target but as material and non-material damages prevented due to flooding. These damages can relatively easy be expressed as monetary values and compared with investment and operational costs. Additionally, other impacts as the impact of natural water retention measures on water quality or sediments can also be included in the cost-benefit analysis. A disadvantage of most of these applications is that they are restricted to a small set of predefined scenarios (often in function of the intended protection level) and are hard to apply for the selection of an optimized and spatially diverse programme of measures, out of large amounts of potential measures. Another difficulty of a cost-benefit analysis compared to a cost-effectiveness analysis is that all environmental effects (reduced damages) or benefits need to be valued in monetary terms to allow for a comparison with the costs, which adds an additional level of complexity. For the monetary valuation of benefits large challenges still remain. Most commonly, specific value estimates for project sites are not available and monetary values estimated at another study site are transposed to the project site. This technique is also referred to as Benefits Transfer. The most important reason for using previous research results in new policy contexts is that it avoids expensive and time consuming original research to quickly inform decision making (Brouwer, 2000). Benefits Transfer has been the subject of considerable controversy in the economics literature (Brouwer, 2000; Christie et al., 2004) as it has often been used inappropriately. A limitation of Benefits Transfer studies is that most existing valuation studies only produce localised value estimates, i.e. site-specific values, and pay limited attention to important spatial characteristics in the valuation of land use change, open space and fragmentation (Bateman et al. 2006), which makes Benefits Transfer less reliable.

1.2.3. Interaction with hydrological models

Hydrological models are simplified, conceptual representations of a part of the hydrologic cycle. They are primarily used for hydrologic prediction and for understanding hydrologic processes (Merritt et al., 2003). Models differ in terms of complexity, processes considered, and the data required for model calibration and model use. In general there is no 'best' model for all applications. The most appropriate model will depend on the intended use and the characteristics of the catchment being considered. Three types of models are typically distinguished (Merritt et al., 2003). Empirical models are generally the simplest of all three model types. These models are based on data and are using mathematical and statistical concepts to link a certain input (for instance rainfall) to the model output (for instance runoff). Such models are generally based on the assumption of stationarity, i.e., it is assumed that underlying conditions remain unchanged for the duration of the study period. This assumption limits the potential for such models to be applied for predicting the effects of catchment change. PC Raster – Polflow (De Wit, 2001) is an example of a hydrological model making use of empirical process descriptions. Conceptual models are typically based on the representation of a catchment as a series of internal storages. They usually incorporate the underlying transfer mechanisms of water and pollutants transport in their structure, representing flow paths in the catchment as a series of storages, each requiring some characterisation of its dynamic behaviour. Examples are PDM (Moore, 2007), NAM (Danish Hydraulic Institute, 1982), SWAT (Neitsch et al., 2005) and VHM (Taye et al., 2011). Process-Based Models try to represent the physical processes observed in the real world. Standard equations used in such models are the equations of conservation of mass and momentum for flow and the equation of conservation of mass. Typically, such models contain representations of surface runoff, subsurface flow, evapotranspiration, and channel flow, but they can be far more complicated. Examples are MIKE-SHE (Danish Hydraulic Institute, 1993), MODHMS (HydroGeoLogic Incorporated, 2006), and HYDROGEOSPHERE (Therrien et al., 2009). The distinction between the different types of models is not sharp and therefore can be somewhat subjective. They are likely to contain a mix of modules from each of these categories.

Integrated economic and hydrological models are also referred to as hydro-economic models. Hydro-economic models aim to capture the complexity of interactions between water and the economy (Brouwer and Hofkes, 2008). In the scientific literature, typically a distinction is made between two different approaches for hydro-economic modelling: the modular approach and the holistic approach. In the modular approach a connection is built between the hydrological and economic model, and output data from one module usually provides the necessary input for the other. Feedback loops or iterations may be needed, requiring appropriate model interfaces among model components. In the holistic approach there is one single control unit with both the hydrologic and economic component tightly interwoven in a consistent model (Heinz et al., 2007). Modular approaches have the advantage that it is easier to develop, calibrate and solve individual models. The disadvantage is however that each model must be updated and run separately and it can be difficult to connect models with different scales. Holistic models are easier to represent causal relationships and interdependencies and perform scenario analyses. A disadvantage is that all models must be solved at once and that the increased complexity of the holistic model requires simpler model components (Harou et al., 2009). Finding the right balance between model complexity, model integration and sufficient model accuracy to evenly compare alternative courses of action is an important challenge in hydro-economic modelling (Harou et al., 2009).

1.2.4. Decision support systems

The concept of a DSS can be extremely broad and its definitions vary. In this thesis a DSS is considered as ‘‘an interactive, flexible, and adaptable computer-based information system, especially developed for supporting the solution of a non-structured management problem for improved decision-making” (Gourbesville, 2008). Examples of DSSs for water management include AQUATOOL for water scarcity issues (Pulido Velazquez et al., 2008), FLUMAGIS for flood management, water and ecological quality issues (Volk et al., 2008) and MULINO (Giupponi, 2007), which focuses less on modelling the consequences of measure but more on participatory processes, problem identification and presenting model outputs. Hirschfeld et al. (2005) present a conceptual DSS which is able to perform a cost-benefit analysis within an interdisciplinary spatial decision support system for an integrated management of the Werra River Basin including surface water quality and river morphology. BASINFORM, presented in Bernd et al. (2012) includes quantification methods to identify the need for action, modelling tools for quantifying the impacts of management measures and a method for selecting cost-effective combinations of measures to improve surface water quality.

Despite the significant effort and large amounts of money spent on developing DSSs, there are many that are never or hardly used. Possible reasons are that the users find the system too detailed, time consuming and costly to use (Uran and Janssen, 2003). A lack of appropriate and methodological stakeholder interaction and no clear definition of ‘what end-users really need and want’ have also been documented as general shortcomings (De Kok et al., 2009; Volk et al., 2010). Other reasons are related to the general complexity of the systems, the uncertainty of the model output and on the limited appropriateness for solving the decision question. A major problem of the current DSSs used for the WFD is that they have been developed for quite specific issues and do not cover the transdisciplinary broadness of the WFD in combination with different scales and water aspects (Gourbesville, 2008). Also the issue of cost-effectiveness has so far been often neglected in decision support concerning the WFD (Ward, 2007). Consequently, decisions made in practical river basin management are often not well documented. In particular, it remains unclear how cost-effectiveness has been taken into account by the authorities when selecting measures (Bernd et al., 2012). Important challenges identified by McIntosh et al. (2011) include the need for a participatory process that involves end users and stakeholders throughout the design and development process, adoption challenges concerned with individual and organisational capacities to use DSS, maintaining the focus on developing DSS which are relatively easy and inexpensive to use and update, ensuring DSS longevity and financial sustainability and evaluating the success of DSS in terms of end user involvement and the extent to which DSS achieve intended outcomes.

1.3. Research objectives and hypotheses

The overall objective of this research is to **develop and apply modelling tools that can assist policy makers to compose cost-effective programmes of measures for water management.** Important for these tools is that they are on the one hand applicable for decision making on a national or regional level, but on the other hand also sufficiently detailed to correspond with the choices policy makers are faced with on the local project level.

Specific research objectives are listed in the next paragraphs.

1.3.1. Objective 1: Develop a multi-objective economic optimization model to account for the differences in spatial scales within a cost-effectiveness analysis

Decision making for integrated water management occurs on different scales ranging from the individual household or company level (micro scale) to a more regional/national level (macro scale). The cost-effectiveness of measures can depend heavily on the spatial scale. The most cost-effective measures on a regional scale might not be the most cost-effective on a local scale. To develop a model that is able to deal simultaneously with measures that are implemented on different scales and test how scaling in decision making impacts the cost-effectiveness is the first objective.

1.3.2. Objective 2: Integrate an economic optimization model and a hydrological model to assess the cost-effectiveness of measures to reach specific concentration targets

Whereas most cost-effectiveness analyses are restricted to determine the least cost combination of reaching specific emission reduction targets, a good water status is defined by a range of instream concentration targets. It is possible that the most cost-effective measures to reach annual emission reduction targets are not necessarily the most cost-effective measures to reach specific instream concentration targets. How concentration targets are specifically defined plays an important role. Peak concentration targets are likely to be more influenced by different measures compared to annual or summer average concentration targets. Consequently, cost-effectiveness presumably depends on how concentration targets are defined. To develop a methodology that enables to test this assumption is the second objective.

1.3.3. Objective 3: Develop a risk based approach to select cost-effective programmes of measures instead of optimizing towards predefined environmental targets

The previous two objectives are focused on reaching predefined environmental targets in a cost-effectiveness analysis. The third objective is to develop a methodology that allows to spatially optimize a program of measures in a risk based approach. This means that environmental targets are not predefined but are endogenously determined in the optimization. This risk based approach allows for the implementation of more measures in areas where more environmental benefits can be achieved. To develop a methodology to select a risk based and spatially diverse programme of measures, out of large amounts of potential measures is the third objective. This allows to test whether risk based approaches are able to realize significantly more benefits compared to the application of fixed targets.

1.3.4. Objective 4: Develop an ecosystem service valuation framework to take into account the impact of measures on different water aspects simultaneously for the selection of cost-effective programmes of measures

The economic valuation of ecosystem services presents a promising tool to value the impact of multi-purpose measures on different water aspects simultaneously. By quantifying and valuing the different services these measures deliver, a better view can be obtained on the wider impact of measures instead of the impact on a single environmental issue. Though methodologies for classification, quantification and valuation are improving, applications of the ecosystem services concept stay mainly restricted to illustrating the importance of preserving or restoring ecosystems in regional to global ecosystem service mapping or ecosystem services accounting exercises. Applying this framework in a more localized land or water management context, taking into account spatially sensitive and project site-specific inputs in ecosystem service quantification and valuation functions is the fourth objective. It can be expected that value estimates depending only on the type of land use, as is often applied in ecosystem service valuation, do not sufficiently grasp project specific circumstances to compare scenarios in water management.

1.3.5. Objective 5: Develop web based decision support systems that are able to help policy makers in selecting cost-effective programmes of measures both at local and regional scale

The last objective is to develop web based decision support systems to communicate data and results of hydrological models and economic appraisal techniques to end users. An important challenge is to develop systems that are of added value in existing decision making processes on integrated water management. End user involvement in different steps in the development process (user requirements analysis, system development, testing and application) is a very crucial step which is focused on during the development. Systems should be relatively easy to use and update, without requiring end users to gather extensive datasets and perform detailed modelling exercises themselves. Another specific challenge is to cover the transdisciplinary broadness of the WFD in combination with different scales and water aspects.

1.4. Research design

1.4.1. Research context: policy applications for different end users in Flanders

The research for this thesis was performed at the Flemish Institute of Technological Research (VITO) and derived from several projects related to integrated water management, cost-effectiveness analysis, cost-benefit analysis and valuation of ecosystem services.

A cost-benefit analysis was performed to update the flood risk management of the Scheldt tidal estuary (so called Sigmaplan). The objective was to create win-win situations for accessibility, nature development and flood protection in both the Dutch and Flemish part of the Scheldt tidal estuary (MKBA Sigmaplan en Langetermijndoelstellingen Schelde). This project was financed by the commissioned by the Waterways and Marine Affairs Administration of the Environment and Infrastructure Department of the Ministry of the Flemish Community (Belgium) and the Dutch-Flanders project organisation ProSes established by the Dutch and Flemish governments to develop the long term vision for the Scheldt estuary.

A cost-effectiveness analysis was executed to determine how a good water status can be reached across the Flemish Region at the lowest cost achievable. During the period 2005-2009 this study looked mainly at surface water quality and a limited amount of parameters (COD, N, P) in preparation of the first river basin management plan. In preparation of the second river basin management plan, this study is since 2009 looking at multiple water aspects and the development of a web based tool where all principles of a CEA are integrated and where end users can perform simulations at different spatial scales. This study is the main focus of this thesis. (Milieukostenmodel Water) This project is financed by the Flemish Environment Administration (LNE) and the Flemish Environment Agency (VMM).

Several valuation studies on ecosystem services were performed between 2009 and 2012 financed by the Flemish Environment Administration (LNE), the Agency for Nature and Forest (ANB), the Flemish Agency for Innovation by Science and Technology (IWT) and the Belgian Science Fund (Belspo).

The principles of both the cost-effectiveness analysis and the disproportionate cost analysis were also performed in preparation of river basin management plans for the Walloon Region, commissioned by the Walloon Environment Administration (DGARNE) and the public water association (SPGE), and for the Brussels Region, commissioned by the Brussels Environment Administration (BIM).

This close link to end users ensures the policy relevance of this research. It created the ability to build on extensive datasets on environmental quality, pressures, measures and hydrological models. However, this also required methodologies to be suitable for a large area, the entire region or at least the entire Scheldt river basin which is the large majority of the area. Also, the use of hydrological models previously developed for environment agencies is often a boundary condition.

1.4.2. Multi-disciplinary team effort

The outline of the various steps in the conceptual framework shows that this research is based on a multidisciplinary exercise, requiring the input and collaboration of different scientific disciplines, such as environmental scientists (physical effects of measures), economists (costs of measures), engineers (details about technical measures) and ICT-specialists (web applications).

Across all applications a stepwise approach of activities was applied by the author of this research:

1) Interact with end users to understand and define the problem and potential solutions
2) Choosing the economic appraisal technique (CEA or CBA)
3) Understand required and available hydrological models
4) Collect and construct required databases – process hydrological model output
5) Construct the economic appraisal framework
6) Build/simulate scenarios and interact with hydrological modelers
7) Compare scenarios and report

Most chapters are also published in scientific journals or books. Due to the multi-disciplinary nature of this research and the large datasets it required, these publications are shared with a large series of co-authors.

1.5. Thesis outline

The next chapters represent different cases where the listed objectives are handled in different ways and for different water aspects and areas. Figure 2 describes how these chapters deal with specific aspects from the conceptual framework.

Chapter 2 gives an overview of the state of the art in applying economic appraisal techniques and hydrological models to achieve more cost-effective water management.

Chapter 3 focuses on cost-effectiveness analysis and the development of an economic optimization model. The research objective is to develop a model that contrary to most other existing economic models works at the same scale as hydrological models to allow for easy exchange of scenarios between hydrological and economic models, can be used for local and regional scales and includes upstream-downstream interactions. This model set-up is tested for surface water quality targets in the entire Flemish region and smaller waterbodies. The model was specifically built to demonstrate the importance of scale and potential cost savings due to a spatial diversification of the programme of measures. To deal with the differences in scale, optimization algorithms typically used in a cost-effectiveness analysis are extended with the possibility to include individual measures aimed at reducing individual sources and collective measures that can reduce multiple sources simultaneously. The impact of the reduced pressures on the surface water quality status due to different programs of measures is simulated with the water quality model Pégase.

Chapter 4 focuses on the integration of an economic optimization model and a hydrological model to assess the cost-effectiveness of measures. The objective is to determine cost-effective sets of measures to reach in-stream concentration targets instead of emission reduction targets as performed in the previous chapter. To realize this objective a modular modelling approach is applied that determines the most cost-effective set of reduction measures to reach an in-stream concentration target. The framework is based on the coupling of two models: the hydrological water quality model SWAT (reference) and an economic optimization model (Environmental Costing Model, ECM). SWAT is used to determine the relationship between the modelled in-stream concentration at the river basin outlet and the associated emission reduction. This set-up also allows to deal with smaller time scales and daily variations in hydrological conditions. This allows to optimize towards other types of objectives as summer average or 90 percentile, instead of yearly averages. Daily variations in hydrological flows and different impacts of hydrological conditions on point and diffuse source emissions might influence the cost-effectiveness of measures.

Chapter 5 describes a cost-benefit analysis to determine spatially diverse flood risk management plans for the tidal Scheldt estuary. The objective is to develop a methodology that is able to integrate results from hydrological models in a cost-benefit analysis, simultaneously deal with upstream-downstream effects, take into account the impact of sea level rise and also is able to include a range of environmental impacts which is needed to evenly compare techniques as a storm surge barrier with floodplain restoration. Contrary to previous chapters, the starting point of this methodology is not a predefined target, but a risk based approach is applied. This means that environmental targets are not predefined but are endogenously determined in the optimization. More protective measures are installed in areas where more flood risk can be expected. Scenario development is performed by iterative feedback loops between flood risk simulation models and economic analysis. Besides effects on flood protection, impacts on water quality, sediment management, recreation and climate change regulation are also valued as part of the cost-benefit analysis. Spatial interdependencies are dealt with by stepwise scenario development where the study area is subdivided in 5 subzones and the optimal scenario in subzone 1 is the starting point for subzone 2 and etc.

Chapter 6 focuses on the valuation of benefits and more specifically of ecosystem services, necessary to consider multiple environmental aspects simultaneously in a cost-benefit analysis. The objective is to develop a valuation toolbox that is able to deal with a range of environmental effects that can be created by land and water management. Whereas most applications on ecosystem service valuation are aimed towards large scale ecosystem service accounting exercises (value of ecosystem services in an entire country or region), a methodology was developed to assess and value the impact of individual measures as floodplain restoration or bufferstrips on a range of ecosystem services, which is difficult to do with tools previously developed. This can support the development of cost-effective policies that establish win-win situations across different environmental domains. The chapter discusses the user requirements for ecosystem service valuation, the main tool characteristics, potential policy applications and future improvements. The most relevant characteristics of the project site (e.g. soil, surrounding land use, population living nearby, sociodemographic factors of the beneficiaries) are taken into account to transfer values from other study sites. The results of three case studies demonstrate the importance of taking site specific characteristics (other than size and land use type) into account for the valuation of ecosystem services.

Chapter 7 describes how data and results from hydrological models and economic appraisal techniques can be integrated and made available for end users in decision support. The objective is to provide synthesized information from monitoring, field studies and models on different scales and water aspects. It should allow end users to consult information from monitoring campaigns and modelling tools as developed in the previous chapters. To realize this objective, an innovative web-based decision support tool was developed to provide the necessary data to assess costs, effects, benefits and affordability of packages of measures. Information about status, pressures, costs and effects of measures can be retrieved and simulation results can be generated on different scales, from individual waterbodies to regional level. End users can build up draft packages of measures (scenarios), assess their costs and effects and share these scenarios with other users (e.g. users building scenarios for other aspects or for other waterbodies).

Chapter 8 provides the general discussion and conclusions.

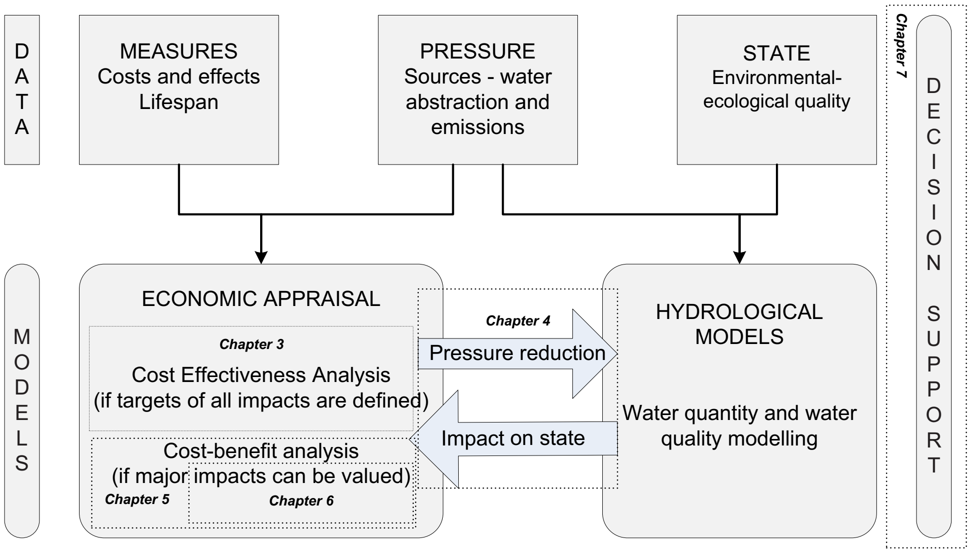

Figure 2: Major methodological focus of chapters vs. conceptual framework

1.6. Beyond the state of the art

An overview on the existing state of the art in applying modelling tools for cost-effective water management is described in the next chapter. Important challenges include the integration of economic and hydrological models, spatial and temporal scaling and the consideration of multiple water aspects simultaneously.

This thesis explores different methods for integrating economic and hydrological simulation models to determine cost effective programmes of measures. This is combined with results of original economic valuation studies. Additionally, innovative web based applications are developed to present models and results to end users. This enables end users to explore the cost effectiveness of scenarios and helps to improve decision making processes in Flanders, Belgium. Though the models in this thesis are applied on large scale study areas (region, river basin), a lot of attention is given to take into account area specific characteristics in both the calculation of costs and effects of measures and the valuation of benefits. This allows providing a more differentiated picture on the cost-effectiveness of measures which can improve decision making on different scales.

Additionally, specific innovative aspects are tackled in every individual chapter. In chapter 3, an economic optimization model considering measures on different scales simultaneously ranging from the individual household or company level (micro scale) to a more regional/national level (macro scale) and which allows to shift emissions to other locations (different water bodies) due to the connection to collective wastewater treatment is applied and is an additional step compared to existing economic optimization models. Chapter 4 demonstrates how marginal abatement cost curves can be derived for instream concentration targets instead of emission reduction targets and how this influences the cost-effectiveness of measures. Chapter 5 demonstrates how a cost-benefit analysis can be applied to select a risk based and spatially diverse programme of measures, out of large amounts of potential measures. This allows to test whether risk based approaches are able to realize significantly more benefits compared to the application of fixed targets, unlike most other existing cost-benefit analyses which focus on a limited set of predefined scenarios. Additionally, ecosystem service valuation is taken into account to include the impact of flood risk prevention on other environmental issues as climate change and water quality. The tool presented in chapter 6 goes beyond other exploratory tools to determine the value of ecosystem services. Whereas other existing exploratory tools are limited to rapid qualitative assessments, this tool includes quantitative and monetary valuation of ecosystem services due to land use change. Additionally, important spatially sensitive and project site-specific characteristics influencing the value are taken into account. Chapter 7 goes beyond existing decision support systems for integrated water management in end user involvement and covering the transdisciplinary broadness of the WFD (water status, identifying pressures and potential measures, assessing cost-effectiveness and disproportionate costs) in combination with different scales (from water body to region) and water aspects (surface water quality, sediments, hydromorphology).

CHAPTER 2. State of the art in applying CEA and CBA for integrated water management

In this paragraph an overview is given of the existing state of the art in applying economic appraisal techniques for water management. The overview is structured according to water aspects.

2.1. Surface water quality

Most scientific literature on applying economic appraisal techniques in water management is focused on surface water quality and more specifically nutrient losses. Besides studying the cost-effectiveness (cost-effect ratio) of a limited amount of scenarios (Lescot et al., 2013; Mewes, 2012; Petersen et al., 2009; van Grieken et al., 2011) for one specific pollutant, other studies (Elofsson, 2003; Fröschl et al., 2008; Gren et al., 1997; Hanley et al., 1998; Lise and van der Veeren, 2002; Panagopoulos et al., 2012; Schleich and White, 1997; van der Veeren and Tol, 2001) made use of economic optimization procedures to determine combinations of measures which are able to reach multiple pre-defined emission reduction targets. Transport coefficients, derived from hydrological models, are applied to take into account upstream-downstream interactions. Panagopoulos et al. (2012) went a step further in simulating several pareto set optimal solutions with a hydrological model to assess the impact on the concentration levels.

The scale of the assessments is mostly focused on investigating the impact of implementing measures on a larger scale in entire river catchments. Costs and effects are grouped for the entire study area or larger sub-catchments or countries to account for spatial differences in cost-effectiveness. Using this level of aggregation only allows for large-scale analyses and corresponding large-scale conclusions, namely which source categories in which sub-catchments are most cost-effective to address (Schleich and White, 1997). Panagopoulos et al. (2011) and van Grieken et al. (2011) study the cost-effect ratios of agricultural measures on a more detailed spatial scale of hydrological response units.

Though a large range of studies exist to study the cost-effectiveness of measures to improve surface water quality, this is mostly limited to nutrients. Besides nutrients, Brouwer and De Blois (2008) studied the most effective measures to reduce colony forming units, which is important to reduce bacteriological contamination at bathing sites in the Netherlands. Hanley et al. (1992) tested the effectiveness of tradable Pollution Permits (TPPs) in the Forth Estuary, Scotland to control inputs of biological oxygen demand more cheaply than the current regulatory system. Recently, Lescot et al. (2013) study the most cost-effective strategies in reducing pesticide-related water pollution. For parameters such as pesticides, metals, dangerous substances, very few or no known studies exist. This has two possible reasons. On the one hand the amount of information available on these water quality parameters is generally more limited compared to the basic water quality variables. On the other hand, the amount of potential measures is limited. Whereas an optimization exercise makes sense in case of many polluters and many possible remediation strategies, this is much less the case for specific pollutants coming from a limited amount of sources that need to be dealt with individually. Modelling studies for these parameters are mostly limited to assess the impact of specific scenarios on water quality (Holvoet et al., 2008; Gevaert et al., 2012)

Cost-benefit analyses are less applied to surface water quality as they mostly focus on single or a limited set of pollutants and this does not require the valuation of benefits. A large series of studies are available however on the valuation of benefits of surface water quality. Mostly, contingent valuation studies are performed surveying willingness to pay to reach a certain water quality (Bateman et al., 2011; Del Saz-Salazar et al., 2009; Martin-Ortega and Berbel, 2010;,Perni et al., 2012). Benefits are mostly defined very generic (WTP to reach good water quality or good ecological status) and very difficult to link with marginal improvements of quality which is the level of detail typically used in a cost-effectiveness analysis. Methodologically, this bridge is difficult to make as quality improvements need to be sufficient for people to be able to express a certain willingness to pay for it, whereas contributions of individual measures to water quality can often be very marginal. The water quality ladder which distinguishes boatable, fishable, swimmable and drinkable levels of quality, is typically applied in contingent valuation studies (for example in Del Saz-Salazar et al., 2009) to increase the level of detail and provides more intermediate points to estimate benefits. However, this is still not sufficiently accurate to be compared with the costs of one additional measure. Barton et al. (2008) were able to compare costs and benefits by making use of a bayesian network approach to conduct decision analysis of nutrient abatement measures. The tool is used for structuring and combining the probabilistic information available in existing cost-effectiveness studies, eutrophication models and data, non-market valuation studies and expert opinion. Differences in details between cost and effect estimations and benefits are dealt with by probabilities. Benefits were estimated as a willingness to pay for bathing. Depending on the selection of measures, the probability that people actually could bathe varies and is multiplied with the benefit.

In most cases, a CEA is used to define the programmes of measures and to determine the costs to reach good ecological status and a CBA is applied separately not to compare scenarios or measures but to determine whether these costs are worth the effort by comparing them with the benefits we can expect from reaching the good status. This corresponds with the disproportionate cost analysis as defined in WATECO, 2003.

2.2. Flood risk management

Whereas surface water quality has more a tradition of cost-effectiveness analysis, flood risk management is more focused on using cost-benefit analysis. Effects in this case are not expressed as the relative contribution to reach a predefined target but as material and non-material damages prevented due to flooding. These damages can relatively easy be expressed as monetary values and included in a cost-benefit analysis. The most widely applied methodology uses hydro-dynamic models to estimate flood impacts and stage damage functions to transfer flood impacts (e.g. depth) into damage extents for buildings, infrastructure, agriculture, etc.. This is linked to pricing/monetization to assess the benefit of flood prevention and compare this benefit with the required investment and maintenance costs. First stage damage approaches appeared in the US in the 1960s. Since then methods for flood damage estimation have been developed in many other countries (e.g. de Kok and Grossman, 2010; Dutta et al., 2003; Kourgialas and Karatzas, 2012; Penning-Rowsell and Chatterton, 1977). Studies related to the Dutch Deltaplan (overview given in Bouma et al., 2005) and the Belgian Sigmaplan (chapter 5) also confirm the use of similar methods in

Belgium and the Netherlands to select an optimal strategy for flood retention. Damages are mostly restricted to material damages due to flooding, which can be extended to incorporate casualties, impacts on cultural heritage and ecosystems. Jonkman et al. (2008) went a step further in using input-output models to study the indirect economic effects. In certain cases, also other impacts (ecosystem services) as ecological or recreational effects of flood retention measures are included as co-benefits (Brouwer and van Ek, 2004).

The selection of measures applied in these examples is mostly restricted to a limited amount of scenarios (e.g. dyke heightening versus floodplain restoration) which are compared to a reference scenario. A ranking or optimization between large ranges of potential measures as is typically applied in a cost-effectiveness analysis is not performed. To determine the optimal location and design of potential floodplains and dykes to achieve flood benefits at the lowest achievable cost is however a very relevant question.

2.3. Water consumption / Droughts

Reduction of water consumption to tackle droughts is studied less often with economic appraisal techniques as compared to water quality and floods. Mostly, the objective of the CEA is to bridge the gap between the total water supply and total projected consumption in dry areas and to assess how water consumption should be distributed between sectors (agriculture versus tourism) to maximize societal welfare with the available amounts of water. In the context of the WFD, the objective is extended to achieve the good ecological status and to reach a sustainable rate of water consumption while maintaining minimum environmental stream flows and groundwater levels. Both approaches are applied in Berbel et al. (2011). They performed a cost-effectiveness analysis (cost effect ratios) of water-saving measures for the case of the Guadalquivir River Basin in Southern Spain making use of two effectiveness indicators related to pressures (the amount of m^3 reduced extraction or water saving) and impact (effect on river flow). To estimate the effect on river flow the AQUATOOL (Pulido-Velazquez et al., 2008) is used and corrects for factors as return flows (return of drinking water after sanitation) and connectivity to river systems. Blanco-Gutiérrez et al. (2011) used a multi-scale modelling approach to explore the environmental and socio-economic impacts of alternative water conservation measures at the farm- and basin-levels. In this case, the impact of economic instruments as water pricing, quota and closure of unlicensed wells is tested on the agricultural income given constraints on water use at the farm level and water use at the subbasin level. They also examine how small, medium size and large farms with different levels of water consumption and farm incomes react differently to the proposed measures for water savings. Riegels et al. (2011) used a hydro-ecological modelling approach to optimize resource costs (added value of water consumption) of compliance with ecological status requirements (minimum environmental flow).

Most modelling exercises focus on the economic and environmental impact of use restrictions, which is less relevant for most parts of the Flemish region. A different approach is performed by Haeffner (2009). He calculated cost-effect ratios comparing the cost per m^3 of a more extensive list of measures including water saving measures as rainwater harvesting, water-efficient toilets and showers, renovation of drinking water infrastructure, alternative water production techniques and reduced water consumption.

In some examples also benefits are added to compare alternative scenarios of water consumption. Benefits are typically expressed as loss of net added value due to production losses (industry, agriculture), replacement costs (use other sources of water e.g. bottled water) and willingness to pay estimates (households, recreation). Often, losses of net added value or replacement costs are considered as costs of the cost-effectiveness analysis. In some cases, technology costs (pumping, reservoirs, …) are compared with economic benefits from water use. Pulido Velazquez et al. (2008) used the AQUATOOL to maximize the net economic benefit from water use balancing agricultural, natural and urban water demand with technological costs for pumping, recharge and treatment. Birol et al. (2010) compared the technology costs and the benefits of a water-resource management plan. Benefits were based on a choice experiment, surveying the willingness to pay of farmers and other residents for improving the water quality and quantity.

2.4. Groundwater quality

With regard to groundwater quality, few applications exist of CEA/CBA. In general a distinction can be made between the site scale and the river basin scale. At the site scale, methods are applied to determine cost-effective soil remediation technologies. At the river basin scale, research efforts are mostly focused on nutrient pollution in groundwater from diffuse agricultural sources. Several examples of nutrient pollution in groundwater on a river basin scale exist (Cardenas et al., 2011; Herivaux et al., 2013; Peña-haro et al., 2009). Cardenas et al. (2011) studied the cost-effect ratios of the application of measures to mitigate nitrate leaching from grassland livestock farming systems in contrasting areas of England and Wales. Losses by leaching are estimated by using N balance models at the farm or plot scale. Peña-haro et al. (2009) described the development and application of a method for exploring optimal management of groundwater nitrate pollution from agriculture. The hydro-economic model suggests the optimal spatial and temporal distribution of fertilizer application that maximizes the net benefits in agriculture constrained by the quality requirements in groundwater at specific control sites. For this purpose an optimization model was constructed which makes use of crop yield functions, nitrate leaching functions and a pollutant concentration response matrix. This matrix shows the concentration over time at different control sites throughout the aquifer resulting from multiple pollutant sources distributed over time and space and is determined with MODFLOW simulations. Herivaux et al. (2013) use a hydro economic model to assess the cost-effectiveness of agro environment schemes in reducing nitrate concentrations in groundwater. Cost-effect ratios were used to compose cost-effective sets of measures (cost divided by reduced nitrate leaching). The potential impacts of these sets on groundwater concentrations are simulated with a 3D-hydrogeological model (M3), which is used to compute the nitrate concentrations in groundwater over a fifty-year period. In a last step, the benefits of improving nitrate concentrations are assessed based on the avoidance-cost method. This method considers that deterioration in groundwater quality creates avoidance costs for tap water producers (treatment costs) and averting expenditures for the tap water consumers supplied by this resource (use of bottled water).

More site specific pollution and groundwater and soil quality issues (e.g. brownfields) are usually less considered when discussing integrated water management. Determining the most appropriate course of action when faced with soil or groundwater contamination does however require the consideration of technologies or approaches that can feasibly remove the contamination to the required target level within realistic time and cost constraints. Economic appraisal techniques as CEA or CBA are less applied for this type of cases or integrated in more wide sustainability assessments (Cappuyns et al., 2013; Onwubuya et al., 2009; Schädler et al., 2011).

2.5. Sediments

Sediment management is still one of the key challenges in integrated water management. As the understanding of sediment systems has evolved, it has become increasingly clear that effective and sustainable management strategies must focus on the entire sediment cycle, incl. erosion losses, point source emissions, sediment transport modelling and dredging strategies (Apitz and White, 2003). Key information necessary to perform economic appraisal and compare alternative courses of action includes the identification of sources of sediment (and associated contaminants and nutrients), the transportation pathways of sediment and contaminants within and between the various environments, and the role of storage elements. Such information is often little available (Owens, 2005). Most existing applications only look into a specific part of the sediment cycle, mostly erosion prevention. Chang et al. (2012) have applied a regression equation and a SWAT model to evaluate pollutant-trapping efficiency levels of riparian buffer strips of various widths by comparing costs of instalment with benefits of reduced silt removal. Veith et al. (2003) combine a genetic optimization algorithm, an erosion model using the Universal Soil Loss Equation, a sediment transport component and a farm cost calculation module to determine cost-effective management strategies to reduce erosion on agricultural lands. Fitness scores for different criteria (costs, erosion losses, sediment yield) were estimated and weighted to determine an optimal combination of measures. Zhou et al. (2009) perform a cost-benefit analysis for a limited set of pre-defined scenarios of erosion prevention in agriculture by comparing costs of measures as buffer strips and no tillage systems with resulting benefits of reduced erosion losses on water quality, air quality and recreational activities (unit values per ton).

Benefit assessments of improved sediment management hardly exist. Apitz (2012) conceptually describes how sediment management influences ecosystem services and hence human welfare. This can potentially be a basis for improved benefit assessments. Slob et al. (2008) conceptually describe how a CBA can be applied in relation to sediment management but on a very macroscopic level. They give a comparison of the costs of dredging in the Netherlands and increasing removal speed of historic backlogs of sediment with the benefits this generates for shipping, agriculture and flood retention. Recreation and ecological benefits are also considered important but are not quantified.

2.6. Hydromorphology and ecological quality

Hydromorphological issues and ecological quality are hardly dealt with in scientific literature related to economic analysis. Though the key objective is realizing good ecological status and hydromorphological issues is one of the focal points of the directive, very little examples exist on how to select cost-effective measures improving hydromorphological and ecological status. Vaughan et al. (2009) confirm this lack of knowledge and the need for more research on the causal understanding of ecology–hydromorphology relationships and the integration of socio-economic concerns. Publications are mostly limited to ex post evaluations of the effectiveness of measures in improving hydromorphological and ecological status (Jähnig et al., 2010; Lorenz et al., 2009).

Some exceptions exist where specific ecological parameters are considered in economic analyses. Lancelot et al. (2011) used an integrated modelling approach to assess and compare costs and ecological effectiveness of nutrient reduction options for mitigating Phaeocystis colony blooms in the Southern North Sea. The applied tool was the coupled river–ocean model that calculates on the one hand nutrient export at the river outlet (Riverstrahler) as a function of nutrient emissions (Seneque) and on the other hand the ecological response of the Phaeocystis-dominated coastal sea to the observed or simulated river loads (MIRO). Kuby et al. (2005) used multi-objective optimization to analyse ecological-economic tradeoffs and to support complex decision-making associated with dam removal in a river system. The models ecological objective is to enhance salmonid migration and spawning by maximizing drainage area reconnected to the sea. The economic objective minimizes loss of hydropower and storage capacity. Both applications are still far removed from determining cost-effective solutions to reach ecological quality ratios as defined by the Water Framework Directive.

2.7. Integrated water management

Very little examples exist where a cost-effective program of measures is selected by modelling tools simultaneously optimizing the contribution to different water aspects as water quality, flood prevention, ecological quality and sediments. Whereas in some cases water quality and erosion losses are integrated (James and Pollman, 2011; Zhou et al., 2009) or the impact of flood prevention measures is considered as an additional benefit (Brouwer and van Ek, 2004), most studies are limited to one single water aspect. An exception can be found in Hajkowicz et al. (2008). They used a cost utility approach to select an optimum portfolio of water quality enhancement projects. A multi-criteria approach is used to estimate a utility parameter, including impact on water quality (nutrients), other environmental impacts as sediments, water use, with site feasibility and societal benefits including pest control, odour nuisance and flood prevention. Utility scores are compared to cost estimations for different measures. Perni and Martinez-Paz (2013) used a participatory approach to estimate the relative importance of a range of impacts (water quality, tourism, water quantity-freshwater use by agriculture, ecological quality and impact on fishery) and to estimate cost-effect ratios for a range of alternative management strategies. Recently, Burek et al. (2013) built a pan European hydro-economic model to simulate the impact of natural water retention measures on flood mitigation, water quality (nutrients) and water scarcity. First results were promising for demonstration reasons but due to its large pan European scale, its applicability for local implementations will be limited.

Gourbesville (2008) indeed confirms that until now, the problem of the current DSSs used for integrated water management and the WFD is that they have been developed for quite specific issues and do not cover the broadness of the WFD. This does not mean however that one integrated modelling systems that cover all water aspects and all regions is the way forward. Harmonization and integration of model results seems practically more feasible.

2.8. Challenges for economic appraisal in water management

Though the potential added value of applying economic appraisal techniques in integrated water management is clear, large challenges still remain to put this into practice. Water and integrated water management in particular still poses several challenges to modelling tools.

2.8.1. Integration of economic and hydrological models

The water system is complex and requires the use of hydrological models in economic appraisal to assess the impact of measures on the water system. Hydrological models include water quality models, flood simulation models and ecological quality models. As stated previously, the integration of economic models and hydrological models is also referred to in scientific literature as hydro-economic modeling. Different possibilities for integration are applied in the scientific literature. The most straightforward form of integration is to perform a scenario simulation. For a limited amount of scenarios a cost calculation is performed in combination with an impact assessment based on the outcomes of a hydrological model (e.g. Petersen et al., 2009; van Grieken et al., 2011). This can be sufficient if the number of scenarios is limited, but as the number of options increases as is often the case in integrated water management, scenario simulation alone cannot examine all possible alternatives. The number of model runs required to examine all alternatives consisting of combinations of n binary management options by simulation is 2^n. Even with the most efficient simulation software, decreasing the time and cost of a single simulation run will not overcome this combinatorial issue (Lund et al., 2008).

Searching for promising combinations of solutions in case of a large amount of potential options requires optimization procedures ("what's best" solutions). Optimization algorithms allow for identification of promising alternatives given a wide range of management options. Nevertheless, optimization algorithms have their own limitations, often requiring simplifications to accommodate optimization solution algorithms (Lund et al., 2008). Typically a simpler formulation of the water system is applied than simulation models. These simpler formulations are often derived from hydrological models. Examples include a simplified representation of the impact of measures on diffuse emissions entering the river system (chapter 3; Mewes, 2012; Lescot et al., 2013), of upstream-downstream interactions between individual waterbodies or subbasins (Hanley et al., 1998; van der Veeren and Tol, 2001) and of the relationship between reducing emissions and concentrations (chapter 4). An additional step to doublecheck the validity of these simplifications can be performed by simulating the optimal outcomes of these simplified optimization algorithms with simulation models (Panagopoulos et al., 2012). Feedback loops or iterations between economic optimization models and hydrological models may be needed to further increase accuracy of model outcomes, but this requires appropriate model interfaces among model components (Riegels et al., 2011). The most holistic approach is based on one integrated model. There is one single control unit with both the hydrologic and economic optimization component tightly interwoven in a consistent model (Heinz et al., 2007, Pulido-Velasquez et al., 2008).

Also more holistic approaches require some form of simplification. Traditional catchment models (e.g. SWAT, MIKE-SHE, INFOWORKS) which are able to simulate catchment hydrology and concentrations on a daily basis at the scale of a subcatchment of the Scheldt river basin may take several hours to weeks for a single model run. If thousands of runs are needed with traditional hydrological models to find the optimal solution for larger basins and multiple parameters simultaneously, this will be very demanding. Parallel computing with computer clusters is promising and can reduce calculation times substantially. However, this will only solve the problem partially. Another solution lies in the reduction of the model complexity (and consequently CPU time) by using "appropriate" model structures performing adequately well for the objective of the model study, keeping CPU time to a minimum (Wagener and Wheater, 1999; Fenicia et al., 2008; Gupta et al., 2008; Van Hoey et al., 2011).

2.8.2. Scaling

The possible scales for economic analysis of integrated water management can be very different. Table 1 presents an overview of potential spatial and temporal scales given by Heinz et al. (2007).

Table 1: Relevant spatial and temporal scales for integrated water management (Heinz et al., 2007)

Spatial scales	Temporal scales
Individual water user	Daily or hourly
Local water districts	Monthly
Regional	Annual
National	Steady state vs. dynamic
International	

The Water Framework Directive requires on the one hand that member states reach good water status in all waterbodies or at the local level but on the other hand that member states set up programs of measures for entire river basin districts (national or international). A combination of both scales is not straightforward. Also, typical scales for hydrological and economic models are different (Brouwer and Hofkes, 2008). Waterbodies, watersheds and basins usually are the geographical unit in hydrological models, while economic models often refer to administrative boundaries of a region (county, province, state) or a country as a whole. Whereas most economic appraisal techniques use aggregated data and are performed on a macroscopic scale, water management is mostly designed on a very local scale. Assessment on local scales requires more extensive databases and more detailed forms of analysis. Network effects are also important to take into account. Measures that impact on areas upstream also have an effect downstream and contribute to the realization of multiple objectives at multiple locations. This can however be complex, expecially if a lot of water bodies are considered.

Temporal scales as defined in the Water Framework Directive are also covering a range of scales. On the one hand, Member States need to reach good water status in 2015 or in case of time derogations in 2021 or 2027. This requires annual time scales and also the use of dynamic models that are able to assess long term impacts of abatement measures. On the other hand, environmental targets are often defined as seasonal averages, 90-percentile or absolute maximum daily values. Brouwer and Hofkes (2008) correctly declare that typical time scales are different for hydrological and economical models. Time scales in hydrological models often refer to days, months or seasons (summer and winter), while in economic models the time scales (intervals and horizon) are usually longer than that (years or decades).

2.8.3. Integration of multiple water aspects simultaneously

Integrated water management is specifically aimed to reach good water status for different aspects simultaneously (water quality and quantity, groundwater and surface water, ecological quality, sediments). Measures which might be less cost-effective for one single water aspect, can be very cost-effective for different water aspects simultaneously. Assessments in the past typically scoped on specific water aspects which may explain the focus on more engineering type of solutions (public wastewater treatment, dykes) as they are more effective for treating wastewater or preventing flooding and less on the application of more natural water retention (e.g. restoration of wetlands, bufferstrips, re-meandering). A more integrated assessment can potentially stimulate an increased implementation of so called multi-purpose measures creating win-win situations for multiple water aspects but also for other environmental objectives as climate change mitigation or air quality. This does not only require an integration of economic and hydrological models but also an integration of different types of hydrological models. As the literature review demonstrated, very little examples have been published and are still remote from use in a decision-making context.

CHAPTER 3. An economic optimization model to set up a cost-effective programme of measures to improve surface water quality at different geographical scales

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Abstract

The objective of this chapter is to develop an economic model to determine cost-effective programmes of measures for multiple parameters and multiple scales simultaneously, in this case for surface water quality. A static economic optimization model (Environmental Costing Model for Flanders - Milieukostenmodel or MKM in Dutch) is developed to select the most cost-effective abatement measures to obtain a given surface water quality target. By means of linear programming the model identifies the least-cost combination of abatement measures to satisfy multi-pollutant reduction targets. The results of the economic optimization model are used to construct scenarios which are the input for a surface water quality model that is able to simulate the impact on daily concentrations and whether concentration targets are reached.

The model was specifically built to demonstrate the importance of scale and potential cost savings due to a spatial diversification of the programme of measures. To deal with the differences in scale, optimization algorithms are extended with the possibility to include individual measures aimed at reducing individual sources and collective measures that can reduce multiple sources simultaneously at a greater scale.

Results demonstrate that cost-effectiveness depends heavily on the geographical scale of the assessment. Measures that are on average cost-effective on a regional scale are not necessarily cost-effective on a subbasin scale. Differences can be large though the same methodology for assessing costs and effects is applied across the different waterbodies and all waterbodies consist of similar lowland rivers. Optimization at the local scale to achieve similar effects of programmes of measures defined at the regional scale can provide cost savings up to 33%.

3.1. Introduction

The European Water Framework Directive (WFD), adopted in 2000, sets ambitious objectives to meet good status of all waters by 2015, 2021 or 2027. To ensure that this goal will be met, member states must publish river basin management plans every six years. The WFD requires that these plans include programmes of cost-effective measures. In the Flemish region of Belgium significant water quality improvements will indeed involve high costs for both the Government and the private sectors. Consequently, it is essential to obtain an overview of available abatement measures, their costs and emission reduction potential, and to find cost-effective solutions to reach these environmental objectives. As multiple emission sources, abatement measures, interactions and trade-offs are involved, the least cost solution cannot be determined by a simple overview of (marginal) costs and emission reduction potentials. Therefore, a hydro-economic modelling exercise is frequently performed to support the selection of the most effective measures.
Different studies dealing with cost-effective water management focused on surface water quality and more specifically nutrient losses. Methodologies described in Fröschl et al. (2008), Lise and van der Veeren (2002), Panagopoulos et al. (2011), Schleich and White (1997) and van der Veeren and Tol (2001) described the use of transport coefficients, derived from more extensive hydrological models, in an economic optimization procedure to determine cost-effective programmes of measures to reach multiple emission reduction targets simultaneously. Similar approaches where transportation of multiple pollutants are simplified for economic optimization models are for instance also applied for air pollution in the RAINS model (Schöpp et al., 1999; Amann et al., 2011).

Decision making in the context of integrated water management can be very complex. Heinz et al. (2007) correctly claim that a wide range of spatial and temporal scales need to be considered. The Water Framework Directive requires on the one hand that member states reach good water status in all waterbodies or at the local level but also on the other hand that member states set up programs of measures for entire river basin districts (national or international). A combination of both scales is not straightforward.

In a decision making context, the relevant scales are mixed and depend on the types of measures which are considered. Most investment decisions related to industry are taken on an individual company level. Decision making for agriculture can happen at different levels. Changing nutrient legislation and fixing limits for animal manure application is decided on the regional level, whereas measures to prevent erosion losses are decided on the farm or even the individual parcel level. Decision making on wastewater treatment is mostly taken at the level of individual projects connecting a street or different streets to the same wastewater treatment plant whereas households can also decide on an individual level to invest in an individual small scale wastewater treatment plant. Often, these decisions are also driven by legislation defined at a European level (Urban Wastewater Directive) or a regional level.

The Environmental Costing Model (Milieukostenmodel or MKM in Dutch) was developed to assist policymakers in designing programs of cost-effective measures to meet the criteria for a good water status. By means of linear programming the model identifies the least-cost combination of abatement measures to satisfy multi-pollutant reduction targets. The model, initially set up for the most important industrial air pollution sources (Eyckmans et al., 2005; Lodewijks et al., 2007), was adapted to optimize quality of surface water. Emission sources incorporated are industry, households, wastewater treatment plants and agriculture. Pollutants targeted are chemical oxygen demand (COD), total nitrogen (N) and total phosphor (P). An extensive database of pollution sources and potential measures was required to determine a cost-effective ranking of measures for the entire Flemish region and simultaneously take into account the different local characteristics of emission sources. This detailed database allowed to examine how a cost-effective ranking of measures depends on the geographical scale.

3.2. Study area

Flanders is a highly urbanized region with a surface of 13,521 km² and a population of more than 6 million inhabitants. The region is part of two international river basin districts, the Scheldt and the Meuse. The water system mainly consists of lowland rivers with wide valleys and slow flow velocities. Highly industrialized areas are the ports of Antwerp and Ghent. Agriculture is mainly intensive and cultivated land occupies 45% of the area.

These pressures have a considerable impact on the quality of surface water. In 2006, the concentration targets set for the modelled parameters COD, N and P were exceeded in respectively 59%, 19% and 6% of all waterbodies.

3.3. Model

The Environmental Costing Model determines by means of mixed integer programming the least-cost combination of abatement measures. The model is static and estimates the cost-effectiveness of measures for one specific year (emission data from 2006).

As quality targets are defined on the waterbody level and measures can be implemented at different geographical scales ranging from regional level to the individual source level, specific challenges need to be tackled in the model:

- The definition of the scale of the specific measures needs to correspond with the level of detail at which decisions are made in water management. For larger industrial companies this is the individual company level. For sewage infrastructure and the construction of additional wastewater treatment plants this corresponds to individual sewage projects (streets) and individual wastewater treatment plants constructed. For agriculture, this depends on the type of measure. Nutrient legislation is for instance defined at the regional level. Measures as bufferstrips and winter cover crops aiming to reduce erosion losses are decided at the local level.
- Collective measures as wastewater treatment plants can treat multiple sources simultaneously. The investment decision for this type of measures influences investment decisions of measures taken at the individual household or company level. If it is more cost-effective to install or improve a collective treatment in a certain area, it will be less cost-effective to install individual treatment for households and industry in that same area and vice versa. Connecting sources to wastewater treatment might also shift pollution from one waterbody to another, causing a 100% reduction in one waterbody but an increasing incoming load in another waterbody, which might cause the need for supplementary measures on other locations.

To deal with these challenges, typical optimization algorithms as described in Hanley et al. (1998) and van der Veeren and Tol (2001) are extended with the possibility to include individual measures aimed at reducing individual sources and collective measures that can reduce multiple sources simultaneously. In case of individual measures, decisions are made independently on an individual source level. For collective measures, decisions are made on a higher scale level.

Figure 3 graphically represents how these two levels are combined. Sources 1, 2 and 3 have the option to do nothing on the individual level (reference) or perform individual measures 1 and 2. Additionally sources 1 and 2 can be both connected to collective measure 1 or collective measure 2. Source 3 only has measures on the individual level and can only remain in the reference scenario (no treatment) in the collective level.

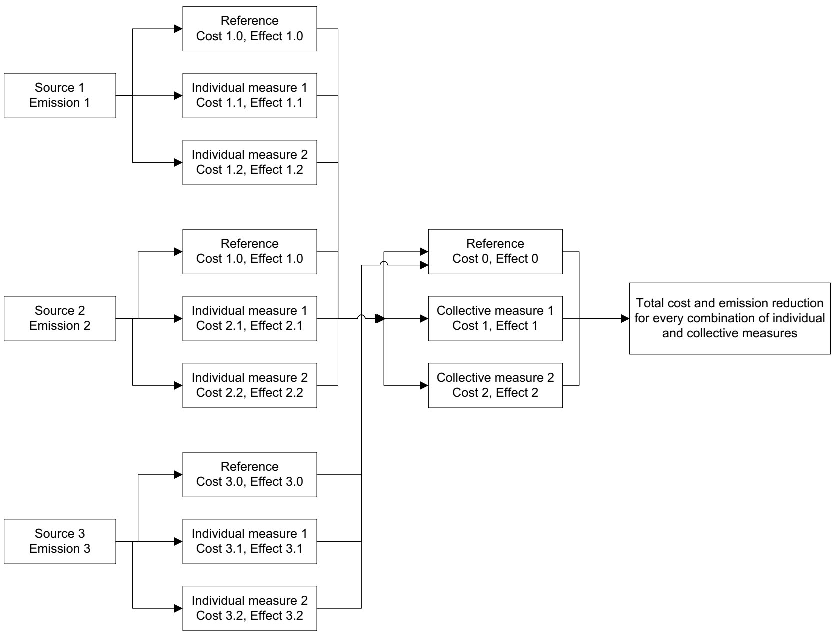

Figure 3: Graphical representation of individual and collective measures for 3 sources.

The objective of the optimization algorithm as stated in equation 1 is to determine a cost-effective ranking of abatement measures. A straightforward cost effect ratio calculation (Lescot et al., 2013; Mewes, 2012; Petersen et al., 2009; van Grieken et al., 2011) is difficult to apply as marginal effects are interdependent and are influenced by the cost-effectiveness of other measures reducing the same source and the cost-effectiveness of collective measures. Instead, a charge t_p on the unabated emissions *Ep* is introduced in the objective function. If the marginal cost of implementing a measure is lower than the charge, the measure will be implemented. If this is not the case, the charge will be paid. By gradually increasing the charge, more measures will be implemented and by comparing the implemented measures from sequential model runs a cost-effective ranking of measures can be set up. The differences in costs and effects between sequential model runs are used to determine the marginal costs of measures which are implemented additionally.

As stated, a distinction is made between individual measures *mi*, implemented for a single source, and collective measures *mc*, which have an impact on multiple sources at once. Individual measures are for instance individual end-of-pipe treatments for one industrial company or a single household. A collective measure is for instance a collective wastewater treatment plant that treats wastewater of a large number of households and several industrial companies simultaneously. Collective measures have an impact both on emissions from households and industrial plants. Costs made to improve collective measures cannot be attributed to individual sources. In equations 2 and 3 a distinction is made between costs and effects that are attributable to an individual source and collective costs that cannot be attributed to individual sources but have an effect on multiple sources. Total costs *C,* expressed in €/year*,* comprise both the discounted investment costs and annual operational costs. A fixed discount rate of 5% was used. The unabated emissions E_p (expressed in kg/year) are calculated by multiplying the sum of emissions in the reference year 2006 ($E0_{s,p}$) with the estimated efficiencies *R* (percentages) of individual and collective measures. The efficiencies are expressed as a percentage by which emissions are reduced.

For each individual source, a single combination of one individual measure and one collective measure can be implemented (decision variable $\alpha_{s,mi,mc}$). The implementation of a collective measure (the construction of a new collective wastewater treatment plant), is not decided on the individual source level. The collective measure (decision variable α_{mc}) will be implemented simultaneously for all sources for which this measure can be selected. This relationship between α_{mc} and $\alpha_{s,mi,mc}$ is stated in equation 4. If for one of the sources the collective measure is preferred above the individual measure, the collective measure will be implemented (equation 4) and the total amount of costs to install this measure is taken into account (equation 2).

As mixed integer programming techniques are applied and decision variables cannot be multiplied in one equation, only one combination of individual and collective measures can be implemented for a specific source. This is represented in equation 5. This has the consequence that combinations of measures (for instance, a secondary and tertiary treatment for industry) are included as a separate measure. This has the disadvantage that the list of measures becomes more extensive. The reference scenario also needs to be represented as a measure with zero cost and efficiency and combinations of measures on the same source are also included as a separate measure. Advantages of this approach are the possibility to include combined effects which differ from the sum of the individual effects of measures which can be the case for industrial treatment (combinations of secondary and tertiary treatment) and the possibility to predefine only feasible combinations of measures (for instance, different levels of more strict nutrient legislation cannot be combined as this would lead to double counting the effects). Equation 5 also guarantees that all emissions represented in equation 3 are included in the target function.

$$\underset{\alpha}{Min}\left(C + \sum_{p=1}^{q} t_p E_p \right) \quad (1)$$

Subject to:

$$C = \sum_{mc=1}^{c}\sum_{mi=1}^{i}\sum_{s=1}^{n} \alpha_{s,mi,mc} C_{s,mi} + \sum_{mc=1}^{c} \alpha_{mc} C_{mc} \quad (2)$$

$$E_p = \sum_{mc=1}^{c}\sum_{mi=1}^{i}\sum_{s=1}^{n} E0_{s,p} \times \left(1 - R_{mi,p}\right) \times \left(1 - R_{mc,p}\right) \times \alpha_{s,mi,mc} \quad (3)$$

$$\begin{cases} \sum_{s=1}^{n}\sum_{mi=1}^{i}\alpha_{s,mi,mc} \geq 1 \Rightarrow \alpha_{mc} = 1 \\ \sum_{s=1}^{n}\sum_{mi=1}^{i}\alpha_{s,mi,mc} = 0 \Rightarrow \alpha_{mc} = 0 \end{cases} \quad \forall mc \tag{4}$$

$$\sum_{mc=1}^{c}\sum_{mi=1}^{i}\alpha_{s,mi,mc} = 1 \quad \forall s \tag{5}$$

$$\alpha_{s,mi,mc} \in \{0,1\} \quad \forall s, mi, mc \tag{6}$$

Besides ranking, which is usually performed for one specific pollutant, the model can also be used to estimate the minimum costs required to satisfy multi-pollutant and multi-location reduction targets. In this case, no charge is introduced in the target function (equation 1), but additional constraints are defined (equations 7 and 8). Equation 7 is used to estimate the amount of emissions emitted in every specific waterbody or in waterbodies upstream. The transport coefficient *tr* represents the spatial interdependency between emission points and waterbodies for which emission targets *ET* are defined. Emission points are defined at the collective level as they might shift to other waterbodies due to connection to a collective treatment. If upstream emissions do not affect specific waterbodies, transport coefficients are set to zero. If upstream emissions do affect specific waterbodies, they are set to one. This is a simplification as not all upstream emissions will reach downstream areas due to biochemical processes. Emission points are defined on the level of collective measures as it often occurs that these points shift to other waterbodies when households or industry are connected to wastewater treatment plants.

$$E_{p,w} = \sum_{mc=1}^{c}\sum_{mi=1}^{i}\sum_{s=1}^{n} E0_{s,p} \times \left(1 - R_{mi,p}\right) \times \left(1 - R_{mc,p}\right) \times \alpha_{s,mi,mc} \times tr_{mc,w} \tag{7}$$

$$E_{p,w} \leq ET_{p,w} \tag{8}$$

3.4. Data

3.4.1. Emission sources

The different emission sources of P, N and COD incorporated in the model are listed in Table 2. The emissions from industrial sources and waste water treatment plants are based on observed emissions, monitored by the Flemish Environment Agency. The households not connected to a WWTP are a large source of pollution of surface water. In Flanders, 67% of the households was connected to collective treatment in 2006. An additional 20% is connected to a sewage system, but these sewages still have to be connected to a treatment plant. The remaining 13% is not connected to a sewage system. As the costs of connecting these households to a sewage system depend heavily on local circumstances (distance required connecting to sewage system, infrastructure required to connect sewage to WWTP), a high level of detail was required. Households situated in the same street or street network that will be connected simultaneously to a sewage system are included as a single source. To estimate nutrient losses from agriculture, the SENTWA model (System for the Evaluation of Nutrient Transport to Water) developed by Nolte et al. (1991) and adapted by Pauwelyn et al. (1997) and Verlinden et al. (2002), was used. The model calculates nutrient losses to surface water, using data on livestock numbers, nutrient excretions, manure transports and nutrient inputs on cultivated land; hydrologic, geomorphic and meteorological conditions; and soil use and agricultural techniques and practices. The model takes different types of nutrient losses into account: direct losses (e.g. loss of mineral fertilizers during application or animal manure during pasturing), subsurface runoff and surface runoff (direct or linked with erosion). Losses are calculated based on precipitation, crop development stage and spread of agricultural activities. SENTWA results are available at the scale of hydrographic zones, which is a scale similar to waterbodies.

Table 2: Emission sources

Source	Data
Industry	Measurements Flemish Environment Agency outgoing loads for 1479 individual companies.
WWTP	Measurements Flemish Environment Agency incoming and outgoing loads for 246 individual stations.
Households not connected	Amount of people not connected at municipal level multiplied by emission factors Flemish Environment Agency (N: 9,7 g/day/IE, P: 1,4 g/day/IE, COD: 89 g/day/IE. People are grouped into individual sources depending on specific sewage or wastewater treatment construction projects (7350 separate sources)
Agriculture	Diffuse emissions estimated by SENTWA model for 265 hydrographic zones.

More information on the data is available in annex 1.

3.4.2. Measures

3.4.2.1. Basic versus supplementary measures

The Water Framework Directive makes a clear distinction between basic and supplementary measures. “Basic” measures are first of all measures necessary to comply with existing European or national water legislation, such as the Urban Wastewater Directive and the Nitrates Directive. Also measures that are already foreseen in ongoing policy are part of the basic measures. For instance, as off 2007 a new Manure Action Plan became effective, which enforced stricter manuring limits in comparison with the reference situation of 2006 and compulsory manure processing if farm gate nitrogen surplus exceeds 5000 kg N/year. "Supplementary" measures are those measures designed and implemented in addition to the basic measures, with the aim of achieving a good water status. (European Commission, 2000). According to the WFD, basic measures are to be implemented under all circumstances and are no subject of the cost-effectiveness analysis. However, for comparison with the reference situation of 2006, it is still important to include the impact of basic measures in the analysis. The reduction realized by basic measures decreases the reduction potential of supplementary measures and decreases the cost efficiency. Also, it is necessary to estimate the total reduction potential of all measures, including the basic measures. By setting costs of basic measures at zero, these measures are all selected first in a cost-effective ranking of measures and do not influence the ranking of supplementary measures.

An overview of basic and supplementary measures included in the model is given in Table 3.

Table 3: Basic and supplementary measures

Source	Measure	Basic	Sup.
Industry	Maximum concentration targets BAT	x	
	Maximum concentration targets Urban Wastewater Directive		x
WWTP	Construction new or renovate existing WWTP > 2000 IE to reach efficiency targets Urban Wastewater Directive	x	
	Construction new or renovate existing WWTP < 2000 IE to reach efficiency targets Urban Wastewater Directive		x
Households not treated by a WWTP	Connecting sewage to existing treatment with new collectors (projects planned before or during 2006)	x	
	Connecting sewage to existing treatment with new collectors (projects planned after 2006)		x
	Extending the sewage system (grouped according to cost compared with small scale individual treatment households: cost sewage < cost individual treatment (low-cost sewage), cost sewage < 2 x cost individual treatment (medium-cost sewage), cost sewage > 2 x cost individual treatment (high-cost sewage)		x
	Small scale individual treatment for remote houses		x
Agriculture	Existing nutrient legislation ((including EU Nitrates Directive derogation)	x	
	Livestock reduction (poultry and other livestock)		x
	Increased dairy cattle efficiency		x
	Increased feed efficiency (pigs and poultry)		x
	More strict nutrient legislation (exclusion of Nitrates Directive derogation)		x
	Tuned fertilisation (only up to crop requirements)		x
	Buffer strips along watercourses		x
	Reduced tillage		x
	Winter cover crops		x

3.4.2.2. Industry

When setting up potential measures for industry, a distinction was made between companies emitting directly to surface water and companies connected to a WWTP. For companies connected to a WWTP, no supplementary measures were defined. Measures aimed at improving the efficiency of WWTP also have impact on the industrial emissions treated by the plant. It was not examined if companies emitting directly to surface water in the reference scenario can be connected to a WWTP.

The starting point for setting up additional measures for individual industrial companies is concentration targets. Based on observed concentrations and the predefined targets, the required reduction to reach these targets was estimated. A distinction was made between targets based on Best Available Technologies (BAT), which are the basic measure, and more strict industrial concentration targets based on the targets defined for WWTPs in the Urban Wastewater Directive (UWWD).

Once the required reduction potential for each company was calculated, potential end-of-pipe technologies were selected to estimate the costs. Based on expert knowledge, a simplified stepwise approach was applied for each individual company based on the observed concentration, the industrial sector of the company and the technologies already implemented. Costs and efficiency of the available technologies are given in Table 4. The estimated physical lifespan of industrial end-of-pipe technologies was assumed to be 10 years.

Table 4: Costs and removal rate end-of-pipe technologies industry

Technology	Removal rate (%)			Investment (k€)			Maintenance (€/m^3)
	COD	N	P	< 50m^3/day	50-500m^3/day	500-5,000m^3/day	
Physico-chemical treatment	10-60	10-20	65-75	25-100	100-400	400-900	0.55-1
Biological treatment (aerobic)	70-90	20-30	30-50	20-100	100-600	600-4000	0.45-1.5
Biological nitrogen removal	80-95	80-85	30-50	/	100-200	200-1000	0.02-0.08
Biological phosphorus removal	10-30	5-10	80-85	25-75	75-350	350-600	0.15-0.55
Sand filtration	5-10	5-10	5-10	5-10	10-25	25-75	0.05-0.10

3.4.2.3. Urban Wastewater Treatment Plants (WWTPs)

A series of investment and optimization projects to construct new WWTPs and renovate existing WWTPs was defined by the Flemish Ministry of the Environment to comply with the European Urban Wastewater Directive and to further optimize the existing infrastructure. Aquafin, the company that operates all supra municipal infrastructure (main sewers and wastewater treatment plants) that is needed to treat domestic wastewater in Flanders, made detailed cost estimates for all these projects. Besides these known projects, additional renovation projects were defined when the observed efficiency of stations was below the legal objective as defined in Table 5. Cost estimates were based on average renovation costs of similar treatment plants. The physical lifespan of a WWTP and renovations was estimated at 40 years.

The expected efficiency gains after renovating an existing WWTP were based on the legal targets as given in Table 5. When the observed efficiency in 2006 for a certain parameter was below the legal target, it was assumed that the efficiency increases after renovation up to this target. For new WWTPs the efficiency estimates were based on the legal targets and the average observed efficiencies of comparable WWTPs using the same technology and having a similar capacity. If the average observed efficiency was above the legal target, this average is applied instead of the legal target. This was especially the case for WWTPs with a capacity of more than 2.000 inhabitants. Depending on the technology applied, average observed efficiencies for COD are between 85% and 95% and for P between 80% and 90%.

Table 5: Removal rate targets WWTPs in Flemish region

Capacity (IE)	COD (%)	N (%)	P (%)
>10.000	75	80	80
4.000 – 10.000	75	80	80
2.000 – 4.000	75	60	80
< 2.000	75	/	/

3.4.2.4. Household wastewater not treated by a WWTP

A distinction was made between households connected to a sewage system but not to a WWTP and households not connected to a sewage system. The Flemish Ministry of the Environment defined projects for households to extend the main sewage collector network and connect existing sewage to a WWTP with new collectors. These projects are similar to the WWTP investment- and optimization projects mentioned above. Costs were also assessed for each individual project (Aquafin, 2006).

For the majority of the households not connected to sewage, the construction of additional sewage was included as a measure. For the remaining, most remote houses, the construction of a small scale individual treatment plant is assessed. 3 types of sewage projects were distinguished depending on the costs to connect these households to the sewage network, compared with the costs to install an individual treatment.

3.4.2.5. Agriculture

For agriculture a series of potential abatement measures can be distinguished, i.e. measures aimed at reducing nutrient production by cattle, restricting nutrient application to crops and reducing nutrient losses from fields. More details on the costs and effects of measures for agriculture can be found in ILVO, 2007.

Nutrient production abatement

- Livestock reduction

Between 2001 and 2004 the Flemish government organized a buyout for farmers who ceased livestock production. Here a similar reduction in livestock numbers was assumed, using the Standard Gross Margin per animal as the yearly cost of production capacity lost. As the spread of response to a virtual buyout is impossible to predict, the previous reduction percentage per animal category was applied per community. Following Deuninck (2006) it was also assumed that only small farms would accept buyout.

- Increased dairy cattle efficiency

Increasing milk production per cow (from the current mean of 7,156 kg/cow·year to 9,000 kg/cow·year obtained by the more efficient farms) results in an increasing excretion per cow. However, as less cows, and consequently less replacement cattle, are needed to produce the same amount of milk, total excretion is decreased (Wustenberghs et al., 2007). The efficiency increase can only be attained by decreasing grazing time and increasing the share of concentrates compared to roughage in the cows' feed ration. This implies additional feeding costs, but also part of the grassland acreage that can be replaced with more profitable arable cultures. Based on ADLO (2003) it was calculated that this measure would result in an income increase of 0.71 €/100 l milk. This means that an environmental benefit is obtained at negative costs. This measure is not a measure aimed at improving water quality as such, but can be considered as an environmental benefit due to the expected improvement in efficiency of dairy farming in the upcoming years.

- Increased feed efficiency for pigs and poultry

So far, increasing feed efficiency in intensive animal husbandry was mainly focused on phosphorus. Improvements are still possible for nitrogen. By better tuning the protein content of pig or poultry feed to their needs in different growth phases, their nutrient excretion is decreased. These feeds are more expensive, especially for poultry. For pigs, the installations necessary for phased feeding require additional investments. Feeding costs are similar to the existing situation. When implementing this measure, the cost per kg excretion reduction is 3.6 times higher for poultry compared to pigs (ILVO, 2007).

Nutrient application abatement

- Tuned fertilization

Tuned fertilization means that fertilization is tuned to crop requirements (Vanden Auweele et al., 2004), instead of manuring fields and grasslands at inappropriate times to dispose of manure. Professional advice based on soil analysis can help farmers to achieve this. Manure is only applied up to the crops' N requirements or to the legal limits and chemical fertilizer is only added if N needs are larger than what can legally be supplied by manure. P needs are assumed to be fulfilled by animal manure. This means a reduction of mineral N and P application with resp. 37% and 100%. However, advisory costs outweigh the economies on chemical fertilizer.

- Manure export or processing (exclusion of EU Nitrates Directive derogation)

The possibility to request derogation from the EU Nitrates Directive was foreseen as a basic measure for agriculture. The basic limits for animal manure application are 250 kg N/ha·year on grassland and 200 kg N/ha·year on maize cultivated after grass, winter wheat followed by a winter cover crop and beets. Without this derogation the limit is 170 kg N/ha·year on all crops. Compared with the basic measure, this reduces the manure disposal capacity by 4% (Claeys et al., 2008) and manure export or processing needs to increase with 48% compared to the basic scenario. Processing costs are estimated specifically per animal type (0.704 €/kg N poultry, 3.46 €/kg N pigs, 4.72 €/kg N cattle).

Nutrient loss abatement

- Grass buffer strips

Buffer strips along watercourses are estimated to reduce particle runoff from fields by 51 to 94% (MESAM, 2007). Surface water pollution by particle bound phosphate and COD of surface water is supposed to be reduced proportionally. Buffer strips are only taken into account for phosphorus and not for nitrate pollution abatement, as they have little effect through denitrification and nitrate is mainly lost through subsurface runoff (Dhondt et al., 2001). Cost of buffer strips are related to installation and maintenance, the loss of added value from previous production and reduced by the remaining grass production value. For the cost estimations 6 m wide grass strips were assumed to be installed on all plots next to a watercourse that are potentially erosion sensitive and bear annual crops. This amounts to 0.6% of the utilized agricultural area.

- Reduced tillage

Reduced tillage is a farming practice that minimizes soil disturbance and allows crop residue to remain on the ground instead of being thrown away or incorporated into the soil. It is estimated that this technique reduces particle runoff by 42% (MESAM, 2007) to 93% (Gillijns et al., 2004). The costs of reduced tillage are buying the appropriate machinery and production losses. The production losses can range from 0 to 60€/ha (Hubrechts, 2006). Reduced tillage is supposed to be practiced on all potentially erosion sensitive plots that bear annual crops. This amounts to 12.2% of the utilized agricultural area.

- Winter cover crops

Winter cover crops reduce erosion and take up nutrients (especially nitrogen) that remain in the soil after the main crop is harvested. N losses can be reduced by 25 to 35 kg N/ha·year (den Boer et al., 2002). P- and COD-losses are estimated at the same level as runoff reduction, i.e. 50% (van der Welle and Decleer, 2001) and 100% (MESAM, 2007). The costs are related to buying seed and cultivation. Cost savings are realized thanks to a decreased need for chemical fertilizer. Winter cover crops can be sown on all plots where the main crop is harvested early enough to allow sufficient development before winter.

3.5. Results

3.5.1. Cost-effective ranking of measures on a regional scale

To set up a cost-effective ranking on a regional scale, the cost and reduction potential of the measures in the database of the ECM were aggregated to the regional Flemish scale and used as input for the model (more information on the data can be found in annex 1). Sequential optimization runs were performed with gradually increasing charges t_p in the target function for every individual pollutant. By comparing solutions between sequential runs, the additional costs and effects can be derived to calculate marginal costs.

The marginal abatement cost curves represented in Figure 4, Figure 5 and Figure 6 were set up in preparation of the draft river basin management plans and the draft program of measures of Flanders published in 2008. (Coördinatie Integraal Waterbeleid, 2008). The objective was to determine a cost-effective ranking of measures on the scale of the region as a whole.

The most cost-effective measures for reducing COD emissions in Flanders proved to be renovation of existing WWTPs and construction of new ones, and connecting existing sewage networks with collectors to these plants (Figure 4). Extending industrial end-of-pipe treatment to achieve concentration targets imposed by the Urban Wastewater Directive (UWWD) is more effective than extending the sewage network, though the impact is limited. As the contribution of agriculture to COD is unknown, the impact of agricultural measures on COD could not be estimated. These measures are not included in the cost curve for COD.

Agricultural measures are important for reducing emissions of N and P (Figure 5 and Figure 6). Some agricultural measures were very cost-effective compared to measures aimed at reducing N emissions from household or industrial point sources. Cost-effective environmental measures in agriculture include mainly measures that also improve farm efficiency: efficient dairy cattle, winter cover crops and feed efficiency for pigs. Other agricultural measures, such as reducing livestock and increasing feed efficiency for poultry proved less cost-effective than measures concerning WWTP, collective wastewater treatment for households and industry. Renovating existing and constructing new WWTPs and connecting existing sewage networks with collectors is more cost-effective than connecting additional households to the sewage network. The efficiency for tuned fertilization, buffer strips and reduced tillage is comparable with renovating and constructing WWTPs. However, the impact of these measures is limited to either N or P. Individual wastewater treatment for households is very cost ineffective for nutrients.

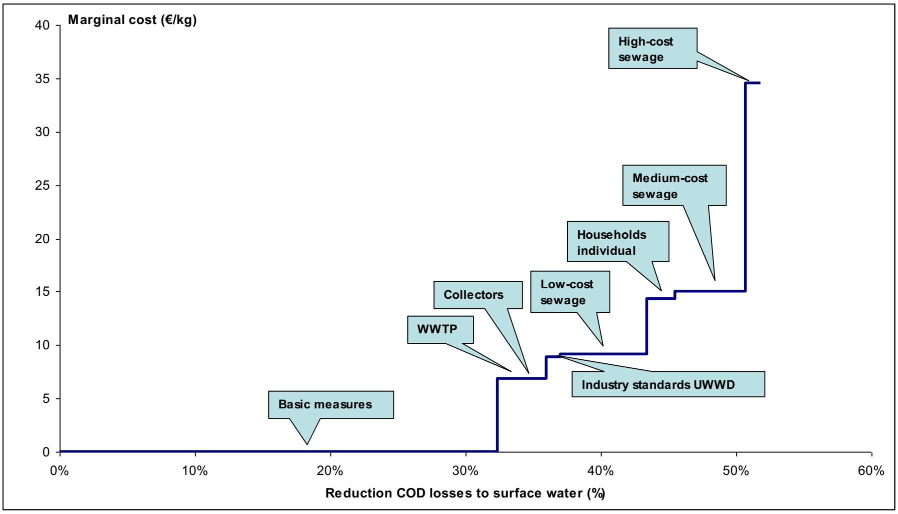

Figure 4: Marginal abatement cost curves for reducing COD losses in Flanders on regional scale

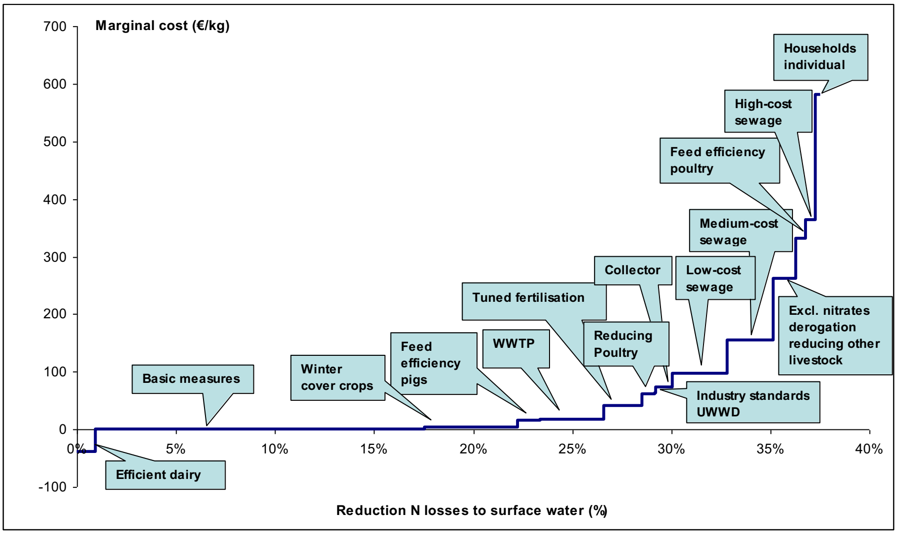

Figure 5: Marginal abatement cost curves for reducing N losses in Flanders on regional scale

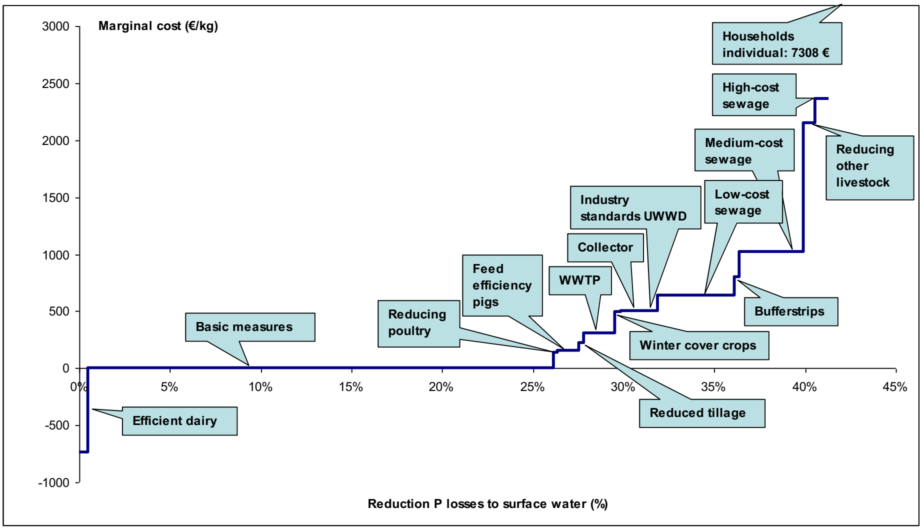

Figure 6: Marginal abatement cost curves for reducing P losses in Flanders on regional scale

Based on the cost-effectiveness analysis for the Flemish region and other considerations as stakeholder acceptance and technical constraints, a package of supplementary measures to be implemented by 2015 was selected for the first river basin management plan. This package included renovating existing and constructing new WWTPs and connecting existing sewage networks with collectors to these plants, extending industrial end-of-pipe treatment to achieve UWWD concentration targets, efficient dairy cattle, winter cover crops, improve feed efficiency for pigs, tuned fertilization and buffer strips. This package represents a total annual cost of 104 million €. The total regional amount of emissions for COD, N and P are reduced with resp. 44%, 30% and 29%. Marginal costs for the most expensive measures in this package are 9 €/kg COD (UWWD targets industry), 74 €/kg N (connecting existing sewage networks with collectors) and 800 €/kg P (bufferstrips).

3.5.2. Impact of scale on cost-effective ranking of measures

A remark often given during public consultation of the draft program of measures was that a cost-effective ranking of measures might differ significantly between waterbodies or subbasins. Therefore the cost-effectiveness analysis was also performed for all 11 Flemish subbasins separately. Results in Figure 7 indeed indicate that marginal costs on a subbasin scale can differ heavily from averages on a regional scale. The most cost-effective measures on a regional scale (efficient dairy cattle, winter cover crops, improved feed efficiency for pigs) are still the most cost-effective on a subbasin scale. However, the other measures included in the cost-effective program of measures for 2015 (renovating existing and constructing new WWTPs, connecting existing sewage networks with collectors to these plants, extending industrial end-of-pipe treatment to achieve UWWD concentration targets) were not always more cost-effective on a subbasin scale compared to other measures not included in this program (low-cost sewage, exclusion of nitrates derogation, reducing livestock). This is strongly related to local conditions. Sewage costs are influenced by the degree of urban sprawl and population density. If urban areas are less populated and on average more sewage is required to connect the same amount of people, this measure becomes less cost-effective. Also, the degree of population already treated collectively at the moment differs heavily between subbasins. Basins as the Brugse Polders and Maas have a relatively high treatment rate (up to 90%), whereas the Ijzer, Leie and the Bovenschelde have a relatively low treatment rate (below 60%). This means that the quick wins for basins as the Brugse Polders and Maas are already made in the past, whereas this is not the case for the others. Cost-effectiveness for agricultural measures also has large differences. This is due to differences in the agricultural sector across subbasins and differences in impacts of reducing diffuse emissions due to soil texture, with more immediate impacts in sandy soils compared to clay and loam soils. As we scope on short term impacts with the SENTWA model, we can expect more impact in sandy soils and lower cost effect ratios.

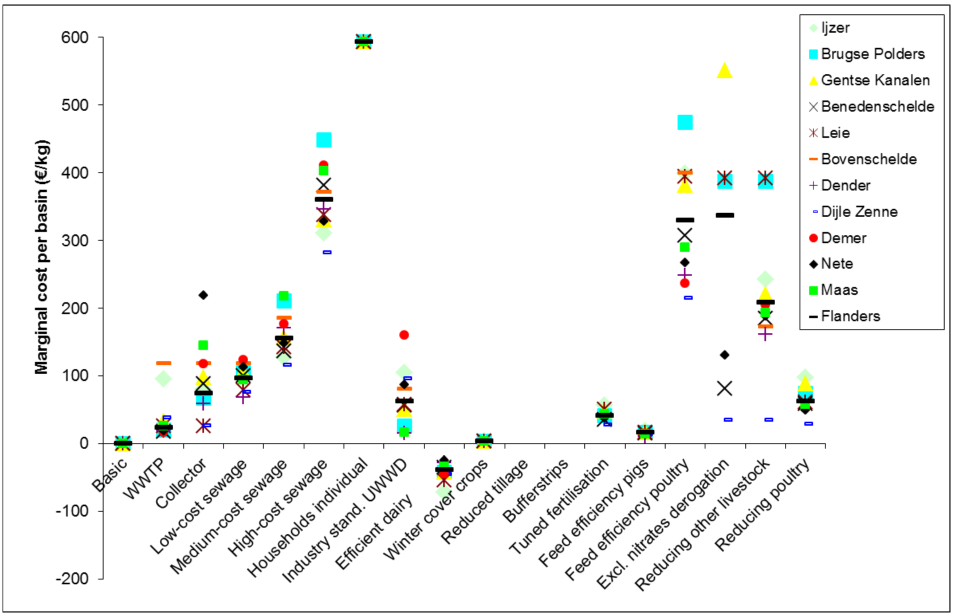

Figure 7: Marginal abatement costs N of supplementary measures on a subbasin scale. (Each point reflects the average marginal cost (Y-axis) per subbasin for all measures in the database (X-axis))

To illustrate further differences on the local scale (individual sources) Figure 8 compares the marginal abatement cost curve for simulations performed at the regional scale (measures implemented simultaneously for entire region) and at the local scale (measures can be implemented individually at the source level). At the cutoff point of 30%, marginal costs are similar but the total annual costs to achieve this objective are reduced with 25 million €/year (25%). Major differences between measures selected at the local scale vs. measures selected at the regional scale are related to industry and agriculture. At the local scale, the realization of UWWD targets for industry is rarely selected, where this is the case for the regional scale. For agriculture, reducing poultry and exclusion of nitrates derogation is often selected at the local scale whereas this is not the case on regional scale. In a limited amount of waterbodies, the construction of low-cost and medium-cost sewage to remote houses is selected. The construction of high-cost sewage and individual wastewater treatment for households is never selected in both scales.

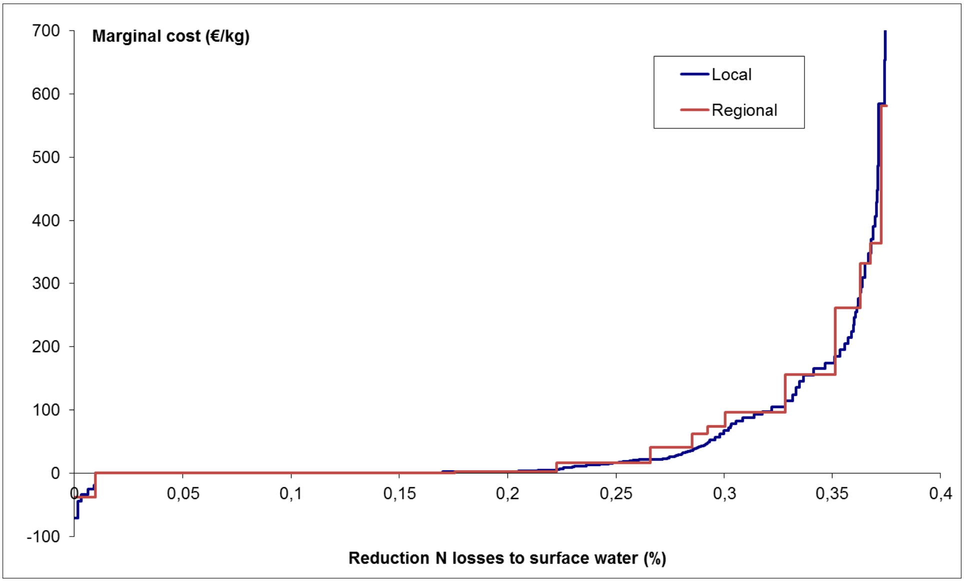

Figure 8: Marginal abatement cost curves N of supplementary measures on a regional vs. local scale.

3.5.3. Impact of scale on multi-objective optimization results

To demonstrate the potential impact of scale on multi-objective optimization, the impacts of the selected supplementary measures in the first river basin management plan (regional level) on the emissions in every individual waterbody are used as emission reduction targets (Table 6). If the program of measures is not diversified on a local scale, the total annual cost is equal to the annual cost of the scenario discussed before (104 million €). If the selection of measures can be defined differently for each source and equal emission reductions need to be achieved for every waterbody individually, the total annual costs decrease with 22%. If it is also allowed to achieve similar emission reductions in the waterbody itself and upstream areas, the total annual costs decrease with 33%.

Measures which are much less selected compared to the regional scenario are the realization of UWWD targets for industry, the construction or renovation of WWTP and collectors for households and bufferstrips for agriculture. Measures which are selected often at the local scale but which are not included in the regional program of measures are low-cost sewage and livestock reduction of poultry.

Table 6: Implementation levels and costs of the regional program of measures vs. source specific multi-objective optimization at local waterbodies or local waterbodies including upstream areas

Source	**Measure**	**Region**	**Local**	**Local upstream**
Industry	Maximum concentration targets Urban Wastewater Directive	100%	33%	34%
Households	Construction new or renovate existing WWTP < 2000 IE to reach efficiency targets Urban Wastewater Directive	100%	51%	42%
	Connecting sewage to existing treatment with new collectors (projects planned after 2006)	100%	45%	32%
	Extending the sewage system (low-cost sewage)	0%	23%	24%
	Extending the sewage system (medium-cost sewage)	0%	9%	9%
	Extending the sewage system (high-cost sewage)	0%	1%	0%
	Small scale individual treatment for remote houses	0%	0%	0%
Agriculture	Livestock reduction (poultry)	0%	52%	82%
	Livestock reduction (other livestock)	0%	1%	1%
	Increased dairy cattle efficiency	100%	98%	99%
	Increased feed efficiency (pigs)	100%	96%	97%
	Increased feed efficiency (poultry)	0%	1%	0%
	More strict nutrient legislation (exclusion of Nitrates Directive derogation)	0%	2%	2%
	Tuned fertilisation (only up to crop requirements)	100%	90%	77%
	Buffer strips along watercourses	100%	19%	6%
	Reduced tillage	0%	1%	3%
	Winter cover crops	100%	98%	97%
Total annual costs (million €)		**104**	**81**	**70**
Cost savings vs. regional approach (%)		**0%**	**22%**	**33%**

3.6. Impact on water quality

The economic model represented before uses optimization procedures to achieve emission reduction targets. To assess the impact on water quality and make a comparison with the concentration targets representing the good status, a hydrological model is required. The impact of the draft program of measures on surface water quality was simulated with the Pégase model (Smitz et al., 1999; Deliège et al., 2010). This model was specifically built for the Scheldt basin (large majority of waterbodies in Flanders) to estimate the impact of scenarios on water quality. The same emission data and data on efficiency of measures were used in Pégase as in the Environmental Costing Model. Additionally, the model uses geographical data on water courses, hydrological contours, digital terrain models, land cover and hydro-meteorological data of daily water flows, daily temperatures to model water quantity and quality at the scale of individual waterbodies (main river stretches). More information on the river network model and the calibration results can be found in annex 2. The model simulates hydrological flow, water temperature and water quality (COD, BOD, N, P, Dissolved Oxygen) on a daily basis, taking into account transportation and dilution processes, biochemical processes and interaction between surface water, air and sediments. Calibration results in paragraph 3.9 and additional comparisons between model results and observed concentrations at approximately 100 observation points showed that the model is less reliable for N and P, especially at the basins of the Leie, the Dijle and the Bovenschelde (Aquapole, 2008) and that results for these parameters need to be interpreted carefully.

In Figure 9, the results of the model simulation with Pégase are represented at the level of the waterbodies. A distinction was made between waterbodies reaching very good, good, moderate, insufficient and bad water status. Concentrations modelled at the outflow points of every waterbody were compared with the legal standards. Four different scenarios were simulated: the reference scenario 2006, the basic scenario implementing all basic measures, the supplementary 2015 scenario implementing the supplementary measures considered cost-effective (see paragraph 3.5.1) and a supplementary maximum scenario implementing all supplementary measures discussed before.

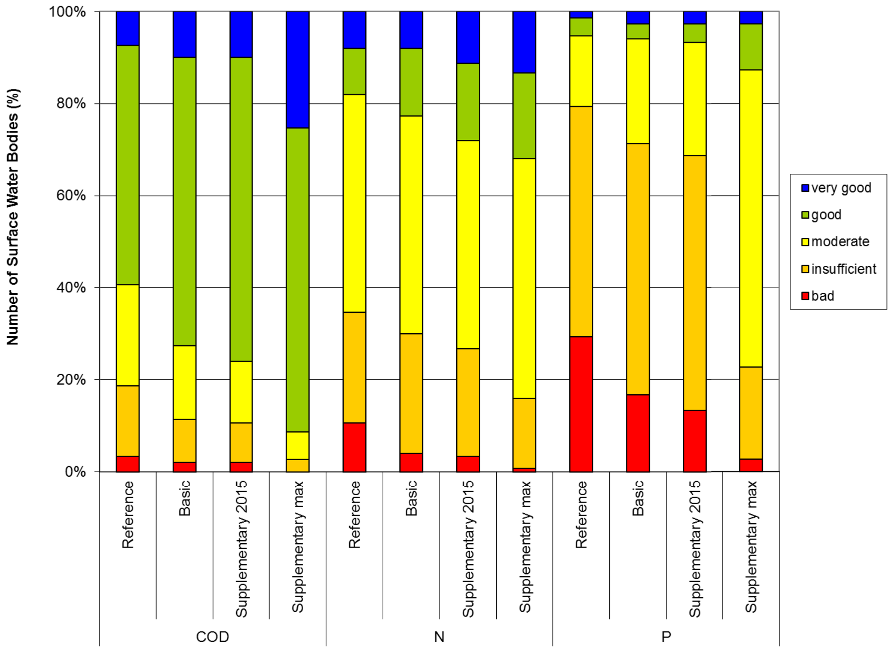

Figure 9: Impact of basic and supplementary measures on the amount of waterbodies reaching good status (Coördinatiecommissie Integraal Waterbeleid, 2009b)

Results for the reference scenario indicated the poor surface water quality in Flanders in 2006. After implementing the basic measures and supplementary measures the amount of waterbodies reaching good status increased only slightly. Even the results for the maximum scenario show that for N and P only a small amount of waterbodies (resp. 32% and 15%) would reach good status. This is why for all waterbodies exemptions were requested in the first draft river basin management plans (Coördinatiecommissie Integraal Waterbeleid, 2009b+c).

3.7. Discussion

An economic model was developed to determine cost-effective programmes of measures for multiple parameters and multiple scales simultaneously. By means of linear programming the model identifies the least-cost combination of abatement measures to satisfy multi-pollutant reduction targets. The model was specifically built to demonstrate the importance of scale and potential cost savings due to a spatial diversification of the programme of measures. To deal with the differences in scale, typical optimization algorithms as described in Hanley et al. (1998) and van der Veeren and Tol (2001) are extended in this chapter with the possibility to include individual measures aimed at reducing individual sources and collective measures that can reduce multiple sources simultaneously. For individual measures decisions are made independently on an individual source level. For collective measures, investment decisions are made on a higher scale level. Costs cannot be divided across all individual sources, influenced by a collective measure, but need to be considered in total for the optimization. Also, emission points can shift between waterbodies due to the implementation of measures. This causes emissions to reduce in one waterbody but to increase in another waterbody, which might cause the need for additional investments elsewhere. The impact of potential spatial shifts was also taken into account in the optimization model.

A number of weaknesses in the model set-up can still be identified. The estimation of emissions from different sources is an important starting point of the cost-effectiveness analysis. Not for all sources, these estimations are equally reliable. Emissions from industry and WWTP are based on monitoring data. This is however not the case for agriculture and households not connected to a wastewater treatment plant. Agricultural emissions are derived from the SENTWA model. Research performed by Seuntjens et al. (2008) and the ongoing development of ArcNemo, a new spatially distributed nutrient emission model to quantify N and P losses from agriculture to surface waters (Van Opstal et al., 2013) demonstrate the importance of considering nutrient emissions more dynamically. It can be expected that the nutrient emissions coming from groundwater due to excessive use of manure in the past and also the reduction potential of agricultural measures are underestimated by the SENTWA model. SENTWA is only able to predict the short-term impact (1 year) of reduced nutrient use on nutrient losses. This causes an underestimation of the total effect of agricultural measures, especially for phosphorus. SENTWA predicts a very limited immediate impact of reducing nutrient application levels on phosphorus losses (for more strict nutrient legislation and manure processing, SENTWA even predicted no effect), but it can be expected that more gradually additional savings in the longer term can be achieved as this will gradually reduce soil P contents, which will reduce losses of P attached to soil particles and by leaching. This is confirmed by a recent review by Schoumans et al., 2014. Also, SENTWA is not able to predict the contribution of agriculture for COD. Including the new estimations for emissions and the effectiveness of measures might significantly shift the results of the cost-effectiveness analysis.

Combined sewer overflows (CSOs) are not considered due to a lack of information on the amount of emissions coming from these sources. A recent article by Langeveld et al. (2013) demonstrates the potential impact these overflows can have on surface water quality. Not considering CSOs will probably be less important when annual averages are considered but might become more important for peak emissions as concentration targets also focus on 90 percentile concentrations. Also for households not connected to a WWTP, monitoring data are not available. Emissions are derived from standard emission factors per inhabitant-equivalent and the amount of inhabitants located in a certain area. Parasite water entering the system and the impact of septic tanks at the individual household level are not included. Another data weakness is related to industry. The potential for additional end-of-pipe measures in industry might be overestimated as the list of implemented end-of-pipe technologies is not always based on recent data (for specific companies data on existing technologies is older than 2003). A last data weakness is that emissions from 2006 are considered to be static, whereas the target year lies in 2015. Autonomous development due to population and economic growth will probably lead to increasing emissions if no additional measures are taken.

An important weakness of the model set-up is that targets are defined as emission reduction targets whereas legal targets for good status are defined as concentrations (summer average, 90 percentile values). Measures which are cost-effective to reduce annual emissions, might be less cost-effective to reach 90 percentile concentration targets. Interaction with a water quality model is required to determine cost-effective programmes of measures. In this chapter the Pégase model is used to assess the impact of predefined scenarios on water quality. Results of the economic model are used as input for the composition of these scenarios. However, no feedback from the water quality model towards the economic model is performed to iteratively improve the composition of the scenarios.

Also important to notice is that the results in this chapter were used in preparation of the first river basin management plan published in 2009. Model inputs are gathered between 2006 and 2008. For some of the measures, the relevance today is limited as the economic and policy context drastically changed. Especially for agriculture, measures as reducing livestock (voluntary buy-out) and the exclusion of the Nitrates Directive are no longer relevant today. Also, existing market prices for manure processing and revenues per animal type can be very different from the prices used in this chapter (reference year 2006). This can have an important impact on the cost-effectiveness of measures. The price of manure processing for pigs and cattle has remained more or less the same, although the manure processing capacity has almost tripled in ten years time (VCM, 2013). However, the price of processing poultry manure has reduced significantly (VCM, oral communication 2014), as manure from poultry is an interesting product to mix with pig manure to improve processing efficiency and the processor is able to get sufficient funding from processing pig manure and selling the end product (fertilizer). As a result practically all poutry manure is being processed. The measure 'livestock reduction poultry' would therefore have little effect on water quality.

Comparing the model results for agriculture with the current level of implementation, gives an indication on the validity of the model results, though not necessarily the most cost-effective measures are also implemented in real life. Other reasons as stakeholder acceptance and a high administrative complexity might lead to the selection of other measures. The most effective measures for agriculture determined by the model were increasing dairy efficiency, winter cover crops and increasing feed efficiency for pigs. Though no exact numbers are available on the amount of winter cover crops applied, subsidies for winter cover crops were stopped in 2009 as this can be considered as a normal farming practice. This is an indication that this measure is being applied widely in the meantime. Dairy efficiency has also increased. A 6% increase in efficiency was for instance reported in ADLO (2009) between 2004 and 2008. However, this has not resulted in a decreasing amount of dairy cattle, but to increasing production levels (Van der Straeten et al., 2012). Consequently, this measure cannot be considered today as an environmental measure which reduces the amount of nutrient losses. It is difficult to determine whether feed efficiency for pigs has improved during the last decade. The growing amount of farms applying nutrient balance systems to report their annual nutrient production levels is an indication that this type of measure is also being implemented (VLM, 2013). Also the fact that trends in nutrient production levels do not follow the trends in the amount of animals is an indication that feed efficiency is increasing (VMM-MIRA, 2012). The major improvements in the amount of emissions coming from agriculture are however related to additional manure processing and manure export (VLM, 2013), which was considered not very cost-effective by the model. This has also allowed the amount of livestock (poultry, pigs) to increase and simultaneously reduce the amounts of animal manure applied on agricultural land. A system of tradable nutrient allocation rights (NARs) was also introduced to control manure use on farmland. NARs limit the use of nutrients per hectare of land and can allow transporting manure to another farm, which is more cost effective from a farmer's perspective compared to manure processing (Van der Straete, 2012). Transporting manures between water bodies is an option which is not considered in this chapter. For tuned fertilization, it is difficult to judge how frequently this measure is applied today as no general statistics are available. In this chapter, it is assumed that due to the implementation of this measure the amount of chemical fertilizer reduces. Statistics on the annual use of chemical fertilizer are available for Flanders (Departement Landbouw en Visserij, 2011). The use of chemical fertilizer (N) has remained more or less the same between 2006 and 2010, whereas the use of P-fertilizer has reduced by 45%. This is potentially an indication that tuned fertilization is implemented more frequently. However, the reduction of chemical fertilizer is not only due to nutrient legislation but is also caused by increasing market prices for chemical fertilizer. No clear conclusion can be drawn for this measure. In general, it can be concluded that regular updates are required to assess the cost-effectiveness of measures. Especially for the agricultural sector, both the costs and effects of measures can change drastically in a relatively short timespan.

3.8. Conclusion

The WFD requires member states to use cost-effectiveness for the RBMP (river basin management plans) and this chapter describes how the Environmental Costing Model was used to assist the Flemish administrations for the scientific underpinning of the selection of measures for the draft RBMP for the Flemish Region in Belgium. A cost-effective ranking of measures was the basis for compiling the program of cost-effective measures for the first river basin management plan. The model was specifically applied here to demonstrate the impact of spatial scales on cost-effectiveness. As can be expected, results demonstrate that cost-effectiveness depends heavily on the geographical scale of the assessment. Measures that are on average cost-effective on a regional scale are not necessarily cost-effective on a subbasin scale. Differences can be large though the same methodology for assessing costs and effects is applied across the different waterbodies and all waterbodies consist of similar lowland rivers. Optimization at the local scale to achieve similar emission reduction effects of programmes of measures defined at the regional scale can provide large cost savings.

This analysis clearly demonstrates that conventional measures as urban wastewater treatment and far going nutrient application abatement are not sufficient to reach good water status for the selected parameters. An important challenge for administrators and scientists is to develop and test new measures and innovative technologies to reach good status in highly urbanized and agricultural areas.

Acknowledgements

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CHAPTER 4. Coupling a hydrological water quality model and an economic optimization model to set up a cost-effective emission reduction scenario for nitrogen in the Grote Nete catchment

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Abstract

This chapter focuses on the integration of the economic optimization model and a hydrological model. This allows determining the most cost-effective set of reduction measures to reach an in-stream concentration target instead of an emission load reduction target. The framework is based on the coupling of two models: the hydrological water quality model SWAT and an economic optimization model (Environmental Costing Model, ECM). SWAT is used to determine the relationship between the modelled in-stream concentration at the river basin outlet and the associated emission reduction. The ECM is used to set up marginal abatement cost curves for nutrients and oxygen demanding substances. Results for nitrogen are presented for the Grote Nete river basin in Belgium for the year 2006.

Contrary to many other waterbodies in Flanders (chapter 3) results show that the good status for total nitrogen can be reached in this study area and that it is feasible to achieve good status in specific pilot basins. The most cost-effective measures are more productive dairy cattle, implementing basic measures as defined in the WFD, winter cover crops, improved efficiency of WWTP, enhanced fodder efficiency for pigs, further treatment of industrial wastewater and tuned fertilization.

This set-up allows dealing with smaller time scales and daily variations in hydrological conditions which is innovative compared to most other existing cost-effectiveness analyses. It makes it possible to optimize towards other types of objectives as summer average or 90 percentile, instead of yearly averages. Daily variations in hydrological flows and different impacts on point and diffuse source emissions clearly influence the cost-effectiveness of measures.

4.1. Introduction

The European Union Water Framework Directive (2000/60/EC), further abbreviated as WFD, requires member states, amongst others, to set up programs of cost-effective pollution abatement measures as part of the river basin management plans (RBMP). Consequently, in Europe, a shift is ongoing from classical methods such as ‘trial and error’ and ‘worst polluter first’ to an assessment of cost and impact of pollution abatement measures.

Yet, the assessment of the cost-effectiveness of emission reduction measures has been one of the bottlenecks in designing the RBMPs. Despite the simplicity of the concept of cost-effectiveness (e.g. explained in Brouwer and De Blois, 2008), the availability of European Guidance documents (WATECO, 2002 and Interwies et al., 2004) and numerous publications on cost-effectiveness analysis for surface water quality improvements (e.g. Schleich et al., 1997, Lise and Van Der Veeren, 2002, Arabi et al., 2006, Fröschl et al., 2008), the development of a cost-effective Programme of Measures for the RBMPs has not been straightforward. An important reason for this is the requirement for multi-scale and multi-disciplinary inputs from environmental scientists (effectiveness), economists (costs), engineers (technical details of measures) and river basin managers (targets and policy priorities). It becomes evident that this is a challenging task which needs support from appropriate information systems and modelling tools that are able to cope with the complexity of the water system and planning process (Hattermann and Kundzewicz, 2010). Despite their availability, modelling tools have only been used to a limited extent in river basins for the selection of cost-effective Programme of Measures in the first generation river basin management plans (European Commission, 2012).

In Europe, several tools and methodologies have been developed that can be used by water authorities for planning and managing water resources in an integrated way at the scale of a river basin. Many of them have been integrated in the European ‘Catchmod’ project cluster (Hattermann and Kundzewicz, 2010). Turpin et al. (2005) and Volk et al. (2008) linked SWAT to an economic model for European watersheds. Similar hydrologic-economic modelling with SWAT is published in the US e.g. by Attwood et al. (2000), Gassman et al. (2002, 2006), Osei et al. (2003), Qiu (2005) and Arabi et al. (2006).

This chapter presents a tool which is used for the development of the RBMP of the Scheldt river basin (Belgium). We present a generic framework which allows to determine the most cost-effective set of reduction measures to reach an in-stream concentration target. Concentration targets are legally defined for every individual parameter. The framework is based on a coupling of two models: the hydrological water quality model SWAT (Neitsch et al., 2005) and the Environmental Costing Model, abbreviated as ECM as described in chapter 3.

The methodology discussed is used to assess the combined impact of measures on both point and diffuse sources and includes measures across sectors covering industry, agriculture, waste water treatment plants (WWTP) and households. Both the economic and the hydrological model make use of the same emission databases and are built at the scale of a river basin. Especially for the economic model, this level of detail is different to most economic models which usually follow administrative boundaries as countries or regions (Brouwer and Hofkes, 2008). This means up- or downscaling algorithms are not required since both models are built at the same scale and measures are defined on an individual source level. The databases have furthermore been negotiated and accepted by the competent authority.

The tool is developed for nitrogen, phosphorus and oxygen demanding substances. For the purpose of presenting the methodology this paper focuses on nitrogen pollution in a part of the Scheldt river basin, namely the Grote Nete river basin, in Belgium for the year 2006 (reference year of the data). The Grote Nete basin is a basin for which large efforts were made in the past, but still problems remain to reach the quality standards.

4.2. Methodology

4.2.1. The study area

The watershed of the Grote Nete covers approximately 400 km^2 and is situated in Flanders, the Northern region of Belgium as shown in Figure 10. It is a typical lowland area with slopes of the river bed below 2%. The dominant soil type is sand, with patches of loamy alluvial sediments. Average precipitation ranges from ca. 740 to 800 mm/y. The river basin is used intensively, having a high population density (200 inhabitants/km^2), high livestock density (average values are 150 cows/km^2, 300 pigs/km^2 and 4000 chickens/km^2) and intensive industry. About 60% of the total area is used for agriculture, mainly dairy and fodder production (pasture and corn land uses). Although large investments are made in order to improve the surface water quality, environmental pressures remain high and originate from all sectors. In 2006, approximately 30% of the study areas inhabitants were not connected to a waste water treatment plant and discharge directly into surface water. These households contribute 23% of the nitrogen emission loads. The agricultural sector used in average 220 kgN/ha of fertilizer, of which 81% is animal manure and 19% artificial fertilizer and cause 35% of the nitrogen emission loads. Industry and waste water treatment plants contribute respectively 15% and 26% of the nitrogen emission load.

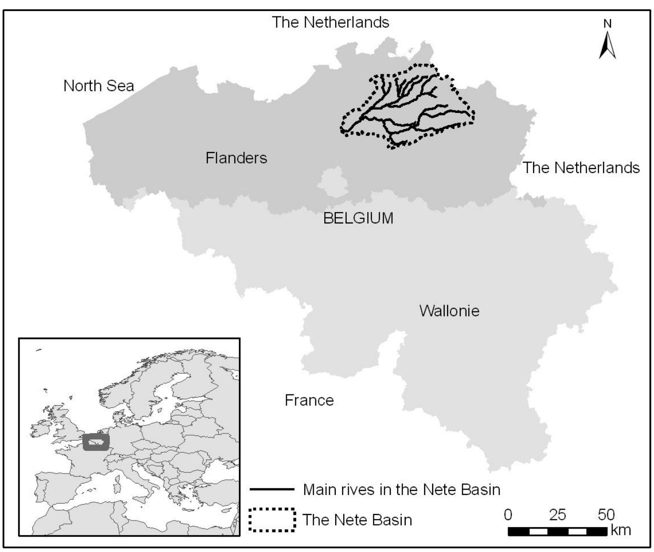

Figure 10: Location of the Grote Nete study area in Flanders, Belgium

4.2.2. The SWAT model

The Soil and Water Assessement Tool (SWAT) has gained international acceptance as a robust watershed modelling tool. Gassman et al. (2007) give an overview of the more than 50 peer-reviewed publications on SWAT for pollutant assessments, linked to a hydrological assessment. SWAT has proven to be effective to simulate the impact of point and non-point emission reduction measures. SWAT integrates both land phase and in-stream processes and is suited to simulate alternative land uses and best management practices (BMPs), such as fertilizer and manure application rates and timing, cover crops (perennial grasses), crop rotations, filter strips, conservation tillage, grassed waterways, and wetlands. In SWAT, point-source measures are implemented as a scenario with a reduced input load which is then routed through the system. The amount of load reduction needs to be quantified with external tools. The measures used in this work are described in section 2.4. Point source measures consist of emission load reductions from industrial and public waste water. The agricultural measures are considered to reduce non-point sources only.

The use of SWAT for impact assessment of measures on nitrogen is reported by Chaplot et al. (2004), Arabi et al. (2006), Bracmort et al. (2006), Gassman et al. (2006), Santhi et al. (2006), Tong and Naramngam (2007), Nendel (2009), Pandey et al. (2009), Sahu and Gu (2009) and Volk et al. (2009).

The presented model is set-up and calibrated for the period 2002-2006 for flow, nitrogen components, phosphorus, BOD and dissolved oxygen. Only the results for flow and nitrogen are presented in Figure 11.

In order to model the nitrogen load balance, data from different sources with different time steps and scale were collected. Point source data from industry and WWTP on discharges and emission loads are available for individual companies or stations on an annual basis. In-stream water quality measurements are available at monthly basis whereas the data on emission loads of unconnected households and the mass of fertilizer applied are available on annual basis and at the scale of the municipality. The latter two emission sources, which correspond to 50% of the total emission load for nitrogen, are converted to 14 sub-catchments and are entered into SWAT as constant daily values.

Firstly, the SWAT model is calibrated for flow. The Nash-Sutcliff Efficiency (NSE) reached is 0.72. A better calibration is not feasible and a systematic underestimation of summer flow is observed. Due to the overgrowth by weeds in summer, the water is backed up in a significant part of the studied catchment. Secondly, the nitrogen components are calibrated against the residual between the modelled and the observed average concentrations. This simple objective function was chosen, in view of the fact that only a limited amount of monthly in-stream water quality data was available. For total nitrogen, the average observed concentration was 4.5 mgN/l whereas the average modelled concentration was 4.6 mgN/l. For nitrate, a residual of zero was obtained: both the average modelled and observed concentration were 2.1 mgN/l. Monitoring data on both water quality and emissions are not sufficiently available to calibrate the model on a more detailed time basis. This makes the model less robust to predict 90-percentile concentrations. Predicted versus observed concentrations are represented on a daily basis in Figure 11. More details on the data of the SWAT model can be found in annex 3.

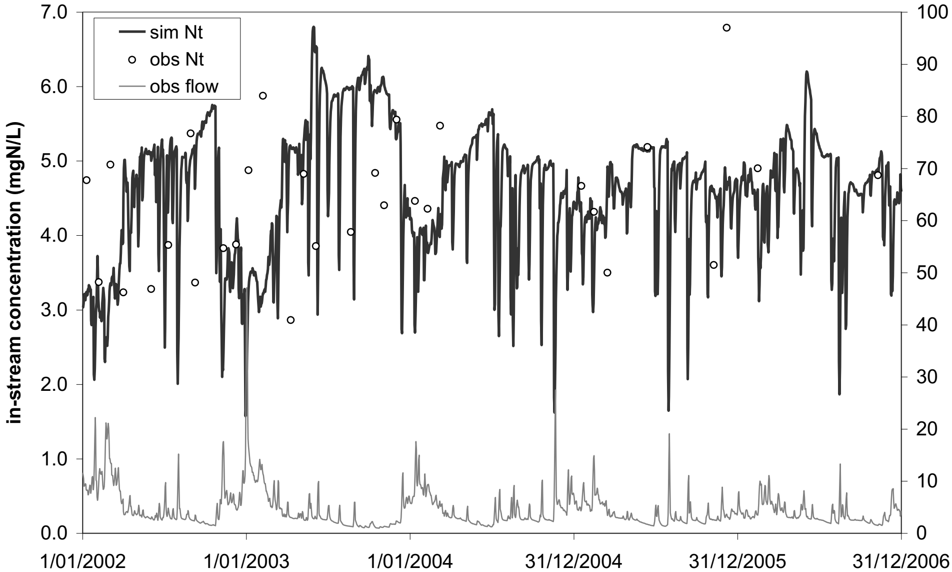

Figure 11: Output of SWAT for total nitrogen (mgN/l) and observations of total nitrogen and flow (m^3/s) at outflow point of model (HIC station 0731, VMM station 255000)

4.2.3. The Environmental Costing Model

The Environmental Costing Model or ECM already described and applied in chapter 3 is developed to assist policymakers in designing programs of cost-effective measures to meet the criteria for a good water status according to the WFD. The model, initially set up for industrial air pollution (Eyckmans et al., 2005; Lodewijks and Meynaerts, 2007), has been adapted to optimize the surface water quality management. Emission sources incorporated are industry, households and agriculture. Pollutants targeted are chemical oxygen demand (COD), total nitrogen (N) and total phosphorus (P).

The ECM, programmed in GAMS (Rosenthal, 2008), determines the least-cost combination of abatement measures by means of mixed integer programming. For a given pollutant, *p*, the ECM minimizes the objective function given by the following equation as also discussed more in detail in the previous chapter:

$$Min\left(C + t_p E_p\right) \qquad (1)$$

where C is the total cost of the pollution abatement measures in €/year; E_p the residual export emission load of pollutant p and t_p the (virtual) tax placed upon the residual export emission load.

More details on the economic optimization model are given in chapter 3.

4.2.4. Description of emission reduction measures

The emission reduction measures listed in this chapter are defined in the draft river basin management plan of the Scheldt river basin (CIW, 2008) and are considered relevant in the emission reduction of nutrients by policy makers and experts (industry, agriculture, waste water treatment). In the optimization algorithm, it is assumed that, when selected, a measure is implemented uniformly by all emission sources in the study area. A distinction between sources situated upstream and downstream is not made. Information about the measures, such as costs and reduction efficiency, is collected on an individual source level, but then summed for all sources in the basin. Although many authors, mainly in the US (Srivastava et al., 2002, Whittaker et al., 2003, Arabi et al., 2006) have proven that a uniform collective implementation is much less cost-effective than a spatially distributed optimization, a cost optimization on the individual source level is not applied because this is not requested by policy makers for the purpose of the RBMP, which was restricted to defining measures on a regional level. As the previous chapter demonstrated, local optimization could lead to significant cost savings. This is also why the development of more locally specific and diversified programmes of measures is one of the key targets for the second river basin management plan.

The ECM makes a distinction between basic and supplementary measures, as defined in the WFD. Basic measures are measures necessary to comply with existing European or national water legislation or measures that are already foreseen in ongoing policy, such as the Urban Waste Water Treatment Directive (91/271/EEC) and the Nitrates Directive (91/676/EEC). Supplementary measures are implemented in addition to the basic measures in order to achieve good water status. Basic measures are to be implemented anyhow and cannot be decided upon based on a cost-effectiveness analysis as is the case for supplementary measures. Yet, it is important to include the impact of basic measures in cost-effective optimization as their expected emission reductions will affect the reduction potential of supplementary measures and thereby their cost-effectiveness. The cost of basic measures is artificially set to zero to ensure that basic measures are selected in the first iteration steps. The implementation of the basic measures is the starting basis to evaluate the cost-effectiveness of supplementary measures. The measures included in the optimization are given in Table 7. The index letter is referred to in the marginal cost abatement function (Figure 15).

Table 7: Measures included in the optimization algorithm. A distinction between basic and supplementary measures is made according to the WFD definition. The index letter is referred to in the marginal cost abatement function

Source	Measure	Index	Basic	Sup.
WWTP	Construction or renovation of existing WWTP > 2000 IE to reach efficiency targets of European Urban Wastewater Directive	a	X	
	Construction or renovation of existing WWTP < 2000 IE to reach efficiency targets of European Urban Wastewater Directive	b		X
Households (waste water not treated)	Connection of existing sewers to new collectors (projects planned before or during 2006)	a	X	
	Connection of existing sewers to new collectors (projects planned after 2006)	c		X
	Extension of the sewerage network; divided in three groups according to the cost: 1) smaller than cost of individual treatment (low-cost sewage), 2) cost of sewerage < 2 x cost individual treatment (medium-cost sewage), 3) cost of sewerage > 2 x cost individual treatment (high-cost sewage)	d/e/f		X
	Individual waste water treatment for remote houses	g		X
Industry	Implement Best Available Technologies (BAT) and associated concentration targets	a	X	
	Implement standards of Urban Wastewater Directive for industrial waste water	h		X
Agriculture	Comply with existing nutrient legislation, including derogation of European Nitrates Directive	a	X	
	Increased dairy cattle productivity	i		X
	Winter cover crops	j		X
	Conservation tillage	k		X
	Buffer strips along watercourses	l		X
	Fertilization without excess (maximum up to crop requirements)	m		X
	Increased feed efficiency (pigs and poultry)	n/o		X
	More strict nutrient legislation (exclusion of Nitrates Directive derogation)	p		X
	Livestock reduction (poultry and other livestock)	q/r		X

The basic measures (measure 'a') include the construction or renovation of existing WWTP bigger than 2,000 Inhabitant Equivalents (IE), the connection of existing sewers to new waste water collectors, the implementation of Best Available Technologies (BAT) and associated concentration targets in industrial companies and compliance with the existing nutrient legislation (Nitrate Directive).

Supplementary measures have been defined across sectors, including improvements of waste water treatment plants (WWTPs) and agricultural measures. For waste water treatment plants, the renovation or construction of smaller WWTPs (<2,000IE) is defined (measure 'b'). Cost estimates are available for each individual station and are based on average renovation costs of similar WWTPs. The expected efficiency gains after renovating an existing WWTP are based on the legal targets (80% for stations with a capacity > 4,000 IE and 60% for stations with a capacity < 4,000 IE).

For households not connected to a WWTP, a distinction is made between households connected to a sewage system and households not connected to a sewage system. For the first group of households, the existing sewers are connected to new collectors (measure ‘c’). For the second group, the construction of new sewers is defined as a supplementary measure, based on the distance to existing sewage networks and thus costs to connect (measures ‘d/e/f’). For the most remote houses, the construction of a small scale individual treatment plant is assessed (measure ‘g’). Costs are assessed for each individual sewage project, based on the available investment plans for waste water collection and the amount of sewage required to connect households.

For individual industrial companies, the starting point for defining supplementary measures are concentration targets of the wastewater effluent. Based on differences between observed concentrations and targets, the required reduction potential is calculated for each company. A distinction is made between targets based on BAT (measure ‘a’) and more stringent concentration targets based on the targets for WWTPs in the Urban Wastewater Directive (measure ‘h’). Once the required reduction potential for each company is calculated, wastewater treatment technologies are selected to estimate the costs. Potential end-of-pipe technologies are selected depending on the observed concentration, the industrial sector and the technologies already implemented.

For agriculture a series of supplementary abatement measures are distinguished, i.e. measures aimed at reducing nutrient production by cattle, restricting nutrient application to crops and reducing nutrient loss from fields. For livestock reduction of poultry (measure ‘q’) and pig-cattle (measure ‘r’), the reduction over the period 2001-2004 is extrapolated. The yearly cost of production capacity lost is calculated using the Standard Gross Margin per animal. With measure ‘i’, the dairy cattle efficiency is increased through more efficient farming from the current mean of 7,156 kg/cow·year to 9,000 kg/cow·year. Although the latter results in an increasing excretion per cow, a lower total amount of cattle is needed to produce the same amount of milk. Thus, total excretion on river basin scale decreases. More productive dairy cattle results in an increased income (negative cost) of 0.71 €/100 l milk. The nutrient excretion of pigs and poultry is decreased by better tuning the protein content of pig or poultry feed (measure n/o) to their needs in during different growth phases. These feeds are more expensive, though, and the installations necessary for phased feeding require additional investments. For the restriction of nutrient application to crops, two measures have been defined: Tuned fertilization (measure ‘m’) and implementing a more strict fertilization limit (measure ‘p’). Tuned fertilization means that excess fertilization is avoided. Manure is only applied up to the crops’ N-requirements or to the legal limits and chemical fertilizer is only added if N-needs are larger than what can legally be supplied by manure. This means a reduction of mineral N with 37%. Professional advice based on soil analysis can help farmers to achieve this. However, advisory costs outweigh the reduced costs of chemical fertilizer. A further reduction in animal manure application fertilizer is proposed in measure ‘p’. Application rates from 250 kg N/ha·year on grassland and 200 kg N/ha·year on maize are reduced to 170 kg N/ha·year on all crops. Compared with the basic measure, this reduces the manure disposal area is reduced by 4% (Claeys et al., 2008) and manure export or processing needs to increase with 48% compared to the basic scenario.

A last group of agricultural measures aims to reduce nutrient losses from fields. Buffer strips along watercourses (measure ‘l’) are estimated to reduce particle runoff from fields by 51 to 94% (MESAM, 2007), but have little effect on nitrate abatement. Conservation tillage (measure ‘k’) reduces particle runoff by 42% (MESAM, 2007) to 93% (Gillijns et al., 2004). The costs of reduced tillage relate to the acquirement of the appropriate machinery and production losses which can go up to 60€/ha (Huybrechts, 2006). Finally, winter cover crops (measure ‘j’) reduce erosion and take up nutrients (especially nitrogen) that remain in the soil after the main crop is harvested. Losses can be reduced by 25 to 35 kg N/ha·year (den Boer et al., 2002). The costs are related to buying seed and cultivation. Cost savings are realized thanks to a decreased need for chemical fertilizer.

For more details on the measures and the cost and effect assessment, we refer to the previous chapter. More details on the dataset for the Grote Nete are available in annex 3.

4.2.5. Coupling of SWAT and ECM

The ECM as standalone does not allow assessing whether a specific load reduction achieves a water quality standard expressed as a concentration. For this purpose, an interaction with a surface water quality model, such as SWAT, is required. SWAT also simulates the export load and in-stream processes which are missing in the ECM. As shown in Figure 12, data is exchanged between separately running models. Firstly, the ECM calculates the required load reduction of measures and the corresponding costs to implement these measures. Secondly, SWAT is used to simulate resulting changes of the in-stream concentration due to load reductions. Based on multiple scenario runs, as described below, a relationship is set up between the total load reduction and the in-stream concentration. Based on the SWAT results described in Figure 13 and Figure 14, a first-order (linear) approximation of the relationship between load reductions and in-stream concentrations is estimated. This first-order approximation is then inserted into the ECM. This approach is valid as long as the modelled relationships between load reductions and concentrations are linear (though the SWAT model is non-linear) and targets are not pushing the model too far away from the starting point. Similar approaches are used for integrated assessment models for climate change (Nordhaus and Yang, 1996; Eyckmans and Tulkens, 2003).

As measures targeting point sources have a different sensitivity than measures targeting diffuse sources, their marginal costs cannot be compared directly. The latter however is a prerequisite for an integrated cost-effective ranking of both types of measures. In order to do so, the marginal costs of diffuse measures have been scaled to the level of point source measures by using the ratio of the sensitivity values. Hence, the optimization algorithm becomes equation 2:

$$Min\left(C + t_p \left[E_p(point) + \frac{\sigma(diffuse)_p}{\sigma(point)_p} E_p(diffuse) \right] \right) \quad (2)$$

where in comparison to Equation 1, the residual export emission load E_p is split into point and diffuse sources, σ is added as the sensitivity of the in-stream concentration, respectively for point sources and diffuse sources.

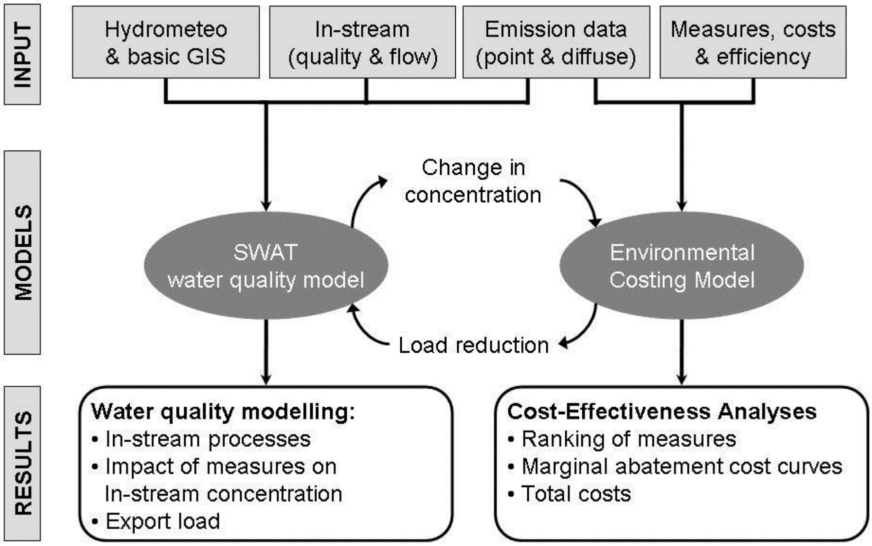

Figure 12: Overview of the input-output and coupling between SWAT and ECM

To set up the relationship between the load reduction and the in-stream concentration, three load reduction scenarios have been applied, as shown in Table 8: 1) a reduction of point sources only; 2) a reduction of fertilizer application only and 3) a combined and equal reduction of diffuse and point sources. In all three scenarios, the emission loads of the target source have been reduced with steps of 10% of nitrogen emissions.

Table 8: Scenarios applied for emission reductions

Scenario	Name	Point sources	Diffuse sources
Scenario 1	X% POINT RED	reduced with 10 steps of 10%	0% reduction
Scenario 2	X% FERT RED	0% reduction	reduced with 10 steps of 10%
Scenario 3	X% BOTH RED	reduced with 10 steps of 10%	reduced with 10 steps of 10%

4.3. Results & discussion

4.3.1. Impact of emission reduction on water quality

For the three scenarios, the relationship between in-stream concentration and emission load reduction at the outflow point of the study area is shown in Figure 13 for total nitrogen and in Figure 14 for nitrate. The two horizontal lines correspond to the water quality standards for good and very good status. In order to comply with the WFD, at least the category 'good' needs to be achieved. The standards fixed for the study area in Flanders for total nitrogen and nitrate are shown in Table 9 (CIW, 2008). Note that that the standard for nitrate (NO3-) is expressed as a 90 percentile whereas for total nitrogen (N), the standard is a summer half-annual average.

Table 9: Flemish standards for total nitrogen and nitrate (CIW, 2008)

Class	NO_3^- (mgN/L)	N (mgN/L)
Calculation method	90 percentile	summer half-annual average
Very good	≤ 2	≤ 3
Good	2 – 10	3 – 4

The required emission reduction percentages to achieve the water quality standards for total nitrogen and nitrate can be derived from Figure 13 and Figure 14 and are summarized in Table 10. Good status for total nitrogen can be achieved when: 1) point emission loads are reduced with 30%; 2) diffuse emissions are reduced with 70% or 3) both point and diffuse sources are reduced by 20%. To achieve the very good status, efforts need to be doubled. In that case, only reducing agricultural emissions will not be sufficient to reach the target. For nitrate, a good status is already obtained. The very good status can only be achieved when 50% of the agricultural emissions are cut or when 35% of both point source and diffuse emissions are reduced. The very good status cannot be reached by only reducing the point sources.

Table 10: Required emission reduction percentages to achieve the WFD standards. "NO" means the standard cannot be achieved

Scenario	**Name**	**Total Nitrogen (N)**		**Nitrate (NO_3^-)**	
		Good status	Very good status	Good status	Very good status
Scenario 1	X% POINT RED	30%	60%	0%	NO
Scenario 2	X% FERT RED	70%	NO	0%	50%
Scenario 3	X% BOTH RED	20%	40%	0%	35%

Based on the slopes of the relationships in Figure 13 and Figure 14, the sensitivity can be assessed (Table 11) with a linear regression. For total nitrogen, a reduction in point sources shows the largest sensitivity whereas a reduction in agricultural sources has the largest sensitivity for nitrate. This is explained by the large fractions of organic nitrogen and ammonia in the effluent from industry, WWTP and households. For agricultural emissions, the majority of the in-stream nitrogen loads originates from the nitrate dissolved in the base flow. Peak loads of nitrogen are mainly composed of organic nitrogen. Less manure application drastically reduces the nitrate loads in the base flow, especially in summer when the contribution of base flow to total flow is maximal.

Whereas reduction targets in percentages are useful for rough planning, the sensitivity in mass units is needed in order to set up the marginal abatement cost curves. Given that a 1% reduction of the export load of total nitrogen corresponds to 8.8 kgN for diffuse sources and 17.5 kgN for point sources, the following sensitivity values are found: -0.0018 mgN l^{-1} / kgN reduction of point sources and -0.0012 mgN l^{-1} / kgN reduction of fertilizer. The sensitivity for point sources is 50% higher than for diffuse sources. Yet, the sensitivity for diffuse sources is based on the export load. When compared to the applied fertilizer, the sensitivity of the in-stream concentration for total nitrogen to a kgN reduction is an order of magnitude lower as it needs to be multiplied by the export load coefficient. Modelling results in SWAT showed that the export load coefficients for each subbasin range between 4% and 17% with an average of 8.6%. The variability can be explained by differences in the distance to the outlet, the degree of excess manure and the availability of (natural) organic matter in addition to the applied nutrients.

Table 11: Sensitivity of in-stream concentration to an emission reduction for total nitrogen (in mgN l^{-1} / kgN reduction)

Scenario	Name	**Sensitivity of N** (in summer average mgN l^{-1} / kgN reduction)	**Sensitivity of NO_3^-** (in 90 percentile mgNO_3^- l^{-1} / kgN reduction)
Scenario 1	X% POINT RED	-0.0018	-0.0002
Scenario 2	X% FERT RED	-0.0012	-0.0018
Scenario 3	X% BOTH RED	-0.0017	-0.0011

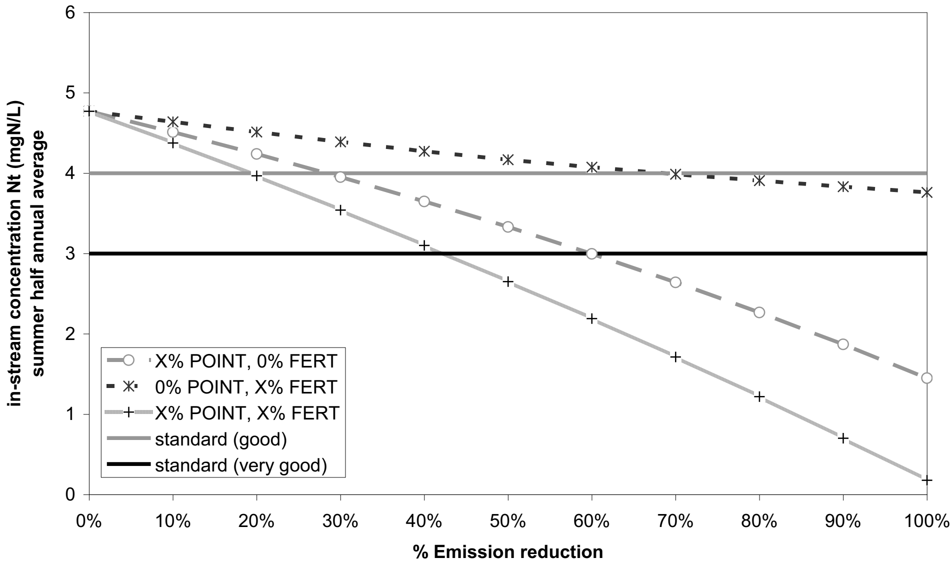

Figure 13: Relationship between in-stream concentration and emission reduction for total nitrogen (summer half-annual averages). The horizontal lines indicate the WFD standards: 4 mgN/l for good status and 3 mgN/L for very good status

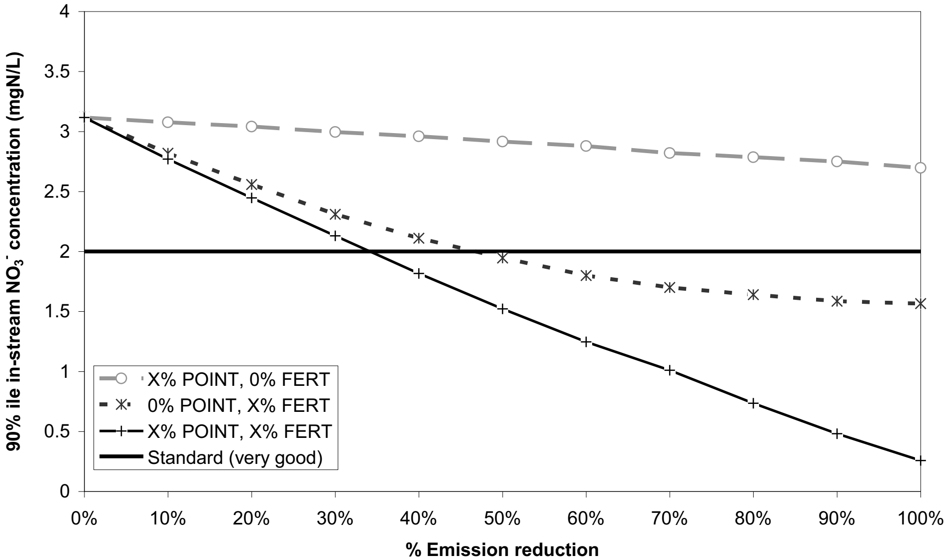

Figure 14: Relationship between in-stream concentration and emission reduction for nitrate (90%ile). The horizontal line indicates the WFD standards for very good status (2 mgN/l). The good status is already reached.

Although the nitrogen pathways and processes considered in SWAT are non-linear, model results shows that the relationship between in-stream concentrations to reductions in point and diffuse sources can be approximated as linear for summer half-annual averages of total nitrogen and 90%ile values of nitrate (up to 30% emission reduction). For nitrate, a saturation effect is observed for diffuse sources at about an emission reduction of more than 30%. The latter is considered to be the new short term equilibrium. Soil – groundwater exchange processes for nitrogen however remain to exist. The full benefits of reduced fertilizer application are only expected on the longer term (10-20 years).

For the study area, the linear relationship is valid for impact assessment of nitrogen abatement in the modelled range of concentrations. Linearization of the prevailing non-linear processes is acceptable given the specific conditions of the study area. Firstly, in-stream conversions of nitrogen components are small as the travel time is less than one day. Secondly, as the discharge is dominated by base flow, the majority of the diffuse export load is dissolved as nitrate into the groundwater. Groundwater processes in SWAT can be considered as being linear due to the semi-lumped approach. Similar results using SWAT or variants are obtained by Chaplot et al. (2004) and Jha et al. (2007) for the intensively manured lowlands of Iowa (US).

4.3.2. Marginal abatement cost curves

The coupled SWAT-ECM model provides the marginal costs of measures. Potential supplementary measures are ranked in order of decreasing cost-effectiveness and consequently plotted as shown in Figure 15 for total nitrogen in function of the associated in-stream concentration into stepwise marginal abatement cost curves integrating both point and diffuse sources and measures across sectors. Hereby, it is assumed that decision-makers will take the most cost-effective measure first and will only invest in additional measures if the required target is not met. The latter explains the stepwise shape of the abatement cost curve. The height of each step corresponds to the marginal cost of an additional reduction measure. The length of a step corresponds to the concentration reduction capacity. The vertical grey line indicates the in-stream average concentration target. The letters refer to the measures listed in Table 7.

Good status (4 mgN/l) for total nitrogen can be reached in the Grote Nete catchment after implementing the following measures: more productive dairy cattle (measure “i” in Table 7), implementing basic measures as defined in the WFD (a), winter cover crops (j), improved WWTP efficiency (b), enhanced fodder efficiency for pigs (n), further treatment of industrial wastewater (h) and tuned fertilization (m). The good status for total nitrogen can be reached at a marginal cost of 53 Euro/kgN removed. The very good status (3 mgN/l) cannot be reached even if all remaining, less cost-effective measures are selected. The cumulative emission reduction of all measures included in the assessment corresponds to an emission reduction of total nitrogen of 38% spread over diffuse and point sources.

The less cost-effective measures are lowering the maximal rates for manure application to the level of the EU Nitrates Directive, including the processing of excess manure (p), reducing the amount of poultry (q), extending the local sewage networks grouped into cheap (d), moderate (e) and expensive (f), extending regional sanitation infrastructure (c), reducing cattle and pigs (r), increasing fodder efficiency for other livestock (o) and individual treatment for household waste water (g). It was assessed that implementing buffer strips along watercourses (l) and reduced tillage (k) would have no additional impact on total nitrogen. These measures are not presented in Figure 15.

As reducing point sources is impacting more the concentration of total nitrogen during the summer compared to reducing diffuse sources, the cost-effectiveness of point source measures as connecting households to collective treatment, increasing the WWTP efficiency and treatment of industrial wastewater is positively influenced compared to the previous chapter.

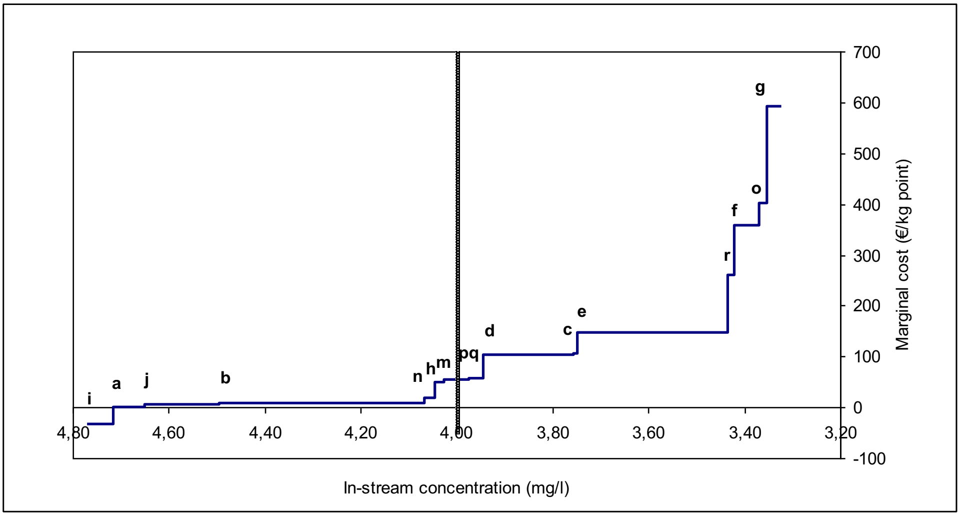

Figure 15: Marginal cost abatement function for total nitrogen. The vertical gray line indicates the 'good status' standard. The letters refer to measures targeting both point and diffuse sources as listed in Table 7

The presented results are based on average estimates for both costs and effects. The ranking of measures might change when minimum or maximum estimates are applied. Although we have not performed an uncertainty analysis on costs and effects, we consider that, the difference in cost-effectiveness between the most cost-effective measures (i, a, j, b) and the other measures is so large that a potential change in ranking among the most effective measures does not alter the selection of these measures. This is also confirmed in the next paragraph where the sensitivity of discount rate and lifespan is tested. The same conclusion is valid for the least cost-effective measures (c, e, r, f, o, g). Even at the extreme case when minimum cost estimates and maximum effectiveness estimates are applied, these measures will not be selected as cost-effective measures. A third group of measures, the moderately cost-effective measures (n, h, m, p, q, d), however have a cost-effectiveness that is more or less equal and some of these measures are required to reach the objective. Based on the cost-effectiveness analysis alone and the uncertainty related to costs and effects, we cannot conclude which of these measures need to be selected to reach the objectives at the lowest cost achievable. Besides cost-effectiveness other criteria as the efforts and capacity needed to get and keep a measure going and stakeholder acceptance certainly play a role when choosing between these measures.

Though the results in Figure 15 demonstrate the possibilities to set up marginal abatement cost curves for instream concentration targets, the ranking of measures is hardly influenced by the use of a water quality model, as the sensitivity of diffuse source emission reductions and point source emission reductions on summer average N concentrations is not sufficiently different to compensate for large differences in costs and effects of agricultural measures versus other measures. This is not the case for 90 percentile NO_3^- concentrations, where concentrations are much more sensitive for reducing diffuse sources compared to point sources. The impact on the ranking of measures in the cost-effectiveness analysis is represented in Figure 16. Measures reducing agricultural losses have much more impact on 90 percentile NO_3^- concentrations compared to point sources. Consequently, they are becoming much more cost-effective for this parameter compared to summer average concentrations for total nitrogen. The target for reaching the very good status for NO_3^- (2 mg/l) cannot be reached as the maximum amount of emission reductions that can be achieved with the measures considered for this case study is 32%, which is less compared to the required 35% if both point and diffuse sources are reduced in the same degree and the required 50% if only diffuse sources are reduced (Table 10).

More details on the results are given in annex 3.

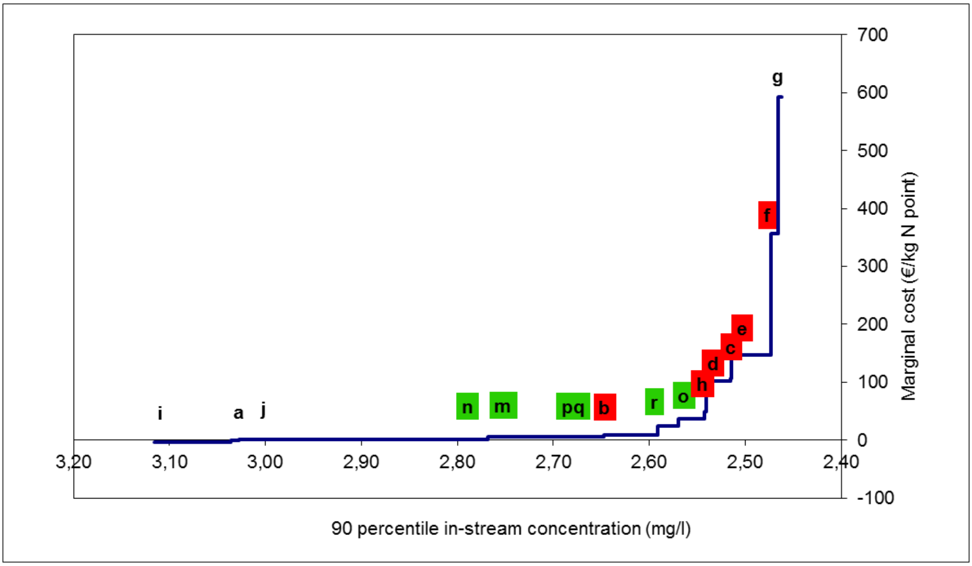

Figure 16: Marginal cost abatement function for nitrate (90 percentile concentrations). The 'very good status' standard (2mg/l) cannot be reached with the measures listed in Table 7. Measures labeled in green (red) are ranked better (worse) compared to total nitrogen.

4.3.3. Sensitivity of results for discount rate and lifespan

The sensitivity of the results on assumptions made for the discount rate is represented in Figure 17. Varying the discount rate has a large impact on measures with a longer lifespan (sewage, wastewater treatment plants) and no influence of measures for which the annual loss of income is used as a cost estimate (agriculture). A higher (lower) discount rate leads to higher (lower) annual equivalent costs. The same ranking of measures is used to represent the graph. The fact that the red curve, representing a discount rate of 7%, is no longer increasing stepwise for measures f and o represents that o is more cost-effective compared to f when a higher discount rate is assumed. However, this is the only shift in ranking of measures due to a change in discount rate and is less relevant as both measures are not cost-effective in this case study. The difference between measure d (low cost sewage) and other measures close to the target line is becoming smaller when a low discount rate is assumed. If in combination with a low discount rate, extremely long lifespans are assumed for sewage projects (100 years), this difference becomes even smaller, but measure d is still less cost-effective compared to the other measures.

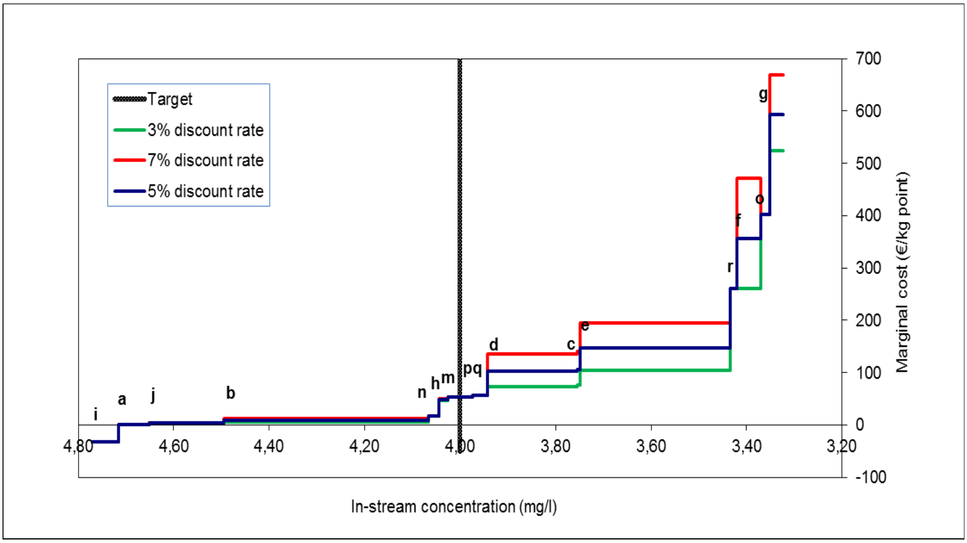

Figure 17: Marginal cost abatement function for total nitrogen and different discount rates. The vertical gray line indicates the 'good status' standard. The letters refer to measures targeting both point and diffuse sources as listed in Table 7.

4.4. Discussion

This chapter describes a possible model set-up to deal with more detailed with smaller time scales and daily variations in hydrological conditions in a cost-effectiveness analysis. The results demonstrate the differences in cost-effectiveness when peak concentrations are targeted compared to annual or seasonal averages. To be able to deal with sufficiently detailed scales in time to assess whether 90 percentile targets are reached input data (emissions, meteo data) and calibration data (water quantity, quality) are ideally available on a daily basis. Though this is the case for meteorological and flow data, this is not the case for emission data (annual averages) and water quality data (12 observations/year). This makes the calibration of the water quality model less straightforward and the model less trustworthy, especially for 90 percentile concentration levels. Also, the lack of data to assess the emissions coming from combined sewer overflows can significantly influence the simulation of peak concentration levels as was demonstrated in Langeveld et al. (2013).

The assessment of the impact of measures as winter cover crops was also largely simplified. The expected impact on emissions entering the river system is estimated on a yearly average basis with SENTWA (chapter 3) and independently of SWAT. Consequently, differences in dynamic impacts on peak concentrations between reducing manure disposal and installing winter cover crops are not taken into account. It can be expected that the relative impact of winter cover crops during peak concentrations is higher compared to reducing manure disposal. How the impact of management options in agriculture can be simulated with SWAT was for instance demonstrated for bufferstrips in Barlund et al. (2007).

A straightforward integration between the hydrological model and the economic model was applied to assess the cost-effectiveness to reach in-stream concentration levels. More sophisticated optimization procedures whereby hydrological models are simulated many times to derive pareto optimal cost-effect combinations of measures are for example performed in Arabi et al. (2006) and Panagopoulos et al. (2012), but also in these examples CPU time is considered an issue and SWAT is not modelled for every possible combination of measures. If thousands of runs are needed with traditional hydrological models to find the optimal solution for larger basins and multiple parameters simultaneously, this will be very demanding. Parallel computing with computer clusters will only solve the problem partially. Another solution lies in the reduction of the model complexity (and consequently CPU time) by using "appropriate" model structures performing adequately well for the objective of the model study, keeping CPU time to a minimum. This is also previously discussed in chapter 2.

4.5. Summary and conclusions

A hydro-economic modelling framework is presented to set up a cost-effective program of measures to achieve an in-stream concentration target. It consists of a modular coupling between the hydrological water quality model SWAT and the economic optimization model ECM. As in most hydro-economic modelling work (Harou et al., 2009), the hydrological processes have been simplified. A semi-linear relationship has been setup, after a series of simulations in SWAT, between point and diffuse emission load reductions and 90 percentile water quality concentrations. This relation is then integrated in the ECM to determine the measures required to achieve water quality targets by means of a marginal abatement cost curve. The latter is considered to be a valid first assessment 1) to quantify the required emission reduction to reach an in-stream concentration target and 2) to compare the cost-effectiveness of measures across sectors and processes on the scale of a river basin.

Results show that the good status for total nitrogen can be reached in the study area. The most cost-effective measures are more productive dairy cattle, implementing basic measures as defined in the WFD, winter cover crops, improved efficiency of WWTP, enhanced fodder efficiency for pigs, further treatment of industrial wastewater and tuned fertilization. An approach aiming at an emission reduction from all sectors is the most cost-effective program of measures to improve the in-stream water quality. The biggest reduction of total nitrogen and more specifically reducing the summer average concentration can be obtained through a reduction of point sources. However, when focusing on nitrate, relevant e.g. for the Nitrate Directive or Groundwater Directive and its 90 percentile concentration target, reducing agricultural sources has the biggest impact. This result indicates to carefully consider which type of parameter and which type of concentration targets are to be reached when selecting a cost-effective set of measures. The existing legal concentration targets for Flanders include a much larger range of water quality parameters with different types of concentration targets. Calculations here are focused on N and NO_3^- . COD and P are also available for the economic model, but are not considered here. COD was not modelled with SWAT. Results for P were available from the SWAT model, but proved to be less reliable for the Grote Nete basin.

Acknowledgements

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CHAPTER 5. A risk based approach to establish a cost-effective flood risk management plan for the Scheldt estuary

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Abstract

The Scheldt is a tidal river that originates in France and flows through Belgium and the Netherlands. The tides create significant flood risks in both the Flemish region in Belgium and the Netherlands. Due to sea level rise and economic development, flood risks will increase during this century. This is the main reason for the Flemish government to update its flood risk management plan. For this purpose, the Flemish government requested a cost-benefit analysis of flood protection measures, considering long-term developments. Measures evaluated include a storm surge barrier, dyke heightening and additional floodplains with or without the development of wetlands. Some of these measures affect the flood risk in both countries. As policies concerning the limitation of flood risk differ significantly between the Netherlands and Flanders, distinctive methodologies were used to estimate the impacts of measures on flood risk. A risk based approach was applied for Flanders by calculating the impacts of flood damage at different levels of recurrence, for the base year (2000) and in case of a sea level rise of 60 cm by 2100. Policy within the Netherlands stipulates a required minimal protection level along the Scheldt against storms with a recurrence period of 1 in 4000 years. It was estimated how flood protection measures would delay further dyke heightening, which is foreseen as protection levels are presently decreasing due to rising sea levels. Impacts of measures (safety benefits) consist of delays in further dyke heightening. The results illustrate the importance of sea level rise.

Flood risks increased 5-fold when a sea level rise of 60 cm was applied. Although more drastic measures such as a storm surge barrier near Antwerp offer more protection for very extreme storms, a cost-optimal combination of dykes and floodplains can offer higher benefits at lower costs.

Unlike the previous chapters where cost effective programmes of measures are composed to reach pre-defined environmental targets, this chapter applies a risk-based approach. This means that the amount of measures to be implemented depends on the benefits they can achieve. Scenario development is performed in a stepwise approach by combining flood risk simulation models and economic analysis. Spatial interdependencies are dealt with by stepwise scenario development where the study area is subdivided in 5 subzones and the optimal scenario in subzone 1 is the starting point for subzone 2 and etc. This stepwise scenario building, in combination with the long time frames taking into account the impact of sea level rise and the valuation of ecosystem services provides an innovative and powerful decision support framework.

5.1. Introduction

The Scheldt river originates from France, crosses Belgium and the Netherlands and ends up in the North Sea (Figure 18). The river has a tidal influence reaching up to 155 km inland, covering an entire gradient from salt over brackish to fresh water areas (Cox et al., 2006). The tidal waves result in a flood risk in the Northern part of Belgium (Flanders region) and the Netherlands. Important flood damages occurred in the Netherlands in 1953 and in Belgium in 1976. In the Netherlands, no damages occurred in 1976 because the Delta-plan was almost completely finished. This gave rise to an accrued public awareness of the inundation risk along the tidal reach of the Scheldt in Flanders, and to the conception of a Flemish so-called Sigmaplan in the beginning of the 1980's. This Sigmaplan was composed of a tidal storm surge barrier downstream Antwerp, combined with a general heightening of the river-embankments and the construction of a number of controlled flood areas. A socio-economic analysis (Berlamont et al., 1982) showed that a storm surge barrier could not be economically justified and as a result the barrier was never constructed. However, due to sea level rise and economic developments it is generally believed that flood risks will increase significantly during the 21st century. This is the main reason why the Flemish government required an update of its flood risk management plan.

The Flemish government wanted to reconsider the necessity of the Sigmaplan while considering several issues. Firstly, besides a "fixed safety standard" approach also a risk based approach had to be applied. The objective of this approach was not to avoid all floods but to limit flood damages at reasonable costs. Densely populated areas or areas with important industrial installations, where most flood damages might occur, have to be protected the most. On the contrary, agricultural and nature conservation areas where less damage is expected, require less protection. There is a general belief that protecting the whole Scheldt river basin in the same degree, as is the case in the Dutch part of the Scheldt, would lead to disproportionate costs. Secondly, as mentioned, the impact of sea level rise had to be considered. As Berlamont et al. (1982) did not consider the longterm impacts of sea level rise and gradually increasing probabilities of extreme flood events, it was expected that the existing measures are insufficient. Thirdly, the effectiveness of floodplain restoration had to be examined considering the potential non-market benefits. It was expected that these benefits were an important distinction between floodplains and more technical approaches such as dyke heightening and storm surge barriers. Fourth, potential positive or negative impacts of flood protection measures in the Netherlands had to be included. A storm surge barrier, for instance, could have negative impacts in the Netherlands as no water can be stored further upstream when the barrier is closed during extreme events. The results of this study are also a step towards corresponding to the requirements of the EU Floods Directive (European Union, 2007) The Directive requires Member States to assess if all water courses and coast lines are at risk from flooding, to map the flood extent and assets and humans at risk in these areas and to take adequate and coordinated measures to reduce this flood risk.

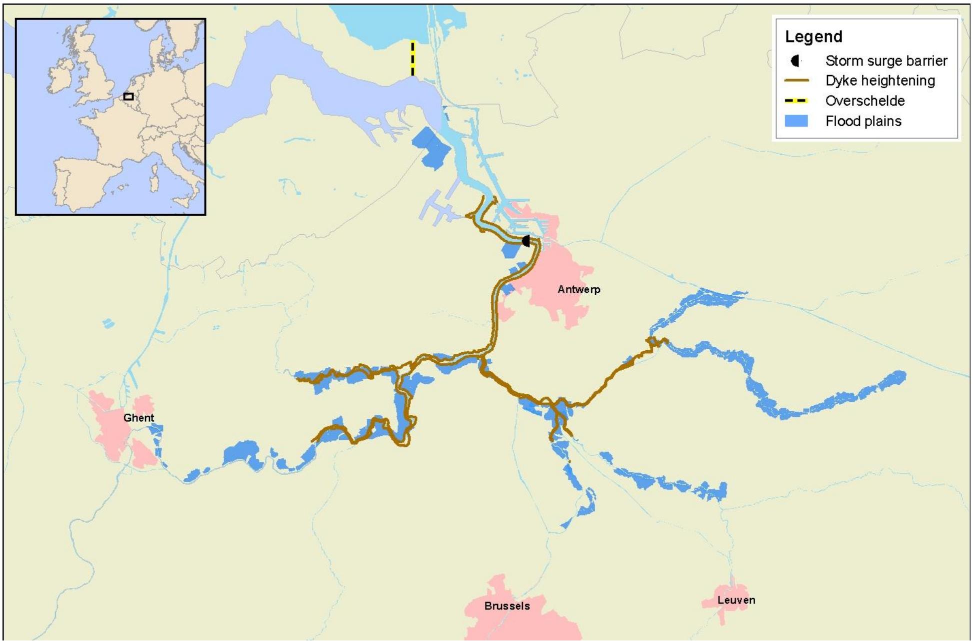

Figure 18: Location of potential flood protection measures in the Scheldt estuary

The use of a cost-benefit analysis framework to select flood protection measures has been applied frequently in the past within the study area. To support the decision making processes in the implementation of the Dutch Deltaplan for instance, the socio-economic consequences were considered in Tinbergen (1959) and van Dantzig (1959). A cost-benefit analysis was also used to assess the original Sigmaplan (Berlamont et al., 1982). Since then, the available methodologies and tools to assess flood-related costs and benefits have evolved drastically. More recent efforts as in Brouwer et al. (2004) and Turner et al. (2007) make use of hydrological models and land use data to estimate flood damage, ecological models and economic valuation tools to assess non-market impacts. Turner et al. (2007) also consider sea level rise by assuming unsatisfactory defences have to be replaced at regular time period. A detailed simulation of the impact of sea level rise on flood frequencies and intensities and resulting damages as presented here rarely occurs.

As stated, both a “fixed safety standard” approach and a “risk optimization” approach were applied. The “fixed safety standard” approach, applied during a first phase, starts from a fixed protection level against flooding for the whole study area. The composed scenarios could theoretically offer safety against inundations caused by storm tides for return periods of 10,000, 4,000, 2,500 and 1,000 years in 2050, though not for all technical solutions all safety levels were studied (or even achievable). Since uniformly high safety levels throughout the basin were not achievable by means of storm barriers or flood control areas alone, the scenarios using those components were always supplemented by heightening of the dykes at the most vulnerable locations. The “risk optimization” approach, applied during a second phase, targets to find an economically optimal combination of dyke heightening and flood control areas in the basin. Considering the fact that flood risk was generated by downstream storm tides, eventually combined with high run-off discharges originating from the upstream tributaries, the basin was subdivided in five zones, each of them centered around “damage centers” (Figure 19). In a first step, the best combination of measures in downstream zone 1 was searched, by trial and error. Based on the results from the “fixed safety standard” scenarios, it could be roughly determined how flood risks evolve if additional dykes or specific floodplains are implemented in a certain area. These results were used to compose interesting combinations of both dyke heightening and flood control areas as a starting point for the composition of alternative scenarios in a specific zone. Based on a comparison of the results of the composed scenarios and a detailed calculation of the remaining flood risks (on a municipality level), it could be determined how much potential benefits still can be realized and hence where and to what extent additional measures can still achieve net benefits. The set of measures in zone 1 having the best result (costs vs. benefits) were kept for the next step. In a second step, the best solution for zone 1 was combined with various sets of possible protection measures in zone 2. Incremental costs and incremental benefits for the whole basin, compared to the cost and benefit realized with measures only in zone 1, were estimated. This resulted in a “best solution” for zone 1 and 2 together. This procedure was successively applied for zone 1, best of zone 1 + zone 2, best of zone (1+2) + zone 3, best of zone (1+2+3) + zone 4, etc. This approach made it possible to determine an optimized risk based scenario with a limited amount of model simulations (+/- 20).

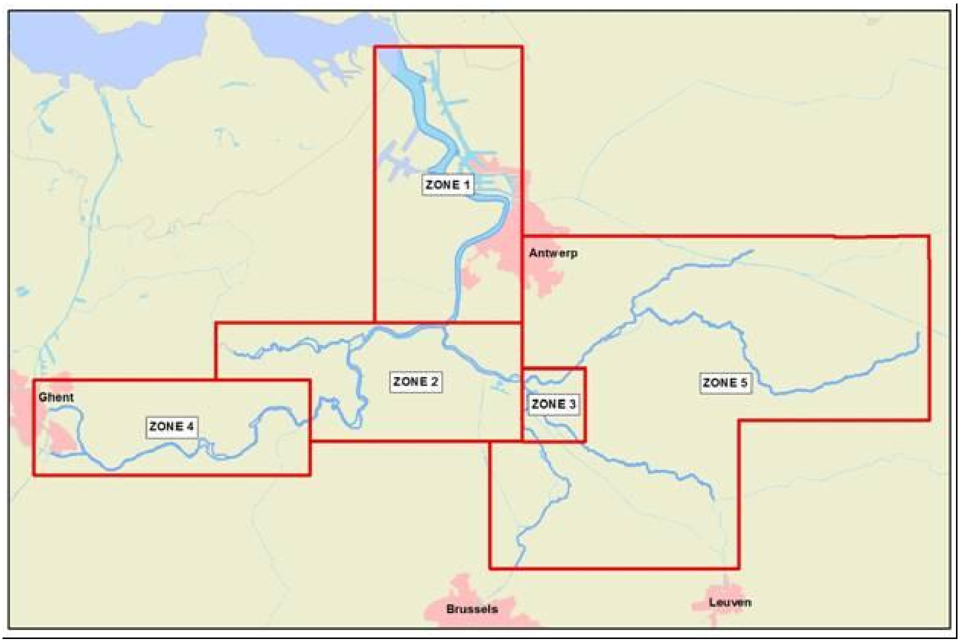

Figure 19: Subdivision zones for bottom up approach

5.2. Methodology

5.2.1. Possible measures

Depending on the predefined protection levels alternative scenarios based on different combinations of technical measures were set up.

An overview of all possible measures is given in Figure 18. Considered measures are a storm surge barrier, construction of the Overschelde, flood control areas and dyke heightening:

- The design of a possible storm surge barrier downstream from Antwerp is based on the existing Maeslant barrier near Rotterdam. The barrier consists of two giant, hollow, semi-circular doors that can be closed in case of an anticipated high water level. The option of installing a much smaller barrier further upstream (Niel) was also examined. Contrary to the barrier in Antwerp, it only protects a part of the estuary.
- The Overschelde is a canal that connects the Western Scheldt with the Eastern Scheldt. The Eastern Scheldt is another tributary of the Scheldt already protected by a barrier. In case of high tides the Overschelde can be used to store storm water from the Western Scheldt on the Eastern Scheldt.
- Flood control areas (FCA) are areas enclosed by a higher outer dyke and a lower inner dyke along the river. When during a storm surge the water level rises above the inner dyke, large amounts of water can temporally be stored in these areas. Usually the frequencies of flooding for flood control areas vary between once every ten years to twice a year. This means agricultural activities might still be possible.
- An alternative set-up is a flood control area with controlled reduced tide (CRT). In case of a CRT, the tidal regime is introduced. During normal tidal cycles water flows in and out the area through well designed culverts. This also means the area is used as a nature conservation area (Cox et al., 2006).
- Dyke heightening is the more classical way of increasing flood protection. Depending on the local circumstances alternative designs are foreseen. For most of the cases, a standard design extending existing dykes on both land and river sides, is possible. In case of space limitations, which is usually the case in urban areas, alternative designs are the construction of a small wall on top of existing dykes and a sheetpile wall. The quay walls in the city centre of Antwerp, need to be reconstructed if increased protection levels are required.

5.2.2. Cost-benefit analysis

A cost-benefit analysis (CBA) was used to compare the economic efficiency of alternative combinations of measures. In CBA, costs and benefits of a project are compared over a fixed time horizon and subject to a discounting procedure. It is used to determine whether flood protection projects will achieve net gains in economic welfare for the society as a whole (Andrews et al., 2006).

The costs included in the analysis are the investment and maintenance costs of measures and the opportunity costs from the loss of economic value of the former land-use in flood control areas. Flood protection benefits are estimated both for Flanders and the Netherlands. Other impacts are the expected ecosystem benefits of floodplains, economic damage for shipping during construction and testing of the large storm surge barrier near Antwerp and visual intrusion due to the construction of new dykes near houses.

Starting date of construction was assumed to be 2010. The construction period depended strongly on the measure, varying between 10 years in case of the storm surge barrier and 4 years in case of construction of floodplains. Safety benefits are achieved after completion of the measures.

The parameters that were used to evaluate alternative measures were net present value and the discounted payback period. Both statistics were calculated, as the lifespan of these kinds of projects is difficult to determine. A fixed discount rate of 4% was applied. Economic growth was based on scenarios developed in CPB (1996) and further updated in Saitua (2004). The standard growth rate was estimated at 2.4% until 2020 and 1.8% after 2020. Costs and benefits were estimated until the year 2100.

5.2.3. Investment and maintenance costs of measures

Costs included the investment costs, maintenance and operation costs of flood protection measures and necessary expropriation costs for houses, industry and agriculture. An overview of the average costs per category is shown in Table 12. The costs of dyke heightening and flood control areas were estimated for each area specifically. Depending on the size and structure of the existing dykes, the required heightening and the available space available, the construction techniques, the required materials and hence the costs differ significantly. The numbers in the tables are average estimates.

5.2.4. Opportunity costs for agriculture

The cost estimates of creating additional flood control areas on existing agricultural area was based on the opportunity costs or the cost of lost earnings from current and future agricultural activities (Dierckx, 2004). Agriculture could be maintained within a flood control area. However, it was expected that high value crops such as vegetables, sugar beets and orchards will be moved into other areas and replaced by low value crops as corn or pasture for livestock. In this case adaptation costs for relocating these high value crops to other areas were included. The relocation costs include costs for replanting, soil improvement, drainage and sprinklers and were assumed to be 10,000 €/ha for sugar beet, potato, vegetables, orchards and tree nurseries. Additionally, a 10% loss of production is assumed for all crops during the first 10 years after relocation. Other costs are the consequences of flood events inside the flood control area. This comprises loss of crops (100% loss in case of floods), administrative costs (250 €/ha) and clean-up costs (150 €/ha).

When agriculture cannot be maintained within the area (reduced tidal area), the relocation costs for high value crops are still considered. Other costs are the loss of added value of low value crops (288 €/ha), a loss of manure deposition capacity (270 €/ha) and the loss of labour (10years of unemployment is assumed at 924 €/ha). The valuation of the loss of manure capacity is based on a tax of 0.99 €/kg N and 0.99 €/kg P2O5 (superheffing), in combination with average existing application levels of 170 kg N/ha and 100 kg P2O5/ha (Dierckx, J., 2004). These estimations are different from the cost estimations for manure processing in chapter 3. The total cost per ha of crop situated inside flood control areas is given in Table 13.

Table 12: Average investment and maintenance costs measures (Resource Analysis, 2004)

Item	Cost (€ 2002)
Storm surge barrier Antwerp	560 mln €
Overschelde	1,570 mln €
Storm surge barrier Niel	55 mln €
Dyke heightening	
- standard	300 – 2,000 €/m
- wall on top	800 – 2,500 €/m
- sheet pile wall	3,500 – 5,000 €/m
- quay wall (Antwerp)	16,100 €/m
Flood control area	
- inner dyke adaptation	770 €/m
- outer dyke construction	840 €/m
- outlet sluices	19,000 €/ha
- inlet sluices CRT	4,000 €/ha
Engineering cost	10% investment cost
Other cost	5% investment cost
Annual maintenance cost	0.5%-1.5% investment cost
Dispropriation cost grounds	
Residential area	700,000 €/ha
Industrial area	24,046 €/ha
Recreational area	12,200 €/ha
Dispropriation cost buildings	
Houses	100,000 €/building
Farms	250,000 €/building
Companies	250,000 €/building
Destruction cost	30,000 €/building

Table 13: Agricultural losses for crops situated inside flood control areas (4% discount rate, 2100) for different frequencies of flooding

Land use	Yearly	4-yearly	10-yearly	Daily (RTA)
Pasture	€ 16,920	€ 7,012	€ 3,255	€ 21,689
Grassland	€ 17,918	€ 9,759	€ 4,629	€ 22,688
Maize	€ 17,959	€ 10,057	€ 4,778	€ 22,729
Cereals	€ 29,142	€ 21,981	€ 9,274	€ 33,912
Sugar beet	€ 29,145	€ 21,984	€ 9,286	€ 33,915
Potato	€ 39,311	€ 32,150	€ 46,191	€ 44,081
Vegetables	€ 36,428	€ 29,268	€ 24,137	€ 41,198
Orchards	€ 65,078	€ 57,918	€ 52,787	€ 69,848

5.2.5. Benefits of reducing flood risks

5.2.5.1. Hydrological modelling

A hydrodynamic branched 1 dimensional model was created of the Scheldt river and most important tributaries using the MIKE11 software package from DHI. The model started at the mouth of the Scheldt in the North Sea and covered the entire tidal reach of the river (in total about 362 km of river). The floodplains along the rivers were included as separate branches, which are dry under normal circumstances. The dykes between the river and the floodplains were included in the model as so called link-channel units. Overtopping of dykes occurs when the simulated water level exceeds the topography of the dyke.

The 1D model uses several boundary conditions: time series of water levels and wind speed at the downstream boundary and discharges at the upstream boundaries. To calculate the flood risk, boundary conditions were used for 12 different return periods, ranging from 1 year to 10,000 years. The methodology used for deriving these boundary conditions was based on Vaes et al. (2002). This methodology consisted of generating QDF (Quantity – Duration – Frequency) relations for each of the boundary conditions. QDF relations give the frequency of the discharge in relation to the time period over which the discharge is averaged. These relations can be transformed into composite hydrograms / limnigrams which contain for each return period for all averaging intervals the appropriate discharge/water level. When these types of boundaries are applied to a hydrodynamic model, the model results in all nodes of the model have the same return period. This results in a uniform flooding map for the river and the floodplains.

To account for climate change, a sea level rise of 60 cm by 2100 was included in the downstream boundary as an average value. This value falls within the ranges of 0.09 m and 0.88 m given in the Third Assessment Report of the IPCC (IPCC, 2001) and Belgian assessments between 0.40 m and 0.70 m made by Schoeters and Vanhaecke (1999). For 2050 the same IPCC reports indicate a sea level rise of 22 cm.

For each scenario 24 simulations were done, 12 for a set of boundaries representative for the current situation and 12 for a set of boundaries representative for the situation in 2100. Based on these results 24 flood maps were calculated for each 20x20 m cell of the DEM.

5.2.5.2. Flood damage assessment

The methodology applied to estimate the avoided flood damages in the Flemish region is described in Vanneuville et al. (2003) and builds further on the Dutch HIS-GIS method (Kok et al., 2002), which is based on observed flood damages during a series of flood events (5) between 1953 and 2003. The total damage in a certain area depends on the water depth, number of units of a damage class within the area and the maximum damage or replacement value. Damage factors indicate the percentage of the replacement value at risk as a function of the inundation depth.

$$D_w = \sum_{i=1}^{n} (\alpha_{i,d} * n_i * D_{\max,i})$$

with:

D_w total damage in a certain flood event

$\alpha_{i,d}$ % damaged for damage class *i* as a function of water depth *d* (between 0 and 1)

n_i number of units damage class *i* within a certain area (number, surface, ...)

$D_{\max,i}$ maximum damage per unit of damage class *i*

Table 14 gives an overview of the damage factors for the different land use classes. Table 15 shows for each damage class the maximum damage and the damage factors applied. A distinction is made between direct and indirect damages. Indirect damages reflect mainly clean-up costs (houses, industry, agriculture), reduced production in and outside flooded areas (industry) and fertility losses (agriculture).

Table 14: Damage functions (Vanneuville et al., 2003) – proportion damaged in function of water depth (1 = 100% damaged)

Waterdepth (cm)	Housing	Household furniture	Vehicles	Industry (surface)	Industry (employee)	Recreation	Agriculture
0	0.00	0.00	0.00	0.00	0.00	0.00	0.00
25	0.01	0.12	0.13	0.10	0.03	0.50	0.25
50	0.03	0.24	0.25	0.20	0.05	1.00	0.50
75	0.04	0.35	0.38	0.30	0.08	1.00	0.58
100	0.05	0.47	0.50	0.40	0.10	1.00	0.64
125	0.06	0.48	0.63	0.50	0.12	1.00	0.70
150	0.08	0.49	0.75	0.60	0.13	1.00	0.76
175	0.09	0.49	0.88	0.70	0.15	1.00	0.82
200	0.11	0.50	1.00	0.80	0.16	1.00	0.88
225	0.17	0.54	1.00	0.83	0.18	1.00	0.91
250	0.23	0.58	1.00	0.85	0.19	1.00	0.93
275	0.29	0.62	1.00	0.88	0.21	1.00	0.95
300	0.35	0.66	1.00	0.90	0.22	1.00	0.96
325	0.43	0.70	1.00	0.93	0.32	1.00	0.98
350	0.52	0.75	1.00	0.95	0.42	1.00	0.99
375	0.60	0.79	1.00	0.98	0.51	1.00	1.00
400	0.68	0.83	1.00	1.00	0.61	1.00	1.00
425	0.76	0.87	1.00	1.00	0.71	1.00	1.00
450	0.84	0.92	1.00	1.00	0.81	1.00	1.00
475	0.92	0.96	1.00	1.00	0.90	1.00	1.00
≥ 500	1.00	1.00	1.00	1.00	1.00	1.00	1.00

In addition to monetary damage, the expected number of casualties was also included. The damage function for victims depends both on water depth and rising speed and was derived from Vrisou van Eck (1999).

The number of victims N is calculated as:

$$N = f_d * f_w * A$$
$$f_d = \exp(1.16 * d - 7.3)$$
$$f_w = 0 \qquad \text{for} \qquad w \leq 0.3$$
$$f_w = 0.37 * w - 0.11 \qquad \text{for} \qquad 0.3 < w < 3.0$$
$$f_w = 1 \qquad \text{for} \qquad w \geq 3.0$$

with A number of people present in a certain area, f_d the drowning factor as function of water depth, f_w drowning factor as function of rising speed, d flood height in meters and w rising speed in meters/hour. A value of a statistical life of € 1 million was applied, based on Bickel et al. (2001). Though disputable, no attention was given to further elaborate on this value as simulations showed a very low number of victims due to the low water levels in case of flooding in Flanders.

Table 15: Damage function and maximum damage for different damage classes (Vanneuville et al., 2003)

Damage class	Unit	Maximum damage	Damage function
Houses, real estate	house	€ 95,569.00	Houses
Houses, furniture	house	€ 47,784.50	Houses, furniture
Houses, indirect	house	1 – 15 % direct	
Cars	car	€ 4,627.00	Vehicles
Industry, direct	m²	€ 96.23	Industry surface
Industry, direct	employee	€ 175,820.00	Industry employee
Industry, indirect		35 – 45 % direct	
Arable land direct	m²	€ 0.704	Agriculture
Arable land indirect	m²	10% direct	
Pastures direct	m²	€ 0.196	Agriculture
Pastures indirect	m²	10% direct	
Orchards direct	m²	€ 3,010	Agriculture
Orchards indirect	m²	10% direct	
Surface water	m²	€ 0.000	None
Recreation	m²	€ 0.054	Recreation
Airport	m²	€ 96.23	Industry surface
Highway	M	€ 3,000.00	Industry surface
Secondary roads	M	€ 800.00	Industry surface
Other roads	M	€ 650.00	Industry surface
Railroads	M	€ 7,500.00	Industry surface

A combination of the Corine land cover and the Small Scale Land Use map for Flanders was used. Both are derived from Landsat TM and Spot images. The resolution is too low to see linear structures as roads, railroads and waterways. This is why topographical maps of the Belgian National Geographic Institute are used as an additional data sources. An overlay of these 3 data sources was made to assess the dominating land use in each grid cell (25 x 25 meters). To estimate the amount of houses, employees, people and cars present in a certain grid cell, total amounts listed in municipal statistics are divided among the total amount of relevant grid cells within this municipality. Evidently, the number of houses is assigned to grid cells classified as houses and employees are divided among gridcells classified as industry. People and cars are divided equally among industry, houses and infrastructure.

5.2.5.3. Flood risk Flanders

The damage calculations were performed for several flooding scenarios with a specific probability of occurrence in both 2000 and 2100 with a sea level rise of 60 cm. For intermediate years, results were interpolated.

The total annual risk is equal to the probability of occurrence multiplied by the corresponding damage and this for the total range of possible occurrences. The total annual flood risk can be calculated using equation:

$$R = \sum_{i=1}^{n} \frac{1}{i}\left(D_i - D_{i-1}\right) \tag{1}$$

with D_i the damage related to a flood with a return period of i years.

As not all return periods can be estimated, a probability weighted marginal damage relationship based on linear interpolation between two known return periods is assumed:

$$R = \sum_{x_i}\left[\left(\frac{\frac{1}{x_{i-1}+1}+\frac{1}{x_{i-1}+2}+\ldots+\frac{1}{x_i}}{x_i - x_{i-1}}\right)\times\left(D_{x_i} - D_{x_{i-1}}\right)\right] \tag{2}$$

with x_i, x_{i-1} consecutive simulated return periods, D_{xi} the damage related to a flood with a return period of x_i years.

How this function approximates the probability damage function is illustrated in Figure 20. For the blue line it is assumed that for all return periods damages are simulated. For the red line it is assumed that only for return periods 1, 2, 5 and 10 years simulated flood damages are available. For intermediate return periods, results are interpolated.

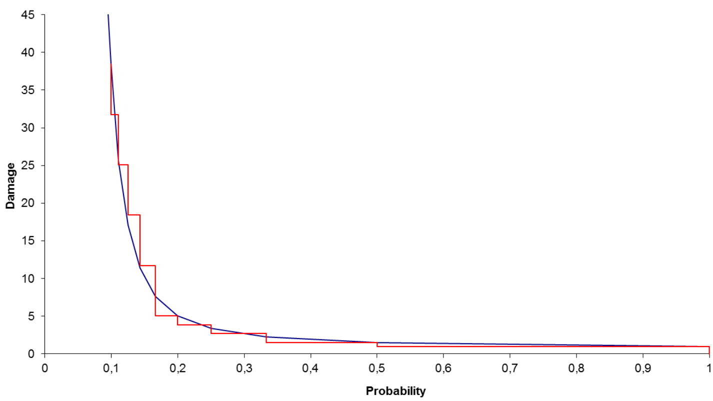

Figure 20: Illustration of probability damage function (blue line) compared to linear interpolation for not simulated return periods (red line)

For the cost-benefit analysis, flood damage simulations were made for x_i = 1; 2; 5; 10; 100; 500; 1,000; 2,500; 4,000 and 10,000 years.

The total annual risk was estimated for the years 2000 and 2100 (after a sea level rise of 60 cm). The difference between the annual risk in the reference scenario and the annual risk after implementing measures equals the annual safety benefit. The safety benefits for intervening years are interpolated based on the estimated annual increase of the sea level.

To account for the increase in the number of people and economic assets located in flood risk zones, long-term economic growth scenarios were applied on the estimated safety benefits. These were based on 3 long-term global development scenarios for Europe determined by the Netherlands Bureau for economic policy analysis (Jansen et al., 1996). The average growth scenario assumed yearly economic growth of 2.4% until 2020 and 1.8% between 2020 and 2030. The same yearly growth was applied for the period after 2030. The other scenarios were tested in the sensitivity analysis (see paragraph 5.3.4).

The applied methodology to estimate flood damage is limited at estimating the total monetary damage and casualties inside the inundated area. This means that other non-monetary damage such as emotional damage was not assessed. Bouma et al. (2005) mention that this kind of damages could be of the same magnitude as direct material damages. Including these effects could significantly increase the benefits.

5.2.5.4. Reduction costs dyke heightening in the Netherlands

Some of the measures also affect the flood risk in the Netherlands. Whereas the Overschelde and floodplains nearby the border were expected to have a positive impact on the water levels of the Dutch part of the Scheldt, a storm surge barrier was expected to have a negative impact as the ability to store water in the Flemish region would be reduced.

The policy concerning the limitation of flood risk differs significantly between the Netherlands and Flanders. Whereas a risk based approach with alternating protection levels had to be applied for the Flemish region, policy within the Netherlands stipulates that a minimal protection level along the Scheldt against storms with a recurrence period of 1 in 4000 years is required. Consequently, a different methodology had to be applied to estimate the impacts of measures on flood risks in the Netherlands.

Due to sea level rise it was expected dykes had to be heightened by a further 1 meter around 2030 and again around 2080 to maintain a 1/4000 safety level in the Netherlands. If however water levels change due to measures aimed at improving the safety against flooding in the Flemish region, a change in the investment scheme of dyke heightening will be necessary. The estimated time shift along various parts of the Dutch part of the Scheldt for different measures is represented in Figure 21. The Overschelde and additional floodplains nearby Antwerp will also decrease water levels on the Dutch part of the Scheldt during 1/4000 flood events. A storm surge barrier however will cause higher water levels on the Dutch part of the Scheldt as the Belgian part of the Scheldt will no longer serve as storage area. This impact will increase for areas close to the Belgian border (Bath, Prosperpolder). By comparing expected water levels for 1/4000 flood events in 2030 for the reference scenario with the expected water levels for other scenarios, it can be estimated how much sooner or later these water levels will be reached due to the Overschelde, floodplains or a storm surge barrier. A storm surge barrier will decrease delay time nearby the border with almost 40 years. Hence, immediate investments for additional dykes are required in this case. The Overschelde will delay investments nearby the border with more than 60 years.

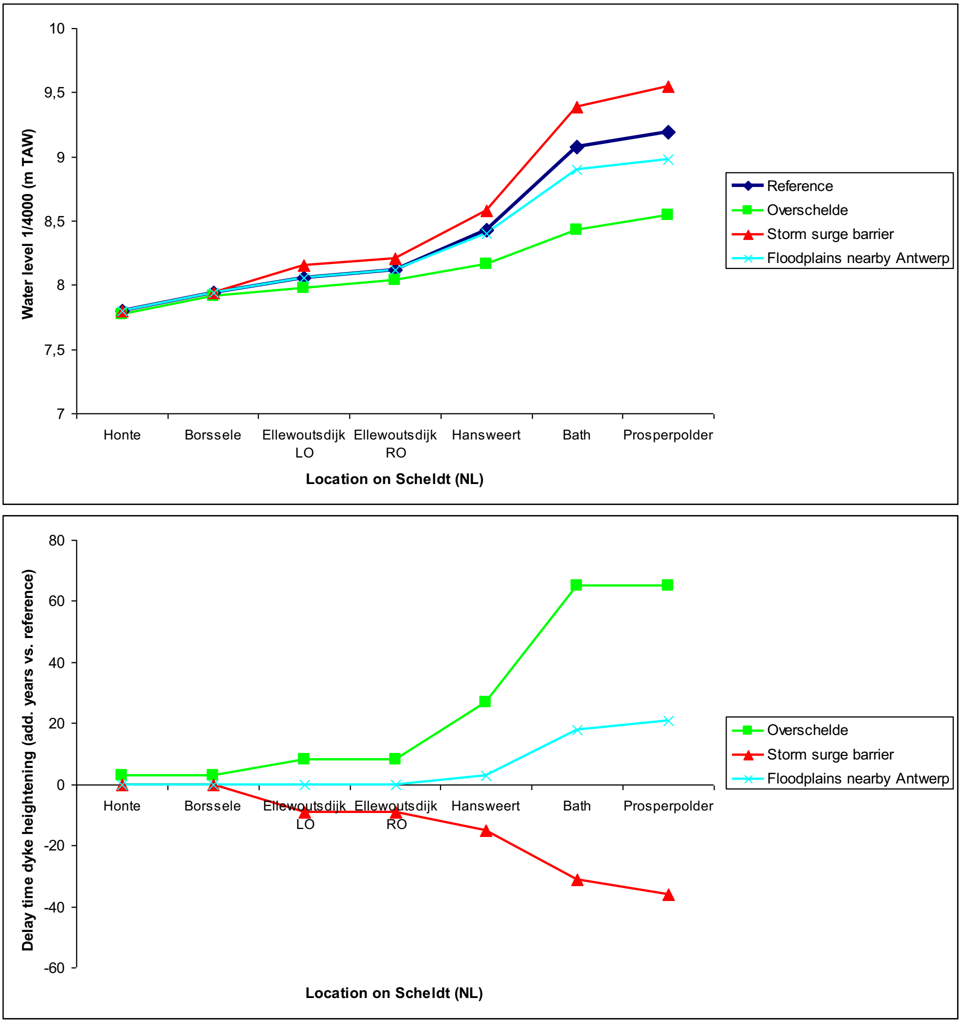

Figure 21: Impact of measures on investment scheme dyke heightening in Dutch part of the Scheldt (figure above indicates the impact of measures on critical water levels, figure below shows the impact of critical water levels on delay time for future investments)

5.2.6. Ecosystem benefits of floodplains

To assess the ecosystem benefits of newly planned floodplains, the so called "ecosystem goods and services" approach was used (De Groot, 2002; Millennium Ecosystem Assessment, 2005). Three groups of ecosystem services were identified: provisioning services, regulating services and cultural services. The so called supporting services were not considered in the CBA because this value is already integrated in the other three groups. Adding their value with the other ecosystem services would lead to double counting.

Valuing these ecosystem services consists of two steps (Table 16):

- quantifying the potential impact using as much as possible area specific models, but where these are missing indicators from literature and expert judgement.
- putting monetary values on the services using market prices, original valuation studies and indicators from literature.

Potential provisioning services of floodplains would be reed, osier, salt crops and possibilities for aquaculture. However, these benefits were not included in the cost-benefit analysis as during debates with stakeholders, it became clear that it was unlikely that the new floodplains would be specifically developed for reed production or aquaculture rather than for natural development.

The new floodplains were expected to have an impact on the water quality through nutrient recycling, aeration and sedimentation, and on climate regulation through fixation of carbon dioxide by photosynthesis of reed and willow and C-burial. This impact on the regulation services was mainly estimated by the MOSES-model (Soetaert and Herman, 1995 a and b). This ecosystem model was developed for the Scheldt estuary in order to study the possible impact of different water management strategies and to prepare a management plan of the estuary. The MOSES-model makes distinctions between the impact of riverine wetlands in the fresh water, brackish and salt zone of the river.

Average values show improved aeration of 10 mol O_2/ha.year, denitrification of 176 kg N/ha.year for fresh water marshes and 107 kg N/ha.year for salt water marshes, N-burial of 252 kg/ha.year and C-burial of 1500 kg/ha.year. For the regulation services that were not modelled we used indicators from literature and expert judgement. Local experts estimated the impact on sedimentation at 200 m^3/ha. C-capture was based on De Groot et al. (2002) and estimated at 6.8 ton C/ha year. P-burial (4 – 56 kg P/ha.year) was taken from Dennhardt et al. (2002).

The monetary values were estimated using the avoided cost or damage approach. Benefits of nutrient regulation an aeration were based on the costs of wastewater treatment in the Netherlands (2.2 €/kg N, 8.5 €/kg P based, 0.14 €/mol O_2 on CIW, 1999). A monetary estimate of damage cost avoided of 20 €/t CO2 or 66 €/t C (Bickel et al., 2005) was used to value the reduced amount of carbon. An extensive literature review performed by Tol (2004) resulted in an average mean value of approximately 67 €/t C (93\$/t C) and a median of approximately 10 €/t C (14 \$/t C). The estimate used is comparable with the average value. The value of soil retention was calculated based on the avoidance of dredging costs (7 €/m^3, expert judgement).

To determine the effect of the planned areas on recreation, the number of visitors the area would attract and the value that people attach to a visit was estimated. To quantify the number of visitors, estimations were based on the accessibility of the new areas (length of walking trail in the planned area) and the amount of annual visits on the existing walking trails (25 visits/day.km walking trail). The value for the recreation function was determined by performing an original contingent valuation study (approx. 800 surveys) amongst recreants and potential recreants asking for their willingness to pay for a visit to newly developed flood control areas within the study area (Ruijgrok and Lorenz, 2004). The average willingness to pay for a visit was 1.68 €. Results showed no significant difference between controlled inundation areas, reduced tidal areas or wetlands. This can be explained by the fact that people attach already a high value to the existing landscape and they perceive the view over the landscape while walking on the inland dyke as a surplus. Depending on the number of flood control areas (the length of the dykes was used as a proxy for the length of the walking trail), the estimated amount of visitors was multiplied with the average willingness to pay for a visit.

To estimate the non-use value, a CVM study with approximately 1600 surveys was carried out specifically for this study (Witteveen & Bos, 2004). The non-use value was determined by asking respondents how much they would be willing to pay per year even if they were not allowed to visit the newly developed areas as described in Ruijgrok, 2001. The average willingness to pay per household to develop more wetland or reduced tidal areas in the Flanders region was 15.5 €. No significant differences occurred between wetlands and reduced tidal areas. These values were not considered in the cost-benefit analysis. First, because no distinction could be made on a ha basis and second, to avoid the danger of double counting as discussed in Barbier (1994) and Andrews et al. (2006). A benefit could be included twice within the evaluation process as people might for instance think of improved water quality and the creation of new habitats when valuing the non-use value. This benefit of improved water quality was already included in the regulation functions.

Table 16: Valuation of ecosystem benefits (€/ha.year) for newly developed ecosystem types (controlled inundation areas, reduced tidal areas, wetlands)

Function	*Quantification (unit/ha.year)*					*Valuation (€/unit)*	
Ecosystem type *Watertype*	*FCA* *Fresh*	*CRT* *Fresh*	*CRT* *Salt-brackish*	*Wetland* *Fresh*	*Source*	*Value*	*Source*
Production functions:* (fish, aquaculture, wood)	pm	pm	pm	Pm	pm	pm	pm
Regulation functions:							
Denitrification		176 Kg	107 Kg	102 Kg	MOSES: Soetaert and Herman (1995)	2.5	CIW (1999)
Decrease of N washed away		252 Kg	252 Kg	252 Kg	VMM (2003)	2.5	CIW (1999)
Decrease of P washed away		31Kg	31 Kg	31 Kg	VMM (2003)	8.5	CIW (1999)
Aeration	pm	23 mol /ha/year	10 mol /ha/year	Pm	MOSES: Soetaert and Herman (1995)	0.14	Witteveen & bos (2004)
Erosion protection		2m^3	2m^3	2m^3	Expert judgement	5	Witteveen& bos (2004)
C-capture in biomass		6.8 ton /ha reed	6.8 ton /ha/reed	Pm	Goossen et al. (1996)	66	Bickel et al. (2005)
Regulation functions only first 15 years after construction:							
Sedimentation		200 m^3	200 m^3	4m^3	Expert judgement	5	Witteveen & Bos (2004)
C-burial		1.5 ton	1.5 ton	Pm	MOSES: Soetaert and Herman (1995)	66	Bickel et al. (2005)
N-burial		148 kg	148 kg	Pm	MOSES: Soetaert and Herman (1995)	2.5	CIW (1999)
P-burial		25 kg	25 kg	Pm	Dennhardt et al. (2002)	8.5	CIW (1999)
Recreational amenities	25 visits/day/km dyke				Witteveen & Bos (2004)	1.68	Witteveen & Bos (2004)
Non-use value*						***pm***	

pm: ‘pro memoriam’ items, not included in the valuation

5.2.7. Shipping

Based on the experience of building a similar storm surge barrier in Rotterdam in 2003 it was estimated that the waterway would be closed approximately 850 hours during construction and testing. Based on the current shipping movements it was estimated that additional internal and external costs of approximately 800 k€ would be caused due to this additional delay time. Compared to the construction costs of the storm surge barrier, this is a relatively small amount (Scheltjens et al., 2004).

5.2.8. Visual intrusion

The impact of visual intrusion due the construction of new dykes around floodplains is based on a hedonic pricing methodology (Luttik, 2000) on 3,000 house transactions in the Netherlands. In this study it was estimated that open space increased housing prices between 6% and 12%. To estimate the number of houses that would suffer from a loss of open space, it was assumed that all houses situated within a buffer of 50 metres and having direct vision on new dykes (no other houses in between) would be subject to a loss of property value.

5.3. Results

5.3.1. Phase 1: Fixed safety standards

As mentioned before the first phase consisted of a comparison between typical alternative protection schemes.

Table 17 shows that the dominant categories are investment costs and flood protection benefits. The impacts of sea level rise on the flood risk are very high and result in a significant increase of the safety benefits. Other impacts such as the ecological benefits from additional floodplains are less significant.

The storm surge barrier near Antwerp has a pay back period around 40 years, which is much shorter than originally assessed in 1981 (Berlamont et al., 1982). This is mainly due to sea level rise. Although the Overschelde could generate safety benefits for both Flanders and the Netherlands, the large investment costs to realise this project do not outweigh the benefits. Policy measures based on higher dykes and floodplains offer substantial flood protection benefits at relatively low costs compared to a storm surge barrier. Although these projects do not guarantee full protection against flooding for very strong and exceptional storms with a recurrence period of 10,000 years, these projects would prevent most of the damage caused by these storms.

Table 17 suggests that floodplains are more cost-effective then dykes. This conclusion cannot be generalized. Due to the large number of options available for construction additional flood control areas or heightening dykes, the costs and flood protection benefits are very location specific. Therefore, a risk optimization approach was applied to find the most cost efficient combination of dykes and floodplains.

Scenario 5 where flood control areas are no longer used for agriculture but used as controlled reduced tidal areas (CRT) creates more net benefits than a similar scenario with flood control areas (scenario 4a). The additional ecological benefits outweigh the additional agricultural losses and investment costs.

Table 17: Costs and benefits of the policy options top down approach (million €)

Scenario	1	2	3a	3b	4a	4b	4c	5
Description	Storm surge barrier	Overschelde	Dyke heightening T4000	Dyke heightening T2500	Flood control areas T4000	Flood control areas T2500	Flood control areas T1000	Controlled reduced tidal areas T4000
Investment and maintenance	387.35	1,597.24	255.04	240.53	216.59	177.41	140.33	233.08
Agriculture	0.74				30.36	28.64	23.22	57.92
Flood protection benefits								
Flanders	739.11	665.11	710.76	691.62	707.39	648.39	624.79	709.79
Netherlands	-11.10	94.87			23.60	23.60	23.60	23.60
Other impacts								
Visual intrusion					-6.68	-6.50	-3.71	-6.68
Ecological benefits					13.35	11.94	8.25	70.62
Shipping	-1							
Total net benefits	339.92	-837.27	455.72	451.09	490.72	471.38	489.38	519.76
Payback period (years)	41	/	28	27	24	22	17	20

Figures are net present values in million € 2002, based on central estimates for sea level rise (60cm in 2100), economic growth (1.8% long term) and discounting (4%).

5.3.2. Phase 2: Risk optimization

Aim of the bottom up approach was to find the optimal combination of dykes and floodplains. In a first step, the best combination of measures in downstream zone 1 was searched, by trial and error. Starting from the best combination in zone 1, alternative measures in zone 2 were tested and this stepwise until zone 5. As measures from zone 3 and 5 have hardly any influence on the water levels in zone 4, both zone 3 and zone 4 built further on the best measures from zone 2.

Dyke heightening is clearly the preferred option in zone 1, as the safety benefits of dyke heightening clearly outweigh the safety benefits from flood control areas. Also, the investment costs for dyke heightening are lower. Increasing dyke heights until 10m TAW (Belgian ordnance level, which is about 2.3 m below local mean sea level) creates additional benefits compared to 9.25 m TAW. The costs for this additional height are however higher than the safety benefits.

The biggest flood risks in the reference scenario were estimated for zone 2. The potential safety benefits within this zone are large, as is shown in Table 18. Contrary to zone 1, dykes achieve much less benefits than flood control areas. The main impact of dykes within this zone was a shift in the flooding location, causing important flood damage elsewhere. The use of flood control areas proved to be the only option to reduce flood risk. Optimizing the amount of flood control areas in this zone is less evident. A small amount of areas (scenario 2-1) achieves the best payback period. However, a larger amount of areas (scenario 2-2) achieves more net benefits. The additional safety benefits in scenario 2-2 were considered to be more important than a shorter payback period. Scenario 2-3 indicates that additional flood control areas will have less impact on safety.

Constructing a small barrier in zone 3 proved to be very inadequate. Though the barrier was able to protect a large zone upstream, the negative impacts downstream were more significant. The flood control areas selected in scenario 3-2 proved to be more efficient than areas selected in scenario 3-1.

Due to the construction of measures in zone 2, the flood risk was already reduced in zone 4. However, additional flood control areas achieved additional net benefits. Dykes within this zone had the same impact as in zone 2. They were able to move the floods but not reduce flood damage.

Scenario 5-1 is a combination of measures in all zones achieving the largest net benefits. In total this scenario comprises the construction of 1,325 ha floodplains and a heightening of 24 km dykes. Compared with the policy option 4c in phase 1 this saves 475 ha floodplains. Compared to 3b, the length of dyke heightening reduced with 316 km. Table 18 shows that this leads to a cost saving of €8 million compared with the cheapest policy option of phase 1. The safety benefits are by far higher than phase 1 policy options with dykes and floodplains. They are even higher than the safety benefits of a large storm surge barrier although the costs are one third of the costs for this measure.

Table 18: Costs and benefits of the policy options bottom up approach (mln €)

Scenario	**1-1**	**1-2**	**1-3**	**2-1**	**2-2**	**2-3**	**2-4**	**3-1**	**3-2**	**3-3**	**4-1**	**4-2**	**4-3**	**5-1**
Zone	**1**	**1**	**1**	**2**	**2**	**2**	**2**	**3**	**3**	**3**	**4**	**4**	**4**	**5**
Description	**FCA 1024ha**	**Dykes 10m (24 km)**	**Dykes 9,25m (24 km)**	**1-3 + FCA 274 ha**	**1-3 + FCA 521 ha**	**1-3 + FCA 1042 ha**	**1-3 + dykes 8.5/9m (119 km)**	**2-2 + FCA 210 ha**	**2-2 + FCA 212 ha**	**2-2 + barrier Niel**	**2-2 + FCA 336 ha**	**2-2 + FCA 684 ha**	**2-2 + dykes 35 km**	**3-2/4-1 + FCA 199 ha**
Investment/maintenance	77.81	26.17	18.25	43.36	73.46	128.41	74.23	100.52	91.52	59.02	92.37	104.87	110.73	131.71
Agriculture	20.17	0.00	0.00	2.07	3.70	5.45	0.00	3.80	6.26	0.00	8.83	15.04	3.70	12.37
Flood protection benefits	99.65	134.14	126.67	463.57	575.31	624.99	271.58	625.45	653.54	-186.5	606.22	609.37	569.00	736.75
Other impacts														
Visual intrusion	-0.50	0.00	0.00	-0.08	0.00	-2.84	0.00	-0.25	-1.00	0.00	-0.73	-1.13	-1.13	-5.18
Ecological benefits	1.92	0.00	0.00	2.40	-0.25	7.46	0.00	4.14	4.28	0.00	5.93	6.09	3.59	8.78
Total net benefits	3.09	107.97	108.42	420.46	501.59	495.75	197.35	525.01	559.04	-245.5	510.21	494.43	457.04	596.26
Payback period (years)	84	15	9	9	12	19	23	15	12	/	15	17	18	16

5.3.3. Consequences of a risk based approach

In the opinion of the Flemish government the benefits of protecting the whole Scheldt river basin against tidal floods with an occurrence of 1/4,000, as is the case in the Dutch part of the Scheldt, would not outweigh the costs. Therefore, a risk based approach had to be applied for the Flemish region. To check the impact of this approach on the final results, it was tested how safety levels varied between regions after implementing the optimal combination of dykes and floodplains. This was estimated by identifying the return period which caused the first flooding of the specific regions. As only a limited number of return periods years were simulated, accuracy of the estimated safety level is limited. When in a certain region the region is flooded during floods with an occurrence of 1/2,500 and not flooded during floods with an occurrence of 1/1,000, a safety level of 1/2,500 is assumed. The actual safety level lies somewhere between 1/1,001 and 1/2,500 years.

After implementing the phase 2 optimal combination safety levels in the city of Antwerp would increase from approximately 1/100 years in the reference scenario to 1/4,000 years. As most of the safety benefits could be achieved in this zone, this zone had the highest protection level. Rural zones had a safety level of about 1/1,000 years. Small cities in the study area had a safety level of around 1/2,500 years. These estimations were based on hydrological conditions during the year 2000. When using a 60 cm sea level rise, safety levels decreased until 1/50 to 1/500 years. This implies that it might be worthwhile to reassess flood protection measures around the year 2050, as more will be known about the exact impacts of climate change and more accurate estimations can be made which eventually might lead to additional measures. The possibility to spread projects in time is also one of the advantages of investing in multiple smaller scale projects instead of a single large-scale project such as a storm surge barrier. This means that the opinion of the Flemish government is confirmed by the model results.

5.3.4. Sensitivity Analysis

5.3.4.1. Discount rate and economic growth

As mentioned before, a fixed discount rate of 4% and economic growth at 2.4% until 2020 and 1.8% after 2020 was assumed in the central estimates. As projects had a long lifespan and impact were considered until 2100, the combination of discount rates and economic growth scenarios had a large impact on results. Alternative discount rates of 3% and 7% were applied. This represented the range of discount rate applied mostly in both the Netherlands and Belgium. Alternative economic growth scenarios were taken from Saitua, 2004. A low economic growth scenario assumed a growth of 1.4% until 2020 and 0.8% after 2020. A high economic growth scenario assumed a growth of 2.8% until 2020 and 2.3% after 2020.

Combining high economic growth estimates with low discount rates increased safety benefits from 737 to 1,672 million € and net benefits from 596 million to 1,518 million €. Combining low economic growth estimates with high discount rates decreased safety benefits to 144 million € and net benefits to 36 million €. Though the variation between these numbers was very large, net benefits were still positive when assuming low economic growth and high discount rates. Considering the ranking of all scenarios mentioned in Table 18, this remained unchanged. The net benefits of the optimal combination were in all cases larger than the net benefits of the phase 1 measures. Especially the fact that the optimal combination had larger benefits than the storm surge barrier under changing assumptions was of great importance for the policy makers. However, the combination of dykes and floodplains which were optimal in the central estimate, did not produce the largest net benefits under all conditions. When assuming a low economic growth and a high discount rate, the safety benefits decreased and consequently a smaller amount of floodplains with lower costs, resulted in larger net benefits. This suggests to gradually implement small scale measures and preferably the most cost-effective (no regret) first. This provides the opportunity to reconsider the decisions during the upcoming decades.

5.3.4.2. Sea level rise

The previous estimates assumed that sea level would rise by 60 cm between 2000 and 2100. As this had large impacts on the results, a sensitivity analysis was performed ranging sea level rises between 0 cm and 120 cm by 2100. To perform this analysis no additional flood simulations were made. Instead, the estimated total annual risks that occurred at a sea level in 2000 and after a sea level rise of 60 cm, were shifted in time. This means that in case of a sea level rise of 0 cm, the total annual risk in 2000 is also valid for 2100, excluding the influence of the economic growth. In case of a sea level rise of 120 cm, the annual risks at a sea level rise of 60 cm, occur in 2061. Safety risks for other years are interpolated or extrapolated based on the estimated annual increase of the sea level. Adding simulation results for 2050, assuming a sea level rise of 22cm was tested for the reference scenario but had no large effects on the total estimated flood risk between 2010 and 2100.

Table 19 shows how the safety benefits for the optimal solution in phase 2 are influenced when applying different assumptions for sea level rise. When no sea level rise was considered, benefits until 2100 did not outweigh costs. In case of a sea level rise of 1.2 m, the safety benefits increase 10-fold compared to no sea level rise and the net benefits increase to 1.2 billion €.

Table 19: Sensitivity of costs and benefits of the optimal bottom up solution for various assumptions on sea level rise

Sea level rise between 2100 and 2000	0 cm	30 cm	60 cm (baseline)	90 cm	120 cm
Investment and maintenance costs	132	132	132	132	132
Loss of agriculture due to floodplain restoration	12	12	12	12	12
Flood protection benefits	138	437	737	1,036	1,335
Ecological benefits & visual intrusion	4	4	4	4	4
Total net benefits	-2	297	596	896	1,195
Payback period (years)	**92**	**24**	**16**	**12**	**10**

Figures are net present values in million Euro 2002, based on central estimates for sea level rise, economic growth and discounting (4%). Non-use values for nature development are not included in the figures.

Though the impact of sea level rise on the benefits is significant, the ranking of measures will not be influenced. An important policy question was whether a storm surge barrier would be more interesting if sea level rise is 120cm instead of 60cm in 2100. The results indicated however that the additional safety benefits in this case were 34 million € compared to the optimised scenario presented in Table 18. This still does not outweigh the additional costs required to construct the barrier. Depending on the rhythm of sea level rise more (if > 60cm) or less (if < 60cm) flood control areas will be included in the optimised scenario.

5.4. Discussion

A methodology was presented that used a combination of hydrological models, damage assessment methods, a cost-benefit analysis including the valuation of ecosystem services to determine a flood risk management plan for the Scheldt river. Both the impact of sea level rise and economic growth between 2010 and 2100 were taken into account. Also, cross boundary impacts (Netherlands) were considered.

Other research papers on predicting flood risks in Brouwer and Van Ek (2004), Hall et al. (2005), De Kok and Grossman (2010), te Linde et al. (2011) and Rojas et al. (2013) use very similar methodologies to estimate current and projected flood risks, taking into account long term economic growth and sea level rise. Projections on future land use change as applied in some of these studies were not available at the time of the study but are being considered in further updates of the methodology (IMDC, 2012). The large difference between these studies and this chapter is the broadness of using this methodology to compose locally optimized scenarios. Most studies are limited either to demonstrating how flood risk increases for different future scenarios of economic growth and/or sea level rise. Some studies also use these methodologies to compare a limited amount of larger scale scenarios (usually floodplain restoration versus dyke heightening), but the composition of locally optimized scenarios is not performed.

Though the sensitivity of some major assumptions on the results was tested in previous paragraphs, a sensitivity analysis was not performed for the underlying models. Recent reviews have compared the Flemish damage assessment method applied in this chapter, with other standard methodologies applied in other European countries. A recent review by Jongman et al. (2013) compared 7 different damage assessment methods with actually observed damages in two case studies (flood events in Germany and UK). Results demonstrate potentially large differences between methods and the observed values. Depending on the case study, the Flemish method is at the middle or higher range of the different damage assessment methods. The Flemish damage assessment method overestimated 2,5 times the observed values for the German case and underestimated more than 4 times the observed values for the UK case. A validation of the Flemish method based on an actual Flemish flood event and reported damage data is not performed by our knowledge. The quantitative results show that the outcomes are very sensitive to uncertainty in both vulnerability (depth–damage functions) and exposure (asset values), whereby the first has a larger effect than the latter. Also, the land use maps are important sources of uncertainty. Typically, damages to infrastructure are underestimated due to an absence of accurate data on the exact location of roads, rails, electricity and communication lines and sewage infrastructure. The location of roads and rails is available in the Flemish method and specific damage assessment methods were developed for these damage categories. For other types of infrastructure, both the location and the damage assessment methods are less well developed.

A comparison of hydrodynamic models, more specifically the predictions of flood depths and the rising speed of water, was performed by Asselman et al. (2007). They performed simulations for two case studies in the Netherlands and Flanders both with the MIKE11 model applied in this chapter and SOBEK, a hydrological model often applied in the Netherlands for flood risk simulations. Results demonstrated major differences in predicting rising speed of floods, how floods propagate spatially across the landscape and the location and starting point of dyke breaching. This causes differences in the severity and the location of predicted flood events. As no further consequences in flood risk assessments were made, it is not possible to evaluate the potential importance for the results of this cost-benefit analysis.

Besides the impacts on flood risk prevention, other ecosystem services are also included in the cost-benefit analysis. Though an area specific model (OMES) was applied to estimate most of the regulating services and specific surveys were taken to estimate the impact on regulation, Benefits Transfer techniques were still required to transfer or distribute the outcomes of study sites to specific project sites. The applied Benefits Transfer methodologies to transfer values from another study site to the project site are only based on the surface of specific types of land use (so called unit value approaches). Important spatial characteristics in the valuation of land use changes were not considered which makes it impossible to determine at which location which type of floodplain is most suited. This makes also the Benefits Transfer less reliable (Brouwer, 2000). Recent approaches (e.g. Bateman et al., 2011; Liekens et al., 2013) apply more accurate value function transfer methods. These methods use functions estimated through valuation applications for a study site, together with information on several parameter values for the policy site to transfer values. Parameter values of the policy site are plugged into the value function to calculate a transferred value that better reflects the characteristics of the policy site (e.g. size of the area, surrounding land use, sociodemographic factors of the beneficiaries nearby, accessibility). This type of methodology is explored in the next chapter.

5.5. Conclusion

The results demonstrate that the risk of flooding will increase significantly due to a combination of sea level rise and autonomous economic development. In view of this increasing risk complementary measures are needed along the Scheldt River to achieve acceptable protection levels. Although more drastic measures as a storm surge barrier near Antwerp offer more protection for very extreme storms, an intelligent combination of dykes and floodplains can offer higher benefits at lower costs. One of the reasons why these smaller projects realise larger net benefits is because they allow for a differentiation in safety levels. Whereas little variation is possible in the location, size and hence costs and effectiveness of a storm surge barrier or Overschelde, many choices are possible with respect to the location and size of floodplains and dykes. Applying a risk-based approach with higher protection levels at high value urban and industrial areas, enables a more efficient allocation of investments. Especially in low-lying countries such as Belgium and the Netherlands, where at the moment a more uniform safety level is applied, a risk-based approach can lead to large cost savings.

As expected, the results show a large influence of sea level rise. The flood risk increased 5-fold when a sea level rise of 60 cm was applied. Hence, the potential safety benefits increased significantly. As a consequence, a measure of the positive net benefits of a storm surge barrier is possible whereas this wasn't the case in a cost-benefit analysis 25 years ago where no sea level rise was considered. Another indication of the importance of sea level rise is shown in the sensitivity analysis. The safety benefits of the optimal solution increase 10-fold when comparing a sea level rise of 1.2m with a scenario without sea level rise. This all shows the importance of considering the impacts of sea level rise when developing long-term flood risk management plans.

An important result was that floodplains proved to be necessary to ensure safety levels in the longer term in the Scheldt basin. The simulations showed that floods could not be prevented only by heightening the existing dykes. In some cases the only consequence was a shift in the location of the flood event. As a result scenarios including floodplains had higher net benefits than scenarios only using dyke heightening, even when only safety benefits and no ecological benefits were included in the analysis.

Though the usefulness of cost-benefit analysis in assessing long term development plans is much debated (Turner, 1978; Joubert et al., 1997), results of this cost-benefit analysis proved to be insightful for policy makers. The results have been used between 2005 and 2010 as a basis for the development of a bi-national long term strategy for the Scheldt estuary and the allocation of the necessary budgets to protect the Scheldt estuary against flooding on a long term basis. The measures which will finally be implemented do not correspond totally with the optimal combination as determined by the cost-benefit analysis. In some cases, the construction of floodplains conflicted greatly with certain stakeholder groups or ongoing policy on other domains, which is difficult to include in a cost-benefit analysis. However, the cost-benefit analysis framework was still applied to check to which degree net benefits decreased compared with the optimal combination. A similar methodology is currently being applied to revise the long term flood management plan for the Belgian coastal region.

The objective of this chapter was to develop a methodology that is able to integrate results from hydrological models in a cost-benefit analysis, simultaneously deal with upstream-downstream effects, take into account the impact of sea level rise and also is able to include a range of environmental impacts which is needed to evenly compare techniques as a storm surge barrier with floodplain restoration. This required also the valuation of ecosystem services. Though results of ecosystem services are much less important compared to the safety benefits we can expect from flood protection, it could however influence the choices we make between individual areas to restore and how to restore the area (controlled reduced tidal area - CRT vs. flood control area - FCA).

Scenario development is performed by combining flood risk simulation models and economic analysis. Spatial interdependencies are dealt with by stepwise scenario development where the study area is subdivided in 5 subzones and the optimal scenario in subzone 1 is the starting point for subzone 2 and etc. This stepwise scenario building, in combination with the long time frames taking into account the impact of sea level rise and the valuation of ecosystem services provides an innovative and powerful decision support framework. Also for surface water quality this could work, however it mostly works vice versa. Starting from upstream areas we estimate what is required to reach the good water status there and this forms the starting point of further optimizations downstream.

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CHAPTER 6. A web application to value the impact of changing land and water management on ecosystem services

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Abstract

This chapter builds further on the valuation of ecosystem services as discussed in the previous chapter. Ecosystem service valuation, performed in the previous chapter, was limited to the demonstration of potential benefits generated by a reduced tidal area or a wetland, compared to a controlled inundation area. As stated previously, the applied Benefits Transfer methodologies to transfer values from another study site to the project site were only considering the surface of specific types of land use (so called unit value approaches). This makes the Benefits Transfer less reliable (Brouwer, 2000) and also makes it impossible to identify where it would be most beneficial to construct which type of floodplain.

The objective of this chapter is to develop and apply an ecosystem service valuation framework that is able to diversify benefits on a local scale, taking into account spatially sensitive and project site-specific input variables, which significantly influence the value of ecosystem services. Assessing the impacts of policies on a wide range of ecosystem services can support the development of cost-effective policies that establish win-win situations across different environmental domains. This gives a broader picture compared to the methodologies discussed in the previous chapters. Whereas most applications on ecosystem service valuation are aimed towards large scale ecosystem service accounting exercises (value of ecosystem services in an entire country or region), methodologies are required that can be linked to the local decisions we are faced with in integrated (land and) water management.

To explore the quantity and value of ecosystem services, the web-based application "nature value explorer" was developed. The application allows to estimate the impact of land use and land cover change in case of nature or river restoration projects on regulating and cultural ecosystem services in Flanders, Belgium. To ensure the applicability in day-to-day decision making as part of environmental impact assessments, user requirements were investigated prior to tool development. Finding the optimal balance between accuracy and complexity on the one hand and flexibility and user-friendliness on the other hand was an important challenge. To date, the nature value explorer has been successful in drawing the interest of policy makers and has been used several times to support decisions in infrastructure projects as well as in nature restoration projects in Flanders. This chapter discusses the user requirements, the main tool characteristics, potential policy applications and future improvements. Three case studies illustrate the functionalities of the tool in day-to-day decision making. The tool can be consulted on www.natuurwaardeverkenner.be.

6.1. Introduction

Humankind benefits from a multitude of resources and processes that are supplied by natural ecosystems, collectively referred to as ecosystem services and goods (Daily, 1997). Degradation of the world's ecosystems during the past fifty years due to urban expansion, agricultural intensification and industrialization has led to a serious decline in ecosystem service delivery (Millennium Ecosystem Assessment, 2005). The key challenge of policy making today is to prevent or reduce this incessant degradation of ecosystems and their services while meeting the increasing demands of society. Since the Millennium Ecosystem Assessment (MA, 2005) the services natural ecosystems deliver are being more and more recognized (e.g. Haines-Young and Potschin, 2011; Seppelt et al., 2011). This is supported by a rapidly growing amount of literature and models on ecosystem service classification, quantification and valuation.

Research on classification focuses on the development of conceptual definitions and a common classification system. One of the most important attempts to classify ecosystem services were subsequently carried out by the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment, 2005) and by The Economics of Ecosystems and Biodiversity (TEEB, 2010), classifying ecosystem services into supporting, provisioning, regulating and cultural services. Recently, the Common International Classification of Ecosystem Services (CICES) was proposed as universal classification system (Haines-Young and Potschin, 2011). Boyd and Banzhaf (2007) and Fisher and Turner (2008) developed an alternative classification that distinguishes only between final ecosystem services, which directly contribute to the goods that are valued by people, and intermediate services, which underpin the final services. So called supporting services thereby fall into the intermediate services category. This distinction attempts to enhance suitability in the domain of ecosystem service valuation and to avoid double counting. Valuation of supporting services separately in combination with final services amounts to a case of double counting.

Quantification of ecosystem services or expressing its importance in physical terms can be performed through ecological models, which vary largely in complexity and spatial detail. These models derive the ecosystem service production potential of a landscape, based on the characteristics of the ecosystem. Recent advances in spatially explicit ecosystem service quantification include several rule based and model based approaches in combination with spatial data (Bagstad et al., 2011; Burkhard, et al., 2009; Kienast et al., 2009; Nelson et al., 2009). For a detailed overview of recent evolutions in spatial explicit assessment of ecosystem services in biophysical terms, we refer to Burkhard et al. (2012).

Monetary valuation or the translation of physical quantities in monetary terms is used to estimate the total impact on different ecosystem services, thus, allowing comparisons between individual services, costs and other economic impacts. Though its use is often subject of debate, besides scoring and weighting individual ecosystem services it remains one of the few options to integrate and estimate a total impact on different ecosystem services. For most ecosystem services, the validity of the estimated monetary values can be significantly improved by using spatially-explicit economic models. Monetary valuation should reflect, for example, that biophysical impacts that occur in densely populated areas are given higher values (positive or negative) than impacts in relatively remote areas. Recreational values are often higher for better accessible nature areas closer to population centres. Monetary valuation is only meaningful when 'marginal changes' are considered, i.e. changes that are relatively small or incremental at the scale of the analysis (Fisher et al., 2008; Turner et al., 2010), that fall within the safe minimum standard of ecosystem functioning. Valuing the benefits of nature areas and undeveloped land has a long history and is based on a variety of monetary valuation methods, including price-based methods such as avoided (damage) costs, revealed preference methods such as travel cost (e.g. Fleischer and Tsur, 2000) and hedonic pricing (Mansfield et al., 2005; McConnell and Wallis, 2005) and stated preference methods such as contingent valuation (e.g. Bateman et al., 2011; Bateman et al., 2002; Mitchell and Carson, 1989) and choice experiments (Adamowicz et al., 1997; Brouwer et al., 2010; Colombo et al., 2006; Hanley et al., 2001). A combination of valuation techniques is often required to comprehensibly value ecosystem goods and services (DEFRA, 2007). The choice of the valuation methods will depend on the characteristics of the case, including the scale of the problem, the types of value deemed to be most relevant, data availability and the availability of human and financial resources.

Though methodologies for classification, quantification and valuation are improving, applications of the ecosystem services concept stay mainly restricted to illustrating the importance of preserving or restoring ecosystems in regional to global ecosystem service mapping or ecosystem services accounting. Important examples of large scale ecosystem service assessments include the National Ecosystem Assessment in the UK (Bateman et al., 2010), the Natural Capital/INVEST project (Kareiva et al. 2011), and the "Valuing the Arc" initiative (Fisher et al., 2011). Its use in day-to-day decision-making processes remains limited however, especially at the planning level (Daily and Matson, 2008). Both limited interest among geographers and excessive complexity of currently available models are depicted in the literature as major reasons for this lack (Haines-Young and Potschin, 2011; Koschke et al., 2012; Seppelt et al., 2011). Frequent occurring mismatches between the spatial scale of research and the spatial scale of applications can be another reason for limited applicability of current research in spatial planning (Meinke et al., 2006). Nevertheless, spatial planning decisions would benefit from systematic considerations of their effects on ecosystem services (Geneletti, 2011). Estimating the impacts of policy on a wide range of ecosystem services can also serve as an element in the development of more cost-effective policy implementation, establishing win-win situations across different environmental domains as water, air and climate change. To date, most tools for environmental impact assessment (e.g. Cost-benefit Analysis, Strategic Environmental Assessment, Life Cycle Analysis) do not include impacts on ecosystems (alterations in vegetation and biodiversity) or do it in a very simplified way (Ahlrot et al., 2011; Rabl and Hollander, 2008; Allacker and De Nocker, 2012; Finnveden et al., 2005).

In this chapter, we present the "nature value explorer" (natuurwaardeverkenner in Dutch), a web application specifically built to explore the quantity and value of ecosystem services in day-to-day decision making in Flanders Belgium, as part of a strategic environmental assessment (SEA) or a cost-benefit analysis (CBA). The Nature Value Explorer combines spatially sensitive and site-specific inputs with generic quantification and valuation functions, allowing effective and straightforward identification of service providing areas to support spatial planning in Flanders. The application is developed to estimate the impact of land use and land cover change on ecosystem services. It does not address degradation of habitat quality.

This tool can be used for the valuation of benefits, necessary to consider multiple environmental aspects simultaneously in a cost-benefit analysis. This gives a broader picture on effects vs. costs compared to the methodologies discussed in the previous chapters. Whereas most applications on ecosystem service valuation are aimed towards large scale ecosystem service accounting exercises (value of ecosystem services in an entire country or region), we need to develop methodologies that can be linked to the decisions we are faced with in integrated (land and) water management. This means we need to be able to assess and value the impact of individual measures as floodplain restoration or bufferstrips on a range of ecosystem services, which is difficult to do with tools previously developed.

As the end-user perspective is a crucial first step in the design of practical tools, we start from an inventory of user requirements for quantification and valuation of ecosystem services in Flemish policy making in Section 2. These requirements are used to define the design characteristics of the web application, described in section 3. Section 3 also describes the applied methodology for quantification and valuation. Some example case studies are presented in Section 4. In Section 5, we discuss the advantages and limitations of this web application.

6.2. User requirements

To ensure practical use of tools in environmental impact assessments and other policy appraisals, the role of practitioners and end-users in the design of policy support tools should be considered prior to their design (Morgan, 2012, Cashmore, 2004, Morrison Saunders 2003). Therefore, an end-user consultation was organised with the main objective to identify what was required to mainstream the ecosystem services approach in daily practice and which sort of tools could support this process. User requirements and potential policy applications for ecosystem service based approaches were derived from 26 individual face to face end-user consultations. The involved end-users are a mix of organizations involved in policy preparation, policy execution, policy evaluation and civil society organizations. The list covers different actors with a prominent role in management of the open space in Flanders (recreation, agriculture, nature, water management).

An important conclusion that can be drawn from all consultations is that the general interest from potential end-users in Flanders in applying the ecosystem services concept in decision making processes is very large. Not only typical nature conservation administrations or civil society organizations express an interest, but also end-users focussed on spatial planning, agriculture and land and water management consider more in-depth knowledge of ecosystem services as added value for policy making and existing assessment frameworks.

The expected advantages to apply ecosystem service based approaches confirm typical advantages listed in TEEB (2010). Demonstrating the importance of nature and biodiversity and arguing for the protection of existing nature or for additional nature development was often mentioned. Developing more effective policies concerning environment and nature and assessing the efficiency of using ecological infrastructure for multiple policy domains (single objective measures versus multiple objective measures, multi-functional versus mono-functional landscapes) was also a strong motivation. Implementing and mainstreaming the use of the ecosystem services concept in cost-benefit analysis (CBA), strategic environmental assessment (SEA) and other assessment frameworks was considered important to reach these objectives.

All end-users agreed on the need for practical, domain- and authority-independent information or tools that are open to many end-users and which can be tailored for specific needs. Specifically for SEA, the need was expressed for an integrated tool that not only makes an inventory of the ecosystem services but also how the policies, plans or programmes affect these services, which trade-offs between different ecosystem services are necessary, which win-win situations (decisions in one sector that may also positively affect goals of other sectors) can be achieved and how specific stakeholders are influenced. This confirms the points made by Geneletti (2011).

Important requirements listed specifically for the tool were user-friendliness, transparency, flexibility and scientific reliability. User-friendliness is especially important for non-specialist users. The tool needs to make clear what a specific service exactly means, how this service can be quantified and valued and where required input data can be found. Crucial input data should be readily available or at least it needs to be made clear how input data can be gathered. The tool also needs to be sufficiently flexible to address future questions and include new insights. A modular approach is considered important where each service can be updated easily in the future and additional services can be included. It should also be possible to adjust the required spatial detail as required for any case study. End-users stated that budgets are often inefficient to develop extensive site-specific and spatially explicit ecosystem assessments for every new project. However, user-friendliness, transparency and flexibility on the one hand, high accuracy and scientific reliability on the other hand, are properties that do not match easily and trade-offs between accuracy and applicability are unavoidable.

6.3. Methodology

6.3.1. General design

The web application allows for the estimation of two groups of final ecosystem services: cultural services and regulating services. For cultural services we consider the amenity and non-use value. Regulating services include nutrient retention and climate regulation (sequestration in soils and biomass), air quality regulation and noise mitigation. Provisioning services, such as food and fibre production or water supply, were not included. The focus of research was on services that are not yet included in existing SEAs and CBAs. Provisioning services such as (loss of) food or wood production are usually already included in the socio economic consequences or the cost side of such SEAs and CBAs. Table 20 gives an overview of the services and quantification and valuation methods used.

Table 20: Quantification and valuation methods ecosystem services applied in nature value explorer

Service	Sources / important site specific variables	Valuation method
Cultural services	Liekens et al. (2013): nature type, size, accessibility, surrounding environment, distance to households, species richness	Choice experiment: marginal willingness to pay (value function)
Denitrification	Seitzinger et al. (2006): residence time, pollution level Pinay et al. (2007): soil moisture and texture, pollution level	Avoided cost method for N (Flemish unit value: 74 €/kg)
N, P, C sequestration in soils	Meersman et al. (2008): soil drainage, vegetation type and soil texture	Avoided cost method for N and P (Flemish unit values: 74 €/kgN, 800 €/kgP) Avoided/damage cost method for C (Global unit value: 183 €/tonC)
N, P, C sequestration in forest biomass	Meta-analysis: tree type, age, forest management	
Impact on air quality	Oosterbaan and Vries (2006): vegetation type	Damage cost for PM (Flemish unit value rural areas 30€/kgPM)
Noise mitigation/buffer function	Huisman (1990): vegetation type, noise intensity, width and distance to houses	Hedonic pricing (area specific housing prices)

Building a tool for assessment of ecosystem services requires a trade-off between the scope (number of services, spatial range), the level of detail and complexity and user-friendliness. The nature value explorer offers less detail than some specific models that focus in detail on a single service or area, but offers a more detailed and accurate assessment than fixed €/ha values per vegetation type. The latter are not preferred as for specific services, the vegetation type is not the major factor influencing the magnitude of the service and is insufficient to capture the spatial variation in the delivery of ecosystem services. At the same time, extensive, process based model calculations were considered too complicated and too computationally intensive to include in a web application to explore the impact on ecosystem services. Instead, quantification functions were developed that on the one hand take into account the main driving factors of the underlying ecological processes such as soil texture, groundwater level and vegetation type and on the other hand require little computation time. The quantification functions build on regional datasets (existing land-use/land-cover and soil map classifications) and studies to increase the accuracy and transparency.

Similar approaches are applied in other practical tools for environmental impact assessments that look at different impact categories and environmental media and integrate ecological with socio-economic analysis (e.g. life cycle impact assessment models or external health costs assessment methods, used in Mirasgedis, 2008; Rabl and Holland, 2008; 2008; Allacker and De Nocker, 2012; Michiels et al., 2012).

Also for monetary valuation, spatially-explicit transferable value functions are used as recommended by Bateman et al. (2011) because they capture the fact that values vary across space and depend on spatial characteristics. These functions do not require time-consuming valuation studies for each case study. Instead, values of the policy site are “plugged into” the value function to calculate a transferred value that sufficiently captures the specific characteristics of the policy site (Brouwer, 2000). When the function includes variables that vary across space, such as land use or socio-demographic characteristics, differences in context can be controlled for.

As stated, the web application does not allow for detailed spatially explicit ecosystem service quantification and grid based computations. Instead, a flexible system of service providing units (SPU) for which end-users can define specific properties (e.g. soil characteristics, vegetation type) and thus the potential to vary the spatial detail was advocated. If budgets are more extensive and the availability of data is not an issue, users can decide to define for each scenario a large amount of SPUs on a relatively small scale (up to 1 ha). More rapid exploratory assessments can also be performed by defining the case study area as a single SPU, limiting the required effort to defining only one input value for different physical characteristics (e.g. soil type, groundwater depth) per vegetation type.

The tool can be consulted on the internet via www.natuurwaardeverkenner.be. More detailed information on the methodologies can be found in the background documents made available on this site. End-users are able to create and save scenarios, share scenarios with other registered users and consult public scenarios. Interactive discussions are stimulated through a discussion forum. User-friendliness is increased by adding information boxes explaining each service and its required input data, a section with frequently asked questions and an information page containing background documents and publications related to the nature value explorer.

6.3.2. Quantification of regulating services

6.3.2.1. Denitrification

The quantification of denitrification processes in wetland ecosystems is based on estimates from Seitzinger et al. (2006). Removal efficiency depends mainly on the residence time of the water in the ecosystems. For terrestrial ecosystems we used estimates from Pinay et al. (2007) to deduct potential denitrification. The quantification of denitrification processes in wetland ecosystems is based on the formula for nitrogen removal in wetlands from Seitzinger et al. (2006). The main parameters are water depth, size of the area and seasonal average daily outflow, which determines the residence time and nitrogen load entering the aquatic ecosystem. For terrestrial alluvial soils we use Pinay et al. (2007). Removal efficiency depends on soil moisture, silt content and herbaceous plant biomass.

6.3.2.2. Carbon and nutrient sequestration soils

Carbon sequestration in soils is based on estimates from Meersman et al. (2008). They performed a multiple regression approach to assess the spatial distribution of Soil Organic Carbon (SOC) and its dependency on soil characteristics in Flanders, Belgium. Based on this model we determine a potential maximal carbon content for a given soil drainage, vegetation type and soil texture. Changes in soil drainage and/or vegetation will change the potential maximal carbon content. The annual carbon sequestration potential is a percentage of the difference in potential carbon content and actual carbon content. This approach is process based and incorporates changes in potential storage and the associated temporal dynamics. Literature estimates of net ecosystem exchange have a very broad range, as they are usually time-specific and do not incorporate long-term dynamics and driving variables such as soil properties, climate, and soil hydrology.

The associated retention of nutrients in soils (or avoided nitrogen leaching) is considered as an additional service. Drainage of formerly wet soils increases nitrogen leaching through mineralisation of the soil organic carbon (Behrendt et al., 2004). The nitrogen (N) and phosphorus (P) content of soils is derived from the carbon content. Based on analyses performed in Flanders, the C/N ratio varies between 10 and 30 depending on the nature type. Based on Koerselman and Meuleman (1996), the average N/P ratio is set at 15.

6.3.2.3. Carbon and nutrient sequestration biomass

Carbon sequestration in forest biomass is based on a meta-analysis (Berger et al., 2007; Cairns et al., 1997; Hees, 1997; Kemmers and Mekkink, 2002; Milne and Brown, 1997; Nabuurs, 2003). Depending on the tree type, age of the forest, type of forest management (intensive, limited or no management) and soil characteristics the level of sequestration differs. N and P content of biomass are set to 4 kg N and 0.4 kg P/ton biomass based on a meta-analysis (André and Ponette, 2003; Hytönen and Saarsalmi, 2009; Maclean and Wein, 1977; Ponette et al., 2003; Uri et al., 2003).

6.3.2.4. Air quality

It is well documented that trees and vegetation can serve as effective sinks for air pollutants and PM10 (particulate matter < 10µg) and thus contribute to air quality improvement and related public health benefits (Nowak et al., 2006; Tiwary et al., 2009). As PM10 is the most important pollutant, accounting for 60% of health impacts from environmental pollution (MIRA, 2008), the focus of this analysis is on PM10.

The amount of PM10 removed from the air depends on several factors including the amount of air pollution, local wind speeds (depending on meteo and landscape) and type of vegetation, especially the total leaf surface and its characteristics. Although literature provides detailed studies for specific tree species at specific locations, the analysis of air quality impacts in integrated modelling is typically based on more generic indicators expressed as yearly values in kg per ha for generic vegetation types (Hein, 2011; Nowak et al., 2006; Tiwary et al., 2009). As there are no data available for Flanders, the estimates in the web application are based on removal factors for individual trees and shrubs from Oosterbaan et al. (2006). The removal factors (expressed in kg/ha) are in the same range (+/- 50 %) of these used by Hein (2011), Nowak (2006) or Tiwary (2009) for grasslands. Trees and vegetation have also impact on other air pollutants, but there is more uncertainty about the removal factors and on the valuation of sinks.

6.3.2.5. Noise mitigation

Nature areas can contribute to the mitigation of noise from for example traffic. The effect of the soil and especially the vegetation is often underestimated in models for noise-simulation (Huisman, 1990; Goossen and Langers, 2003). The service is only important when there are people affected.

Noise mitigation for soft soils and forests is derived from Huisman (1990) and Milieubeheer (2002). Huisman (1990) measured the decrease in decibel (dBA) based on the frequency of the source, the soil characteristics, the meteorological effects and the noise penetration in the forest. He found an average decrease of 6-16 dBA for 100 to 300 m wide forests.

6.3.3. Valuation of regulating services

A variety of monetary valuation methods exists. Each of these methods had its advantages and disadvantages (Champ et al., 2003; De Groot et al., 2002; Freeman, 2003; Hanley and Barbier, 2009). We use a combination of avoided abatement costs (for nutrients and carbon sequestration), damage costs (for air pollution), hedonic pricing (for noise mitigation) and stated preference methods (for cultural services).

6.3.3.1. Nutrient removal and sequestration

The avoided abatement cost method is used to value nutrient removal, as costly abatement measures to obtain environmental goals can be avoided due to the natural denitrification that an ecosystem delivers. Other methods in literature to value nutrient removal include avoided health damage costs and stated preference methods. Avoided health damage costs as applied in van Grinsven et al. (2010) are less relevant for nutrient removal in surface water. Stated preference methods are less suitable to assess the value of removing specific nutrients as respondents will experience difficulties in expressing preferences due to little experience and understanding of the specific service (Bateman et al., 2011). The specific value of an additional kg N or P removed by an ecosystem is derived from the marginal cost curve of N and P removal, which was calculated for the Flemish river basin management plan to reach a good water status according to the European Water Framework Directive (chapter 3). The costs of the measures with the highest marginal cost included in the first programme of measures (Coördinatiecommissie Integraal Waterbeleid, 2009a) to improve water quality are 74€/kg N and 800€/kg P. This marginal cost is not the marginal cost to reach water quality targets, as this is technically not feasible with the more conventional measures included in the programme, but represents the most expensive investments the Flemish Region is making to reduce nutrient emissions (paragraph . Most measures have impact on both N and P, and it is therefore impossible to individually link avoided costs to separate pollutants. To avoid double counting, we estimate the value of nutrient retention for both pollutants but only apply the maximum value. The valuation of nutrients applied here is significantly higher than figures in literature, which vary between 2 and 20 €/kg for N (Gren, 1995; Jenkins et al., 2010) and 70 €/kg for P (Borjesson, 1999). Nutrient pollution is a large problem in Flanders requiring significant efforts in wastewater treatment and reducing diffuse pollution. Most of the cheaper measures, such as advanced treatment in large scale wastewater treatment plants, which are often the basis to value avoided abatement costs for nutrients, have already been taken and less cost-effective measures are necessary to reach environmental objectives, justifying the choice for higher values for N and P removal in the nature value explorer.

6.3.3.2. Carbon sequestration

The benefits of carbon sequestration are not related to the place of sequestration, but rather experienced at a global level, through the impact on climate change. To assess the monetary value of carbon sequestration by ecosystems, three different methods can be used: market prices, marginal damage costs and avoided abatement costs. As impacts are global, the selected data are based on studies at the global level. The range of results from studies estimating the marginal damage costs is very broad, ranging from an external costs close to zero to 160 $\$_{1995}$ /ton CO_2 (Tol, 2008, 2005). These studies also show that marginal damages from emissions in 2030 will be significantly higher compared to emissions in 2010 (Anthoff, 2007). An avoided abatement cost approach as used for nutrient uses the costs of mitigation measures for climate change as an indicator of the social value of carbon sequestration. The costs of the mitigation measures are based on models that estimate the emission reductions required to limit temperature increase to a maximum of 2°C. Studies estimate these abatement costs from 10 to 60 €/ton CO_2 for the year 2010, but these marginal abatement costs will increase in the coming decades as more expensive measures will be required to further limit CO_2 emissions and to ensure that the 2°C target is within reach (Kuik et al., 2009). Based on these range of values in literature, we have taken the value of 50 €/ton CO_2-eq. (183 €/ton C), in line with a study on economic aspects of climate change for the Flemish Environmental Agency (MIRA, 2008). Prices are comparable with prices for CO_2 in the beginning of the EU Emission Trading Scheme but prices in 2012 were much lower (+/- 5€/ton) (Thomson Reuters, 2014).

6.3.3.3. Air quality

Air quality improvement for PM10 has important benefits for public health, especially related to cardiovascular and respiratory impacts (Brunekreef and Forsberg, 2005; WHO, 2006; Michiels et al., 2012). These impacts are typically valued using indicators related to avoided costs for health care and medicine, loss of productivity at the workplace and at home and willingness to pay to avoid suffering and loss of life expectancy. There are several integrated studies that look into the impact of emissions to air on air quality, public health and its valuation (Spadaro and Rabl, 2008; Spadaro and Rabl, 2009). The results of these studies have been widely used in economic assessment tools and policy preparation (Rabl and Holland, 2008). In line with Nowak (2006), we assume that we can use valuation data for external health costs caused by emissions of PM10 from low stacks (e.g. buildings) as a proxy for the PM10 removal by trees and shrubs. The data are based on results from air quality models for Flanders, dose-response functions and valuation data from European research projects (Bickel and Friedrich, 2005; Rabl and Holland 2008; Michiels et al., 2012). We further account for the size and origin of the particles and we attribute health effects only to the PM2.5 fraction from man-made sources within PM10. This results in a value of 30 euro/kg PM10 removed by vegetation.

6.3.3.4. Noise mitigation

For noise mitigation we apply the hedonic property price method. This method examines the premium which people are prepared to pay in order to purchase houses in areas of higher environmental quality, e.g. quieter, less polluted neighbourhoods (Bateman et al., 2011). Theebe (2004) and Udo et al. (2006) indicate that the market value of properties decrease with 0.4% per dBA at lower noise levels (40 dbA) and 1.9% at higher noise levels (60 dBA). This information is applied in combination with area specific housing prices.

6.3.4. Cultural services

Cultural services include use values related to recreation, amenity and education and non-use values related to bequest values. For all of these individual services, specific quantification methods and valuation techniques can be used or stated preference techniques that are able to capture all cultural services in single willingness to pay estimates. Given the lack of valuation studies for specific cultural services in Flanders, a stated preference study (choice experiment) surveying willingness to pay for nature restoration, was performed to capture all cultural services in a single value function. This study is described in detail in Liekens et al. (2013). In a choice experiment, respondents are presented with a number of alternatives from which they are asked to choose their most preferred option. The alternatives can be a good or service, but also policy alternatives or land use change scenarios (Louviere et al., 2000). Each choice alternative is defined in terms of the same elements or so-called "attributes", including a price, and has a unique combination of the levels of the attributes. Examples include varying levels in biodiversity (high-low), accessibility (accessible or not) and size of the area (between 1 and 200 ha). As respondents express their preferences by making choices between different alternatives, they trade-off the different attributes and levels. A statistical function can then be estimated that links choice probabilities to the characteristics of the alternatives. The trade-off between price and other attributes is especially relevant, as this reflects how much a respondent is willing to pay (WTP) for a particular change in this attribute. This allows to determine marginal values for changes in the attributes and combinations of attributes. The statistical functions are estimated using simulated maximum likelihood methods in the software package NLOGIT.

In the choice experiment for the nature value explorer, respondents were asked to choose between different land use changes related to the creation of different types of nature area with different spatial and non-spatial characteristics and impacts on their current tax levels. Agricultural land use, with no particular nature or landscape value is the reference situation in the rural areas where these land use changes can take place. Based on the information obtained in focus groups with lay-people and expert interviews, seven attributes influencing willingness to pay were included in the choice experiment: (a) nature type including marshes, natural grasslands, forests, open water and swamps, heath land, inland dunes, and pioneer vegetation; (b) species richness; spatial attributes including (c) size of the area, (d) accessibility, (e) surrounding land use and (f) distance to the respondents' residence. The monetary attribute was a mandatory annual tax to be paid by all Flemish households into a fund exclusively used for the creation and conservation of nature areas in Flanders. The data were obtained from an internet survey conducted through a marketing bureau panel from which respondents were randomly chosen in three different provinces of Flanders. 3,000 residents filled out the survey. After data cleaning (removing incomplete questionnaires and protest bidders (6%)), approximately 10,000 observations from around 2,300 respondents were included in the analysis. Protest relates to a choice for the opt out based on contextual components that affect respondent trust and confidence in the simulated market, including the payment vehicle, respondent belief in the feasibility of public good provision based on the presented information, or that the market is procedural just (Brouwer and Martín-Ortega, 2012). The analysis of the socio-demographic information of the respondents suggests that the sample is largely representative for the Flemish population (see Liekens et al., 2013).

The resulting value function is used to value a nature area, according to some selected biophysical characteristics (nature type, size, surrounding land use, access and species richness) as well as household related characteristics (income, mean age, member of nature organizations, distance to the created nature area). This function for additional nature development in Flanders expressed in annual € per household can be written as:

WTP in €/household/year = 122 * pioneer vegetation + 93 * mudflat and marsh + 92 * natural grass land + 157 * forest + 133 * open water, reed and swamp + 133 * heath land and inland dunes + 0.05 * size in ha + 28 * species + 34 * availability of walking trails – 0.63 * distance in km + 8 * natural surroundings + 8 * residential surroundings – 15 * industrial surroundings – 0.36 * high number of species * age + 0.01 * monthly net income - 37 * % women+ 108 *% membership.

The results show that respondents value forests higher, but pioneer vegetation, marshes and grass lands lower than open water, swamps and heath land. Respondents are willing to pay more for easily accessible nature, but also express a positive WTP for areas that cannot be accessed through walking trails. As distance to the respondent's residence was an attribute in the survey, we could derive a "distance decay function" (Bateman et al., 2006), which adjusts individual WTP downwards as respondents live further away from the proposed land use change. The number of households with positive WTP can thus be based on our empirical value function, rather than by making arbitrary assumptions about the relevant spatial size of the economic market or restricting the market to reflect the political boundaries of a region. By combining the value function with GIS-data on the number of households per spatial unit in the surroundings (up to 50 km), socio-demographic data and distances to the created area, the total value of cultural services for a specific land use change is calculated.

6.4. Results for selected case studies

6.4.1. Impact of infrastructure project on an existing ecosystem

Most of the applications in the Nature Value Explorer up till now are related to the demonstration of the value of existing nature areas, which are threatened by urban expansion. An example case is related to the planned investments in the road network around the city of Antwerp to mitigate the effects and economic costs of traffic congestion.

One of the projects of the 'Masterplan Antwerp 2020' involves a partial relocation of a secondary road (R11) and the creation of a tunnel under a green area of 16 ha. The Province of Antwerp asked the Research Institute for Nature and Forest (INBO) for an advice on the area's species richness as well as its ecosystem services value. The nature value explorer was used to support a first estimate or 'quick scan' to assess the volume and economic value of the ecosystem services.

The area partly consists of an elevated and forested 'green ribbon' of 10 ha along the current road and a broader part of 6 ha that is managed by a local nature conservation NGO. The area currently supports soft recreation and serves as a playground. The area is composed of deciduous forest (84%) and natural grassland (14%) with some fragments of wetland and pioneer vegetation. Potentially important ecosystem services are the amenity and non-use value (cultural services), air quality regulation (capture of fine particles by vegetation) and noise buffering by forest vegetation. The latter service was however not monetized because the impact of the relief of the area (elevated former railroad way) is considered more significant than the impact due to vegetation and this could not be included in the nature value explorer. Since the soil consisted mostly of well drained sand and was not directly in contact with a water system, C-, N- and P-retention in soil was not included either. C-, N- and P-storage in forest biomass was included because of the high forest cover. Nitrate reduction by biological denitrification (water purification) was also included because of the wetland fragments present in the area.

Table 21 shows that the application of the nature value explorer resulted in an economic value estimate of 1.380 k€/year. 95% of this estimate consisted of the value for cultural services (1.323 k€/year); 28 k€/year water purification; 19 k€/year air quality regulation; 11 k€/year climate regulation.

Table 21: Results case study 1 – ecosystem services small existing nature area

Service	16ha, 13 ha forest, 2 ha grassland, 1 ha wetland and pioneer vegetation		
	Quantity		Value
Regulating services	**Amount**	**Unit**	**k€/year**
Nutrient retention			
N-sequestration soil	NA	kg/year	NA
P-sequestration soil	NA	kg/year	NA
N-sequestration forest biomass	265	kg/year	20
P-sequestration forest biomass	26	kg/year	21
Denitrification	88	kg/year	6
Climate regulation			
C-sequestration in soil	NA	ton/year	NA
C-sequestration in forest biomass	58	ton/year	11
Improvement air quality	525	kg PM	19
Noise mitigation	NA	dBa	NA
Cultural services	1,377,219	Households	1,323
TOTAL			**1,380**

NA: not applicable for case study

6.4.2. Nature redevelopment

A second case study is related to the same masterplan for Antwerp but provides an example of the temporary removal of an existing ecosystem and the partial restoration of an area. 173 ha of nature areas nearby the city are temporary required as construction areas and will be partially redeveloped after completion of the works. However, according to the redevelopment plans, the size of the nature areas will be reduced and the composition of the nature areas will change. The nature conservation agency Natuurpunt Antwerp North estimated the loss of ecosystem services due to the temporary loss and the partial redevelopment.

Ecosystem services affected include C, N and P sequestration in soils and forest biomass, denitrification, air quality and the amenity and non-use value. Sequestration in soil reduces slightly, mainly due to the decreasing total area available for nature development. Sequestration in soil biomass reduces significantly due to the young age of newly planted trees, leading to less biomass sequestration in the first 10 years. Denitrification increases due to the growing amount of wetland. The impact on air quality is dominated by the slightly growing amount of forest. The amenity and non-use value decreases as the surface of the nature area is also decreasing. This is not compensated by the increased amount of forest and wetland. Insufficient information on existing noise levels was available to estimate the impact on noise reduction. An average estimation of the annual value of ecosystem services generated by the existing nature areas was 17.3 million €/year whereas the alternative nature redevelopment was estimated to deliver ecosystem services for 15.9 million €/year (Table 22).

Table 22: Results case study 2 – nature redevelopment

Service	Existing situation: 173ha 60ha grassland, 103ha forest, 10 ha marshes			Redevelopment: 147ha 15ha grassland, 106ha forest, 26ha wetland		
	Quantity		Value	Quantity		Value
Regulating services	**Amount**	**Unit**	**k€/year**		**Unit**	**k€/year**
Nutrient retention						
N-sequestration soil	33,676	kg/year	2,492	24,338	kg/year	1,801
P-sequestration soil	2,246	kg/year	1,797	2,155	kg/year	1,724
N-sequestration forest biomass	1,554	kg/year	115	324	kg/year	24
P-sequestration forest biomass	156	kg/year	125	33	kg/year	26
Denitrification	30,365	kg/year	2,247	36,581	kg/year	2,707
Climate regulation						
C-sequestration in soil	579	ton/year	106	410	ton/year	75
C-sequestration in forest biomass	339	ton/year	62	66	ton/year	12
Improvement air quality	3,864	kg PM	139	3,961	kg PM	142
Noise mitigation	NA	dBa		NA	dBa	
Cultural services	1,352,527	Households	12,143	1,352,527	Households	11,152
TOTAL			**17,314**			**15,915**

NA: not enough information available to quantify service

6.4.3. Floodplain restoration

The European Directive on the assessment and management of flood risks requires Member States to assess if water courses and coast lines are at risk from flooding, to map the extent of flood risk and calculate which assets and humans are at risk in these areas and to take adequate and coordinated measures to reduce this flood risk. The Flemish Environment Agency (VMM) is in compliance with the Directive designing flood risk management plans. These plans include a series of measures aimed towards prevention (adapt land use in flood prone areas), preparedness (inform population on flood risks and what to do) and protection (reduce likelihood of floods and flood damage with dyke heightening and floodplain restoration). A prioritization of measures is developed by IMDC (2012) and is based on an extensive impact assessment considering impacts on material damages due to floods, people at risk, cultural heritage and ecosystems. The nature value explorer is specifically used to assess the impact of natural flood management options on other ecosystem services such as water quality and carbon sequestration. This helps to identify win-win solutions, where both flood risk reduction and improved ecosystem service provision can be achieved.

The results of this case study (Table 23) demonstrate the possibility of ecosystem services created by restoring floodplains. The value expresses the impact on ecosystem services by redesigning agricultural land into a floodplain. Although this conversion has negative consequences on sequestration in soils and air quality, denitrification increases largely and causes the total impact to be positive. Other services such as carbon sequestration in forest biomass and cultural services were not considered. As no additional recreational facilities were created, the end user (VMM) expected that cultural services would not be influenced by this project. However, the creation of additional wetlands can also create non-use values. In this sense, the importance of cultural services is underestimated. The impact expressed as annual values proved to be relatively small compared to the impact of reducing the risks for material damage (Table 24). Flood risks for material damages and people were simulated as part of the impact assessment, making use of similar methodologies as discussed in the previous chapter. People at risk is contrary to the previous chapter not depending on a damage function assessing potential victims but is restricted to estimating the amount of people suffering from floods. Cultural heritage was assessed on a qualitative basis and depends on the presence of historical buildings, protected landscapes and archeological sites.

Table 23: Results case study 3 – floodplain restoration

Service	53 ha floodplain restoration: 5 ha wetland, 32 ha grassland, 16 ha shrubs		
	Quantity		Value
Regulating services	**Amount**	**Unit**	**k€/year**
Nutrient retention			
N-sequestration soil	-23	kg/year	-1.7
P-sequestration soil	-2	kg/year	-1.2
N-sequestration forest biomass	NA	kg/year	NA
P-sequestration forest biomass	NA	kg/year	NA
Denitrification	43	kg/year	3.2
Climate regulation			
C-sequestration in soil	0	ton/year	0
C-sequestration in forest biomass	NA	ton/year	NA
Improvement air quality	-4	kg PM	0.02
Noise mitigation	NA	dBa	NA
Cultural services	NA	Households	NA
TOTAL			**1.7**

NA: not applicable for case study

Table 24: Example impact assessment flood risk management plans including impact on ecosystem services (source: IMDC, 2012)

Impact category	**Indicator**	**alt 0: present situation**	**alt 1: preparedness**	**alt 2: prevention**	**alt 3: protection restoration floodplains**	**alt 4: combin. 1-2-3**
Risk material damage	Risk in 2010 (k€/year)	203				
	Risk in 2100 (k€/year)	444	237	190	55	40
People at risk	Risk in 2010 (people/year)	118				
	Risk in 2100 (people/year)	174	87	131	19	19
Cultural heritage	Landscape (-/0/+)	0	0	0	-	-
	Buildings (-/0/+)	0	0	0	-	-
	Archeology (-/0/+)	0	0	0	-	-
Ecosystem	*ecosystem services (k€/year)*	*0*	*0*	*0*	*1.8*	*1.8*

6.5. Discussion

Since its launch in September 2010, approximately 120 users have registered and 200 scenarios have been simulated. As the cases illustrate, environmental NGOs use the tool as a means to help prevent further losses of natural areas. Administrations also explicitly refer to the tool when setting up new cost-benefit analyses related to infrastructure projects. The web application provides an opportunity to anyone who is interested to perform a quick-scan of the relevant ecosystem services for any given study area in the Flanders Region. The availability of a practical tool overcomes the argument that accounting for ecosystem services is too complex, costly and time consuming. The key innovation that the tool provides is that any stakeholder group, policy-maker or NGO can perform its own ecosystem service assessment which can be used in the public inquiry stage of projects or through public protest. Different sets of input data and results from different stakeholder groups can be compared for a specific case study and used to influence decision making. If a quick scan reveals that projects potentially affect specific ecosystem services in a significant way, more advanced and spatially explicit quantification and valuation methods can still be used.

To double-check whether the web application fulfilled the most important user requirements (user-friendliness, transparency, flexibility and scientific reliability), an internet survey was organised 1.5 year after the launch. In general, end-users confirm the usefulness of instruments of this kind. The end-users express the added value in using the tool to include the ecosystem services concept in existing policy appraisal frameworks. The tool supports the recognition of the importance of preserving ecosystems and related services. The tool also allows exploring ecosystem services and the impact of alternative land use strategies. However, some major concerns still need to be addressed in order to maintain and further increase future use.

Balancing scientific reliability against user-friendliness remains the biggest challenge. Issues related to scientific reliability include a too limited list of services covered, the lack of uncertainty analysis and the methodology used to estimate cultural services. Important lacking services according to end-users are related to provisioning services (mainly food and wood production) and regulating services as pollination, erosion and flood prevention. For provisioning services, such as food and fibre production, models and data are available for quantification and valuation. Although these services might already be included elsewhere in a SEA or CBA, users feel it is essential to include them in the tool to create a more comprehensive estimation of trade-offs between services for alternative land use strategies. Services as flood prevention are very area specific and specific models for quantification and valuation for Flanders do exist (see previous chapter), but these are rather complicated and it is impossible to grasp them in straightforward online calculations. For other services such as pollination and water supply, the current state of the art in understanding, modelling and data does not allow to produce figures for quantification and/or valuation that are scientifically well underpinned and relevant for application in Flanders. To make lacking services more explicit, the future aim is to also qualitatively include services for which quantification and valuation data are unavailable. Missing services will be described but cannot be selected for quantification and valuation. If possible, qualitative scoring of the relative importance of a service will be performed. This will mainly be determined by the vegetation type.

Transparency is another user requirement needing significant improvements. As the concept is often used for communication purposes, models and calculations need to be clear, understandable and transparent. Though users are informed by information boxes, a manual and other background information, they still have a feeling to work with a “black box” that in some cases creates results which are difficult to understand. Future efforts to tackle this issue will focus on assessing uncertainty margins (low and high estimates) and reporting intermediate results. Not only end-results will be reported, but also the applied input data and the underlying calculations. Other ongoing research which will help to better grasp uncertainty includes the use of Bayesian Belief Models (Landuyt et al., 2013), setting up monitoring campaigns for representative indicators for specific services and comparing results with the outcomes of more complex off-line models for specific services (e.g. air quality model, water quality model).

Flexibility and user-friendliness were other key features. However, what this means varies highly across end-users. Knowledgeable end-users indicate that they need to be able to select the services they want to include and in some cases change calculation methods. Less experienced end-users require more background information on the type of information that needs to be gathered and want to see predefined values and methodologies as much as possible. For this type of end-users flexibility refers to making the tool applicable for a wider range of application areas (e.g. urban green infrastructures). Increasing end-user interaction is also a key feature contributing simultaneously to user-friendliness and scientific reliability. The objective is to establish a learning system allowing end-users to define case specific input values, pose questions, exchange results and experiences on a discussion forum and improve calculation procedures. Overall, the web application is aimed to provide a platform where the stakeholders of ecosystem services can exchange knowledge and further enhance practical methodologies. These knowledge exchanges are paramount to develop a multi-disciplinary research domain as ecosystem services assessment.

A major discussion point relates to the assessment of cultural services, because the values are based on only one study and tend to dominate results in most case studies. The assessment is based on an integrated approach that estimates the value for different cultural services in one single indicator, based on stated preferences. The scenarios presented in the choice experiment (additional nature development on agricultural land) are not always suitable for benefit transfer to the case studies (loss of existing nature areas or changing vegetation in existing areas). Therefore, further research into two dimensions is required to further improve this aspect of the tool. First, the benefit transfer function should be supported by additional willingness to pay studies, using similar survey instruments (choice experiment). These studies should not only consider nature development on existing agricultural land, but also nature redevelopment or loss of existing natural areas. Second, recreational services should be analysed separately and with different methods, using indicators for number of visits based on data for Flanders. Visits can be valued using different methodologies, including both stated and revealed preferences (travel costs functions) so that the stakeholders can select the value function that they consider most appropriate for their case study. In addition, there is a large set of valuation studies (e.g. Moons et al., 2008; Zandersen and Tol, 2009) available as a basis for benefit transfer, which can be integrated into the tool. A large issue related to the assessment of cultural services is double counting. The applied methodology assumes that people only have a WTP for cultural services (health, recreation, education, non-use value). In more recent valuation studies it was explicitly surveyed which services people consider when a WTP is expressed. Results confirm that a large share of the people also consider other services, especially regulating services when expressing a willingness to pay (Aertsens et al. 2012). Separate valuation of specific cultural services as recreation is currently being developed and will avoid these doublecounting issues. A disadvantage of this approach is that the valuation will not cover all cultural services. Especially the non-use value, which can be significant, is no longer taken into account.

The value function approach is a more accurate methodology to perform Benefits Transfer. For a lot of services, the size of the area is not always the most determining factor for the value estimations. Instead, other parameters as existing pollution levels, soil characteristics, characteristics of land use and population in surrounding areas have a large impact on results, both for the quantification and valuation of ecosystem services. Incorporating more site specific variables in the value estimates is still possible for future research. Whereas the valuation for carbon is not location specific and global estimates are sufficient, this is not the case for nutrients and air quality. As previously demonstrated in chapter 3, marginal abatement costs are very site specific. Depending on the amount of existing water quality issues, the amount of pollution and the extent at which measures are already implemented, large variations in marginal costs were estimated. However, deriving the spatial function units from which marginal costs are representative for specific projects is not straightforward. Upstream-downstream interactions are relevant and might still create a demand for further reductions in nutrient pollution, even though in some areas water quality targets for specific pollutants are already achieved. For air quality, large value differences are estimated on external health costs for PM between rural and urban areas (Michiels et al., 2012). As the nature value explorer is mainly focused on rural areas and more detailed estimations are not being derived with the existing state of the art in air quality modelling and health impact assessments, it makes little sense to derive more area specific external health cost estimations.

6.6. Conclusions

Policy makers are highly interested in practical tools to assess the impact of policy measures (including land use changes) on ecosystems and the services they deliver. This chapter describes the content and experiences with the ‘nature value explorer’, which is a publicly available web application to assess the impact of land use change on ecosystems services in Flanders. The application illustrates the possibilities and limitations of a simple, ready to use assessment tool to provide scientifically based information for decision making and interaction with stakeholders. Even if some of the scientific underpinnings are subject to debate among scientists and the use of simplified models introduces additional uncertainty, it allows non-specialists to get an impression of the relative importance of different ecosystem services.

First case studies show that the values (expressed in €/ha) of ecosystem services differs a lot between different cases, which confirms the need for tools to take local characteristics into account, rather than using average values per ha for each vegetation type. The relative importance of different services also varies between cases, which illustrates the need for using tools that include many services, rather than use only highly specialized tools that only consider a single services. First cases also show that the total net value of ecosystem services delivered by nature areas is important and relevant to take into account in environmental impact assessments. Values are comparable to market prices for agricultural land in rural environments and for built-up areas in urban environments. This confirms a public willingness to preserve or restore nature areas, though market prices for land might indicate the opposite.

6.7. Comparative assessment tool set-up and results

A web application was discussed to explore the quantity and value of ecosystem services. The application allows to estimate the impact of land use and land cover change in case of nature or river restoration projects on regulating and cultural ecosystem services in Flanders, Belgium. Recent reviews werrreee performed by Bagstad et al. (2013) and Nemec et al. (2013) on similar decision-support tools for ecosystem services quantification and valuation. A large range of tools are currently being developed worldwide ranging from qualitative exploratory tools to extensive spatially explicit bio-physical ecosystem service modelling tools to produce trade-off, flow and uncertainty maps. An important trade-off for all tools which will enhance or limit its widespread adoption is the time required to apply it relative to the depth and quality of information it adds to the decision-making process (Bagstad et al., 2013). Estimated person-hours for a case study range between 10 hours for exploratory tools and 800 hours for spatially explicit modelling. The Nature Value Explorer can be considered as an exploratory tool and probably requires approximately 10 personhours to implement a new case study, depending on the amount of service providing units (SPU) end-users distinguish and the additional amount of GIS analyses that is still required. Unlike most of the applications, the tool is mainly end user driven and end users were involved from the beginning of the development. This helps to increase the chances for widespread policy use, but is still no guarantee. Both review papers correctly declare that some complementarity exists between tools. Exploratory assessments could be used as a first broader analytical 'frame' to identify and prioritize important services potentially impacted by specific projects. Spatially explicit mapping and detailed site assessments focusing on the most relevant services can be made in a second stage. The question remains whether this detailed modelling should be performed with other ecosystem service modelling tools (e.g. INVEST, ARIES) or other existing biophysical modelling tools focusing on a single service (e.g. water quantity and quality models, air quality models). Recently, it was decided for the nature value explorer to go for this second option. External experts for specific services (air quality modellers, hydrologists, noise specialists) were requested to review the existing exploratory methodoliges and describe the existing state of the art in Flanders to model specific services and which experts can be contacted by end users to model these services. This prevents a loss of accuracy and model quality on the one hand (extensive ecosystem service models are mostly still a simplification compared to more specific models) and the time investments to build an extensive ecosystem service model in addition to the existing models. Bagstad et al. (2013) confirm that decision makers felt that the time and cost requirements to run extensive ecosystem service models remain too high to be used in widespread decision making, especially as their added value relative to existing environmental assessments remains to be shown in practice.

Both reviews also demonstrated large differences between the methodologies of the existing tools. Differences exist in the type of services which are modelled, the definitions and indicators used to describe these services and the applied methodologies to quantify and value indicators. The suitability of tools is highly context-dependent. In general, cultural services are less well represented compared to provisioning and regulating services. Also the impact on air quality is not often included in other existing ecosystem service assessment tools. Consequently, results from the presented case studies in this chapter are very difficult to compare with other values generated by other ecosystem service decision support tools.

Global databases on ecosystem service values per biome are however available and can be used to get a feeling on the range of the assessments worldwide and how results from the nature value explorer are situated within this range. A recent literature review by De Groot et al. (2012) was used to develop global estimates of the value of ecosystems and their services in monetary units. They distinguished 10 biomes to give an overview of the value of different ecosystem services. In total, over 320 publications were screened covering over 300 case study locations. Approximately 1350 value estimates were coded and stored in a searchable Ecosystem Service Value Database (ESVD). The results of this literature review expressed in monetary value per hectare per biome are used in Table 25 to estimate the range of values for the three case studies in this chapter according to the results of this literature review. The results on the one hand demonstrate the large variability in estimations from literature. Maximum values can be approximately 50 times higher compared to the minimum values, depending on the biome. A comparison of the results of case study 1 and 2 demonstrates that estimations by the nature value explorer are approximately 5 times higher compared to the maximum value. Results of case study 3 are 400 times lower, but this is difficult to compare as the results of this case study only reflect the impact of a small scale land use change, causing both increasing and decreasing values for specific services. Important reasons for these high values are the high population density which has a direct impact on cultural services and the valuation of air quality and noise mitigation and the relatively small size of the areas valued here whereas in literature mostly, larger nature reserves or national parks are valued. As values for cultural services are only partly depend on the size of the area, this leads to higher values expressed per hectare.

An important conclusion that can be drawn from this comparison is also the unreliability of unit value approaches. Benefits Transfer methodologies that only consider the type of biome and the total surface are very difficult to interpret and very unreliable to use for Benefits Transfer. The number of ecosystem services and estimates per biome varies significantly due to limitations in data availability and reliability and makes a comparison less straightforward. The types of services which are included in the literature review include mostly provisioning services, which are not included in the case studies presented in this chapter. Services as improvement of air quality and noise mitigation included for the case study estimations are not included in the review for most biomes. Also, valuation methodologies for the cultural services, which dominate the results for most case studies, are very different. Cultural services estimated in most articles included in this review are limited to recreation and make use of both direct market prices and stated preference techniques. The estimations for the case study are based on one stated preference study and is not limited to recreation, but includes both use and non-use values. For many ecosystem services, especially cultural services, values per hectare are not a good indicator to compare.

Table 25: Comparison results nature value explorer with results literature review De Groot et al. (2012)

€2010/ha.year *	No. of estimates	Unit numbers literature review				Result nature value explorer
		Mean values	Median values	Minimum values	Maximum values	
Results literature review De Groot et al. (2012)						
Inland wetlands	168	20,013	12,884	2,352	81,764	
Temparate forest	58	2,348	878	217	12,785	
Woodlands	21	1,237	1,186	1,070	1,705	
Grasslands	32	2,237	2,102	97	4,621	
*Application unit numbers per hectare on case studies ** *						
Case study 1		57,256	30,225	6,547	263,551	1,380,000
Case study 2 (existing)		586,530	351,640	52,555	2,454,939	17,314,000
Case study 3		221,440	170,471	35,663	702,913	1,700

** Conversion from int$2007 to €2007: 1int$ = 0.7432 € (World bank indicators 2007, GNI per capita, PPP current international $); conversion from price level 2007 to 2010 based on consumer price indexes Belgium 2007 (106.53) and 2010 (113.69) (FOD Economie, 2013)*

*** Unit numbers for woodlands are used for pioneer vegetation in case study 1 and shrubs in case study 3.*

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CHAPTER 7. Web-based decision support to set up cost-effective programs of measures for multiple water aspects

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Abstract

Chapter 7 describes how data and results from hydrological models and economic appraisal techniques as discussed in previous chapters can be made available for end users in a web based decision support system. The objective is to provide synthesized information from monitoring, field studies and models on different scales and water aspects. This allows end users to consult data and information as presented in the previous chapters.

The decision support system builds further on the cost-effectiveness analysis as presented in chapter 2, which was set up in preparation of the first river basin management plan. The public consultation process of this river basin management plans and a user requirements analysis resulted in recommendations for further development of the model. As a result, the scope of the model was expanded with a more extensive analysis of multiple water aspects, such as surface water quality, hydromorphology and sediments. A web-based decision support tool was developed to make the reporting structure more transparent. Information about status, pressures, costs and effects of measures can be retrieved and simulation results can be generated on different scales, from individual waterbodies to regional level. End users can build up draft packages of measures (scenarios), assess their costs and effects and share these scenarios with other users (e.g. users building scenarios for other aspects or for other waterbodies). The tool will be used by the policy makers in Flanders in preparation of the next generation of river basin management plans.

7.1. Introduction

The European Water Framework Directive (European Commission, 2000), adopted in 2000, requires Member Status to meet good status of all waters. To ensure that this goal will be met, member states have to assess the current state of all waters, to identify significant water management issues and to publish river basin management plans to tackle these issues. Decision making in the context of integrated water management can be very complex. To better understand this complexity and improve decision making, decision support systems (DSS) are frequently being developed. Examples of DSS for water management include AQUATOOL for water scarcity issues (Pulido Velazquez et al., 2008), FLUMAGIS for flood management, water and ecological quality issues (Volk et al., 2008) and MULINO (Giupponi, 2007), which focuses less on modelling the consequences of measure but more on participatory processes, problem identification and presenting model outputs. Hirschfeld et al. (2005) present a conceptual DSS which is able to perform a cost-benefit analysis within an interdisciplinary spatial decision support system for an integrated management of the Werra River Basin including surface water quality and river morphology.

Despite the significant effort and large amounts of money spent on developing DSS, there are many that are never or hardly used. Possible reasons are that the users find the system too time consuming and costly to use (Uran and Janssen, 2003). A lack of appropriate and methodological stakeholder interaction and no clear definition of 'what end-users really need and want' have also been documented as general shortcomings (De Kok et al., 2009; Volk et al., 2010). Other reasons are related to the general complexity of the systems, the uncertainty of the model output and the limited appropriateness for solving the decision question. A specific problem of the current DSS developed for the WFD is that they have been developed for quite specific issues and do not cover the transdisciplinary broadness of the WFD in combination with different scales and water aspects (Gourbesville, 2008). Also the issue of cost-effectiveness has so far been often neglected in decision support concerning WFD (Ward, 2007). As a consequence, decisions made in practical river basin management are often not well documented. In particular, it remains unclear how cost-effectiveness has been taken into account by the authorities when selecting measures (Bernd et al., 2012).

The decision support system discussed in this chapter is a tool specifically aimed to support policy makers in developing cost-effective programs of measures for different water aspects as required for the 2nd generation management plans in Flanders, Belgium. The main objective of this system is to synthesize information from monitoring campaigns, modelling tools and economic appraisal techniques to achieve better informed decision making in integrated water management. To make sure the system is easy to use by different end users, a web based tool is developed that builds on datasets covering the entire Flemish region. End user involvement plays a crucial role during the development process. End users have defined the requirements of the system, delivered the necessary input data, tested different prototypes and are currently using the system for various purposes.

7.2. Study area

Flanders is a highly urbanized region with a surface of 13,521 km² and a population of more than 6 million inhabitants. The region is part of two international river basin districts, the Scheldt and the Meuse. The water system mainly consists of lowland rivers with wide valleys and slow flow velocities. Highly industrialized areas are the ports of Antwerp and Ghent. Agriculture is mainly intensive and cultivated land occupies 45% of the area. Pressures on the water system are high. The assessment of the current status in 2009 (Coordinatiecommissie Integraal Waterbeleid, 2009) indicated that a very small amount of surface and groundwaterbodies are in good status. Significant water management issues are surface and groundwater quality (nutrients, chemicals), flooding (sea level rise), sediments (dredging and processing polluted sediments), hydromorphology, restoring natural conditions and droughts (groundwater quantity in specific areas).

The need for additional measures is clear. However, both from a technical and economic side it is very difficult to reach the objectives. From a technical point of view, it is especially difficult to restore rivers in a highly urbanized area and to tackle diffuse pollution and historic pollution stocks present in groundwater and sediments in a short term. From an economic point of view, reaching good water status is very expensive. A large share (60%) of the environmental expenditures by the government is already going to water policy. Also, the financial burden for the different sectors (households, industry, agriculture) related to water increased significantly in the last decade. The drinking water price for households increased by 63% between 2005 and 2011 (see also chapter 1).

These facts and figures indicate that the added value of setting up cost-effective management plans is high and that important attention needs to be given in establishing win-win situations by implementing measures impacting different water aspects simultaneously. The cost-effectiveness analysis for the first generation river basin management plans was based on a mixture of qualitative assessments based on scores for both costs and effects and a quantitative assessment for basic surface water quality parameters as described in chapter 3 and chapter 4. Though results were used for designing the program of measures, several issues were identified in public consultation procedures. A first issue frequently mentioned is related to scaling issues. Measures which are cost-effective on a regional scale are not necessarily cost-effective on a local waterbody scale. Secondly, cost-effectiveness should be considered more broadly across multiple water aspects. Measures which are cost-effective for a specific water aspect might be less cost-effective to realise the good water status in general. A third issue is related to data transparency. End users want to get familiar with the data behind the analysis. Getting a better view on the basic data and its limitations is a crucial step towards better informed decision making. Especially the data both for costs and effects and calculation methods need to be documented extensively.

To tackle these challenges, a web based tool was proposed that looks into multiple water aspects, provides information for multiple scales and gives a clear view on available data and uncertainty. The design of this tool depended significantly on a user requirements analysis. This web based tool builds on databases and methodologies as developed in chapters 3, 4 and 5.

7.3. User requirements

User requirements for a decision support tool were derived during 2009 and 2010 from approximately 20 separate interviews with stakeholders working on integrated water management. More specifically, expert groups, responsible for setting up programs of measures for specific water aspects, and river basin managers, responsible for setting up management plans on local and regional scales were consulted. From these series of interviews a list of user requirements were defined and feedback on this list was gathered and processed to further refine this list.

A first user requirement is to provide information in a structured way in order to contribute to decision making. This includes a representation of the state of the water system, the pressures coming from different economic sectors and the potential impact of measures. Data on measures need to be detailed, include uncertainty margins and include the source of information. Boundary conditions for applying certain measures are also considered as important information.

The economic analysis needs to include a cost-effectiveness analysis. If no quantitative data exist, qualitative information is also considered useful. Marginal cost curves are considered an informative instrument to get a better view on cost-effectiveness in general. Extensive, multi-objective optimization algorithms are less desired by potential end users. Reasons for this are twofold. On the one hand, optimal solutions do not exist in many cases as not enough technical reduction potential exists to realize all targets. Consequently, multi-objective optimization problems cannot be solved or only be solved by reducing targets to the maximum potential, which in the end leads to a selection of all measures and to relatively little insight in the cost-effectiveness of individual measures. On the other hand, a cost-effectiveness analysis has difficulties in dealing with qualitative information as public acceptance and implementation complexity. End users see more added value in scenario development on a trial & error basis, as the amount of potential measures is not very large (< 100) for the local scale most of them are dealing with. The ability to easily compose and exchange scenarios across different water aspects was considered very interesting.

Besides a cost-effectiveness analysis, also a disproportionate cost analysis was considered important as this can be used as a possible motive for exemptions on reaching the good water status. Though widely discussed and explicitly mentioned in the European Water Framework Directive, no widely accepted methodologies exist on how to determine whether costs are disproportionate. In this tool an indicator approach is applied, combining both affordability assessments, benchmarking indicators and a cost-benefit analysis.

Keeping the data up to date is another big challenge. Status and pressures of water systems evolve. Measures are implemented continuously. This means that on frequent points in time (yearly) data need to be updated. Specifically for the next generation management plans, the proposed reference year is 2012. As results from monitoring campaigns were becoming available during the summer of 2013 at best, data needed to be integrated in a very short time frame (months). Also, end users need to be able to provide feedback on the data presented and to put in more accurate information of local circumstances where available.

7.4. Tool structure

The public consultation process of the river basin management plans and a user requirements analysis resulted in recommendations for further development of the web based application involving a more extensive analysis covering multiple water aspects as water quality, hydromorphology and sediments, a more transparent structure reporting information on different scales and for different scenarios.

The web-based decision support tool focuses on three dimensions:

- Functionalities: information about status, pressure, costs and effects of measures and simulation of costs, effects, cost-effectiveness, disproportionate cost analysis
- Scales: information retrieval and simulation results that can vary from individual waterbodies to a regional level (Flanders).
- Water aspects: surface water quality, sediments, hydromorphology

The objective is to let users build up draft packages of measures (scenarios), assess their costs and effects and share these scenarios with other users (e.g. users building scenarios for other aspects or for other waterbodies).

The web application is developed in cooperation with software engineers using JSF (Java Server Faces), a Java-based framework that supports the construction of web applications. The most important Java libraries used are RichFaces, Hibernate and JFreeChart. Also GIS data on the water system, location of sources and measures can be consulted. To make the GIS data available in the web application, GeoServer is used as GIS server. GeoServer implements the OGC standards WMS and WFS, which are standard protocols for serving GIS data over the Internet. To display these maps in the web application, two javascript libraries are used: OpenLayers and GeoExt.

7.5. Functionalities

7.5.1. Spatial scales

The web application can produce results at different spatial scales: the regional scale (Flanders), the river subbasin scale (Nete, Demer, …) and the scale of individual waterbodies. Map functionalities also make it possible to identify the locations of the monitoring campaigns and main emission points (WWTP, industry, CSO). Reasons for producing results at higher spatial scales is that river basin management plans are reported on the scale of a larger river basin and that spatial differentiation of programs of measures is not always feasible from a policy perspective (for instance emission reduction targets for agriculture or wastewater treatment).

Spatial interdependencies are also specifically challenging for water management. Measures installed upstream, also have an impact downstream. Two approaches are possible. Extending the optimization problem to include multiple location constraints is a first option. Another approach is stepwise scenario building where cost-effective measures are first determined in upstream regions and are a starting point of the analysis in downstream regions. For tidal areas this stepwise approach can be the opposite. To determine measures for tidal floods for instance, downstream areas are more likely to be the starting point and effects propagate upstream. If effects propagate both upstream and downstream, stepwise scenario building is no longer possible and more complex hydrological model calculations in combination with optimization algorithms are required. In this tool users have the possibilities to also consider upstream areas within the Flemish region when selecting a programme of measures for a specific water body. Another possibility that helps to identify the importance of upstream areas is to determine target levels for specific water bodies with and without the assumption that upstream targets for good status are reached.

7.5.2. Information on state, pressures and measures

Databases are set up on pressures, state and measures (Table 26). For pressures it is important to know the contribution of the different sources to an environmental issue. Information on measures consists of costs and effects. Effects are expressed as the effectiveness of reducing pressures from a specific source. Costs are investment and operational costs for installing a certain measure. All costs are transferred to discounted annual costs, similar as described in chapters 3, 4 and 5. More details on the data can be found in annex 4.

Table 26: Overview state, pressures and measures for different water aspects

Water aspect	State	Pressures	Measures
Surface water quality	Concentrations BOD, COD, SS, N, P	Households not connected, WWTP, industry point sources, agriculture diffuse sources	Sewage – WWTP Individual treatment households and industry Reducing livestock Manure treatment Erosion prevention
Sediments	Sediment quantity and quality	Point sources suspended soils, erosion losses	Reducing point sources suspended solids Erosion prevention: buffer strips, cover crops, reduced tillage Dredging, sediment traps
Hydro-Morphology	Quality indices	/	Fish stairs, river restoration

7.5.3. Scenario building and impact assessment

End users identified scenario building (different selections of measures) as an important feature during the consultation process. A number of predefined scenarios relate to the basic measures and the program of measures as defined in the 1st river basin management plan. Users can develop, change, share and publish scenarios. Scenarios are mostly used as a starting point to perform simulations on the impact on pressures and cost-effectiveness analysis. The impact on pressures is expressed as reduced emissions, reduced sediment losses or reduced flood risk. Sediment losses were derived from calculations with the WaTEM/SEDEM model (Water and Tillage Erosion Model / Sediment Delivery Model)(Van Rompaey et al., 2001; Verstraeten et al., 2002). For hydromorphology no quantitative indicators are available in Flanders to assess the impact of measures. A qualitative approach was used to demonstrate whether measures impact on specific aspects of hydromorphology (yes/no).

7.5.4. Cost-effectiveness analysis

A straightforward cost-effect ratio calculation is performed for different water aspects. This formulation implies the ranking of measures by average annual costs per unit effectiveness. The calculation can be represented as the calculation of cost-effectiveness ratios R, which are defined as:

$$R = AEC/Effect \qquad (1)$$

where *AEC* is the Annual Equivalent Cost (euros/year) and *Effect* is the quantitative change in the pressure on the resource or the improvement of the state of the environment (Berbel et al., 2011). Measures with the lowest cost-effect ratio are considered the most cost-effective.

The Annual Equivalent Cost (AEC) can be calculated as:

$$AEC = IC \left[\frac{r(1+r)^n}{(1+r)^n - 1} \right] + OC \qquad (2)$$

With IC = investment cost
r = discount rate in %
n = lifespan of measures in years
OC = annual operational cost

The applied effect indicator is described in Table 27. For surface water quality load reductions expressed as kg/year are applied. This can be estimated for 5 individual parameters. For sediments, effects are expressed as m^3 removed/buffered. This applies for erosion reduction, load reduction of suspended solids in waste water treatment, sediment trapping and dredging. For hydromorphology, no quantitative effect indicator is available and consequently, cost-effect ratios cannot be estimated.

Table 27: Effect-indicators cost-effectiveness analysis

Water aspect	Effect-indicator cost-effectiveness analysis
Surface water quality	Load reduction BOD, COD, N, P (kg load)
Hydromorphology	Qualitative impact on status indicators (+/0)
Sediments	Erosion reduction, load reduction suspended solids (m^3 sediments)

A weighting procedure is applied for the different surface water quality parameters to estimate a weighted effect and assess a weighted cost effect ratio *Rw*. This takes into account the required load reduction target *T* for each pollutant *p* and the maximum reduction potential *RP* of all measures for this pollutant *p*. The further emissions have to be reduced to reach the target and the less measures are available to reach this target, the more important the effect for a specific parameter becomes.

$$Rw = AEC \, / \, Weighted\ Effect \qquad (3)$$

With:

$$\text{Weighted Effect} = \sum_{p=1}^{n} \text{Effect}_p \times {}^{T_p}\!/\!_{RP_p} \quad (4)$$

As no water quality model is available for this specific waterbody to derive load reduction targets, the load reduction target T_p is roughly estimated based on the following formula:

T_p (%) = (observed concentration (mg/l) - target concentration (mg/l)) * flow (m^3/s) * 365*24*3600/1000 (5)

7.5.5. Disproportionate cost analysis

Possible motives for exemptions to reach the good water status are natural conditions (it may take time for the conditions necessary to support good ecological status to be restored), technical feasibility (no technical solution is available, it takes longer to fix the problem than there is time available or there is no information on the cause of the problem) and disproportionate costs (CIS, 2008). The practical interpretation of the term “disproportionate costs” remains disputed: under which circumstances can we consider costs as disproportionate or too high? Which methodologies and threshold values can we apply to assess disproportionality? The WFD itself does not provide specific guidance, but leaves it to the Member States to substantiate the concept and develop practical procedures. Ultimately, the judgment on the disproportionality of costs will be a political decision, but objective criteria can be developed to ensure a transparent decision making process (Goerlach & Pielen, 2007). Though widely discussed in guidance documents and review papers, no generally accepted methodologies exist. The WATECO guidance document (WATECO, 2003) describes two potential approaches. A comparison of overall costs and benefits is considered the most suitable approach. Important is however that disproportionality should not begin at the point where measured costs simply exceed quantifiable benefits. The assessment of costs and benefits will have to include qualitative costs and benefits as well as quantitative. Additionally, the margin by which costs exceed benefits should be appreciable and have a high level of confidence. A second approach is affordability or ability to pay. This analysis might need to be disaggregated to the level of separate socio-economic groups and sectors, especially if ability-to-pay is an issue for a particular group within a basin.

The tool makes use of both a cost-benefit analysis and affordability assessments.

7.5.5.1. Affordability

To be able to perform affordability assessments, the total financial burden for each individual sector is estimated (households, industry, agriculture) and compared with affordability indicators. The selection of affordability indicators and critical threshold values for sufficient affordability is typically established in a political decision process (Stemplewski et al., 2008). A principal direction was derived from indicators and thresholds used by supranational organisations like the Organization of Economic Cooperative and Development (OECD) and criteria used in other policy domains as Best Available Techniques Not Entailing Excessive Costs (BATNEEC). These indicators were double-checked with end users. Instead of a single threshold value ranges were applied, distinguishing between affordable, not affordable and intermediate levels for which no straightforward conclusions can be drawn on affordability (Table 28).

A straightforward affordability indicator for households is the impact of water management costs on purchasing power or income. Additional investments in waste water treatment and drinking water supply lead to increasing water prices and this in turn influences purchasing power. An adequate assessment also considers impacts on different social groups, especially on households with low incomes (Stemplewski et al., 2008). A review on affordability indicators for water was performed by Fankhaus and Tepic (2007). Benchmarks used by the World Bank, UK and US governments vary between 2.5 and 5% of total household income/expenditure. Other studies confirm this range. Studies for the WHO declare 4% of the net income could be spent on water services (Atkins and DHV, 2005). The OECD applies typical thresholds of 2% (Klauer et al., 2008). Based on this review, a range of 2 to 5% is applied on the available household income (after taxes). These indicators are evaluated for both average income and low income (10-percentile) households. Similar water consumption levels are assumed for average and low income households, which might be an overestimation for the low income households as water consumption levels are expected to increase for higher household incomes.

To assess the affordability for industrial companies, the reference value approach as described in Dijkmans (2000) and Vrancken et al. (2006) is used. This approach is also applied in Flanders, Belgium to assess whether environmental technologies are Best Available Techniques Not Entailing Excessive Costs. The reference value approach addresses the annual costs of abatement measures relative to turnover, gross profit and added value and the share of total investment costs of abatement measures in the total average investment costs of the past 5 years. Statistics on revenues, profits and added value for all individual companies are available in Bel-First (Bureau Van Dijk, 2013). The total revenues of 3 years (2009-2011) of all companies causing point source emissions monitored by the Environment Agency and that potentially require a further reduction of these emissions, are compared with the costs. Besides the average of all industrial point sources, also individual companies are distinguished as large differences exist concerning required investments on the one hand and economic performance on the other hand. For agriculture the family labour income per annual working unit can be contrasted with the average gross wage of a full time employed worker in another sector, the so called comparable reference income. If the family labour income is smaller than the comparable reference income, the incentive to search employment in another sector is high. The net farm income is more comparable with added value indicators from industrial companies. Similar affordability criteria are assumed for net farm income as for industry.

Table 28: Criteria applied in disproportionate cost analysis (Meynaerts et al., 2009; Vrancken et al., 2006)

Criteria	Proportionate	Intermediate	Disproportionate
Affordability households			
% available income, average	< 2%	2% - 5%	> 5%
% available income, 10% lowest	< 2%	2% - 5%	> 5%
Affordability industry			
% added value	< 2%	2% - 50%	> 50%
% revenue	< 0,5%	0,5% - 5%	> 5%
Affordability agriculture			
% net farm income	< 2%	2% - 50%	> 50%
Family labour income > average gross wage full time worker	Yes	No	no
Costs vs. Benefits			
Cost-benefit ratio	< 1	> 1	

Affordability is very difficult to assess on the scale of an individual waterbody. For industry it is possible to identify individual companies situated inside the waterbody. For agriculture and households this is more difficult. The financial contributions from households to wastewater treatment are mostly independent from the amount of investments made in the specific waterbody they live in. Also for agriculture it is not easy to assess affordability on a waterbody level as farms mostly operate across different waterbodies. To be able to make an evaluation of the level of expenditures and affordability issues for specific waterbodies, benchmarking indicators are applied. Benchmarking indicators express the total level of expenditures or expenditures per sector in a specific waterbody which can be compared with values from other waterbodies. Benchmarking indicators applied in this tool include total yearly expenditures per km watercourse or per ha watershed, total yearly expenditures for the agricultural sector per ha, total yearly expenditures for households per resident and total yearly expenditures for industry per company. Threshold values to assess these indicators are derived from Flemish affordability criteria and downscaled to the waterbody level based on the characteristics of all waterbodies in Flanders (mostly the length of the watercourses or the surface of the waterbody). If a specific level of yearly expenditures would be below the threshold value for all waterbodies, affordability issues would not occur at a more regional level. The threshold values can provide some insights but should not be treated as absolute borders as choices can be made to invest more in a specific waterbody compared to another waterbody.

7.5.5.2. Benefit assessment

The benefit assessment for specific water bodies is limited to the chemical and ecological status of surface water. It is performed via so called top-down and bottom-up approaches (Table 29). The top-down approach is based on a stated preference survey to assess the willingness to pay of reaching a good surface water status. Results from contingent valuation methods are used for the low estimation of the willingness to pay for a good status of surface water. Results from choice experiments are used for the high estimation of the willingness to pay. From the results of the choice experiments value functions can be derived which allow to estimate the impact of specific characteristics (water quality, biodiversity, nature friendly banks) and specific quality improvements (from bad to average, average to good, good to very good) on the willingness to pay. Specific surveys are performed for the subcatchments of the Dender, Demer, Nete and Leie (Liekens et al., 2009; Liekens et al., 2012a-b). Benefits Transfer techniques used to transfer outcomes from the value function to other waterbodies depend on the existing and future status of the waterbody, the amount of households living in the waterbody and the length of the water courses (navigable and category 1 waterways according to the Flemish Hydrographic Atlas).

The bottom up approach is based on ecosystem service valuation whereby every individual activity related to water is valued separately as demonstrated in chapter 6. The two most relevant services for the Flemish context impacted by an improving surface water quality status are recreation and amenity. For recreation the total amount of trips nearby water were derived from survey results. It was specifically asked how many times per year specific activities were performed nearby or on water (Liekens et al., 2012a-b). Water was defined as public water courses, lakes or ponds and did not include the sea or private ponds. The additional value per trip for an improving water status was derived from studies performed in other European countries (UK, France, Netherlands, see Liekens et al., 2008). Trips were divided across all waterbodies based on the number of inhabitants per waterbody. This means it is assumed that recreational use is restricted to nearby water courses. This is a strong simplification as many types of trips (boating, motorized water sports, angling, swimming) are restricted to specific areas and it can be expected that people are willing to travel larger distances to perform these types or trips. Due to the lack of data on recreational facilities and recreational activities for specific water courses, other methodologies to distribute activities are difficult to apply at this moment.

A hedonic pricing study performed by Brouwer et al. (2007) was use to assess the impact on housing values. They determined a positive impact on values of houses situated within 500 metres of water courses and lakes with good water clarity (0.5% value increase per 10cm improvement of visibility). The amount of houses within 500 metres of the water course (navigable and category 1 waterways) is determined specifically for every waterbody. Double counting between the amenity value and benefits for recreation are a risk. People might be willing to pay more for a house nearby clean water because of the possibilities to perform recreational activities. Though we can expect that the majority of the recreational activities on or nearby water do not occur within 500 metres (boating, motorized watersports, swimming), this still might be the case for activities as walking and cycling. This is why the amenity value is not taken into account in the low estimate and fully taken into account in the high estimate.

Table 29: Benefit assessment for good status surface water

Benefit category	Methodology	Source
a) Top-down estimation		
Willingness to pay for nature friendly banks, good water quality, high biodiversity	Surveys: Value function based on contingent valuation method for low estimate, choice experiment for high estimate	Liekens et al. (2009) Bateman et al. (2011) Liekens et al. (2012a and b)
b) Bottom-up individual ecosystem services		
Recreation: walking/cycling/… ; jogging; swimming; angling; nature related activities (e.g. bird watching); recreational boating; motorized water sports; other activities (e.g. meditation, ...)	Number of activities based on surveys on recreational activities nearby water per household; monetary valuation of trips based on benefits transfer	Liekens et al. (2008) Liekens et al. (2012a and b)
Amenity – housing values	Households within 500m nearby water; benefits transfer for impact clean water on housing price	Brouwer et al. (2007)

7.6. Case study of the Mombeek

To demonstrate how the application works, detailed results are reported for the Mombeek, an individual waterbody situated in the south of the province of Limburg (Figure 22). The Mombeek is a small tributary of the Demer and part of the Scheldt river basin. The waterbody covers a total surface of 8,461 ha. Agriculture is the dominating land use (45%). From the approximately 18,000 inhabitants living in the waterbody, 84% is connected to a wastewater treatment plant. The waterbody was selected as one of the priority areas to significantly improve the water status by 2015.

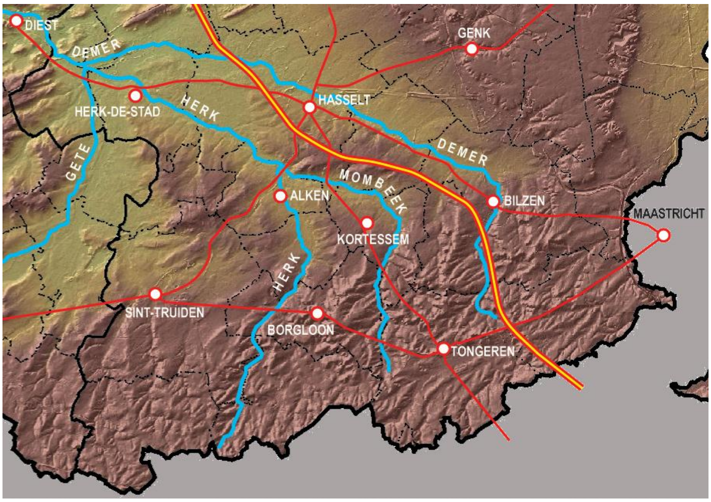

Figure 22: Location of the Mombeek study area (Leenders, 2010)

7.6.1. Surface water quality

Information on surface water quality is available from monitoring campaigns for the years 2006-2012 for all major chemical and biological quality parameters. The observations included in Table 30 for the year 2012 demonstrate that the chemical status is insufficient and that especially observed concentrations for nutrients (total phosphorus and nitrogen) are still far above the water quality objectives. Large load reductions are still required for these parameters (Table 31).

Once the problematic parameters are observed, a next step is to identify the different pressures contributing to insufficient quality. As this is an upstream and mostly rural waterbody, agriculture and non-connected households are the most important known sources for phosphorus (Figure 23). Emissions coming from combined sewer overflows (CSO) are unknown, but are potentially important as a lot of overflows are present in the waterbody (Figure 24). Information on the exact location of these overflows and other emission points is useful for end users to create an improved understanding of the system.

Weighted cost effect ratios indicate a large difference between measures with a much lower ratio in €/kg for agricultural measures (Table 32). This is due to the fact that a larger importance is given to N and P as the load reduction targets for these parameters are large

(Table 31) and the reduction potential of all measures is limited. For P, the reduction potential of all measures is even insufficient to reach the load reduction target.

Table 30: Surface water quality status Mombeek as monitored in 2012

Parameter	Form of assessment	Unit	Observation	Evaluation	Target
Physico chemical status					
One out all out				**Insufficient**	
Suspended solids (SS)	90-percentile	mg/L	54.30	Moderate	<= 50.0
Kjeldahl nitrogen	90-percentile	mg/L	4.10	Good	<= 6.0
Nitrate	90-percentile	mg/L	3.00	Good	<= 10.0
Orthophosphate	Average	mg/L	0.17	Moderate	<= 0.1
Total phosphorus (P)	Summer average	mg/L	0.52	Insufficient	<= 0.14
Total nitrogen (N)	Summer average	mg/L	5.90	Moderate	<= 4.0
Temperature	Maximum	°C	18.00	Very good	<= 25.0
pH	Minimum	pH	7.75	Very good	6.5-8.5
pH	Maximum	pH	8.07	Very good	6.5-8.5
Chloride	90-percentile	mg/L	60.60	Good	<= 120.0
Conductivity	90-percentile	µS/cm	814.70	Moderate	<= 600.0
Sulphate	Average	mg/L	73.58	Good	<= 90.0
Biological Oxygen Demand (BOD)	90-percentile	mg/L	4.75	Good	<= 6.0
Chemical Oxygen Demand (COD)	90-percentile	mg/L	27.50	Good	<= 30.0
Dissolved oxygen (concentration)	10-percentile	mg/L	5.38	Moderate	>= 6.0
Dissolved oxygen (saturation)	Maximum	%	94.00	Very good	70-120
Ecological status					
One out all out				**Bad**	
Phytobentos	Minimum		0.15	Bad	>= 0.6
Macroinvertebrates	Minimum		0.40	Insufficient	>= 0.7
Macrophytes	Minimum		0.35	Insufficient	>= 0.6
Fish	Minimum		0.65	Good	>= 0.6

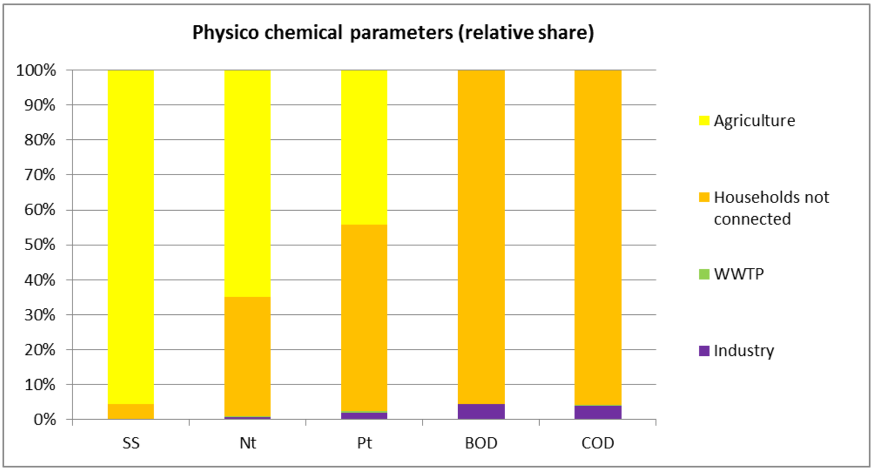

Figure 23: Pressures on physico chemical status of the Mombeek

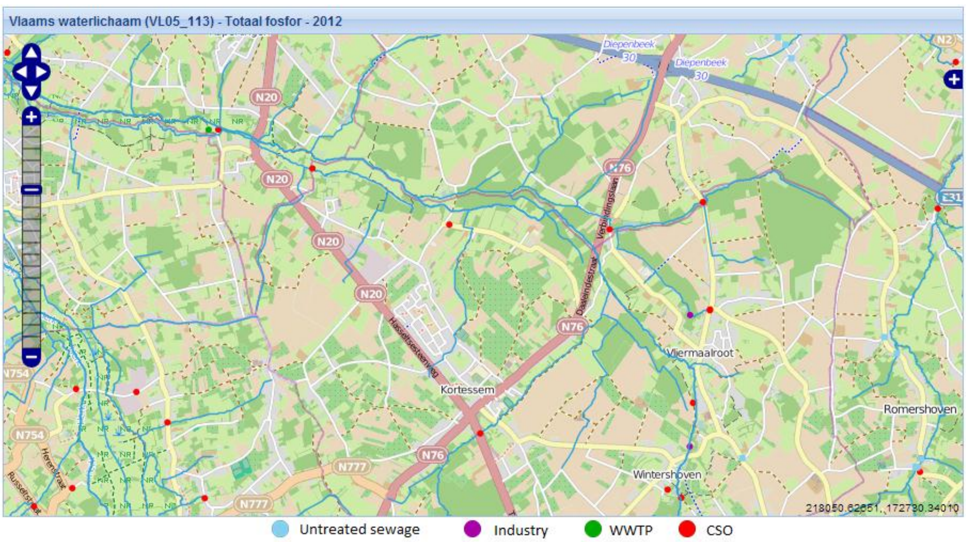

Figure 24: Location of important emission points for the Mombeek

Table 31: Estimation load reduction targets for physico chemical parameters in the Mombeek

Parameter	Observation mg/L	Target mg/L	Flow * m³/s	Pressures kg/year	Load reduction target kg/year	Load reduction target % of pressure
SS	54.3	50	0.17	2,758,179	23,053	1%
N	5.9	4	0.28	105,336	16,777	16%
P	0.52	0.14	0.28	9,699	3,355	34%
BOD	4.75	6	0.17	146,741	0	0%
COD	27.5	30	0.17	342,406	0	0%

** For 90-percentile concentrations we use the 10-percentile flow to estimate the total loads, for the summer average or yearly average the average annual flow is used.*

Table 32: Cost effect ratios for measures to reach a good physico chemical status in the Mombeek

Description	Cost (€/year)	Annual effect (kg/year) SS	N	P	BOD	COD	Weighted cost effect ratio
Reducing manure application to 140 kg N/ha - manure processing	1,013	0	4,423	0	0	0	0.40
Reduced tillage	7,280	706,692	0	213	0	0	0.43
Winter cover crop	3,873	168,705	7,527	68	0	0	0.46
Bufferstrips	15,954	53,228	0	20	0	0	12.46
Tuned fertilisation	24,835	0	3,250	81	0	0	12.60
Extend municipal sewage network - priority 1 projects	259,601	7,783	1,857	266	9,044	19,684	159.19
Extend municipal sewage network - priority 3 projects	1,836,046	53,602	11,979	1,576	62,239	132,375	176.63
Extend municipal sewage network - priority 2 projects	46,671	1,529	261	32	2,120	4,376	201.56
Individual wastewater treatment remote houses	32,351	1,130	95	7	1,673	3,918	355.66

7.6.2. Hydromorphology

The evaluation of hydromorphological quality indices are based on field observations for a large range of parameters performed by the Flemish Environment Agency for trajectories with a length between 200 and 400 metres. An overview of the observed parameters is given in Table 33. Results can be consulted in the application for individual trajectories as demonstrated in Figure 25. The river bed structure for the Mombeek is from moderate to bad quality. Information on the trajectory level is very relevant for hydromorphology in order to identify problem areas where specific measures need to be taken.

The cost-effectiveness analysis for measures aiming to improve the hydromorphological status is very different from surface water quality. As stated in paragraph 2.6, very little examples exist on how to select cost-effective measures improving hydromorphology and ecological status. Publications are mostly limited to ex post evaluations of the effectiveness of measures in improving hydromorphology and ecological status (Jähnig et al., 2010; Lorenz et al., 2009). Quantitative assessments on how the implementation of measures potentially improves hydromorphological quality indices are not available and consequently, a more quantitative cost-effectiveness analysis is not applied. Policy questions for hydromorphology are also different from surface water quality, where choices need to be made between alternative combinations of measures to reduce pollution levels. A specific measure to improve hydromorphology cannot replace another measure in most cases. A more relevant question is where to improve hydromorphological quality by which type of measures given the specific issues identified by field observations. To answer this policy question, a more qualitative approach is developed. Based on an expert consultation it was estimated which type of measure can influence which type of quality index (Table 34). If end users identify a certain problematic index for a specific waterbody (e.g. river bank quality) or smaller trajectory, they can consult which type of measures can be implemented to resolve this issue. Cost estimations for these measures give an idea on the required budgets to improve hydromorphological quality.

Table 33: Hydromorphological quality indices and observed parameters impacting these indices

Index	**Observed parameters**
Profile	River width/depth ratio
	Variation in river width
	Cross section
River bed structure	Sediment banks
	Variation in deep and undeep sections
	Substrate
	Vegetation
	Dead wood
	Presence of shaded areas
River bank structure	Flood defence (dykes, quay walls,…)
	Vegetation
	Trees and shrubs
Flow and flow variation	Peek flow
	Backwater
	Flow variation
	Flow velocity
River continuity (migration barriers)	Longitudinal continuity
	Lateral continuity
Alluvial processes	Meandering
	Erosion
	Flood potential
	Land use in meanders
	Land use in alluvial plain
	Ponds and old meanders

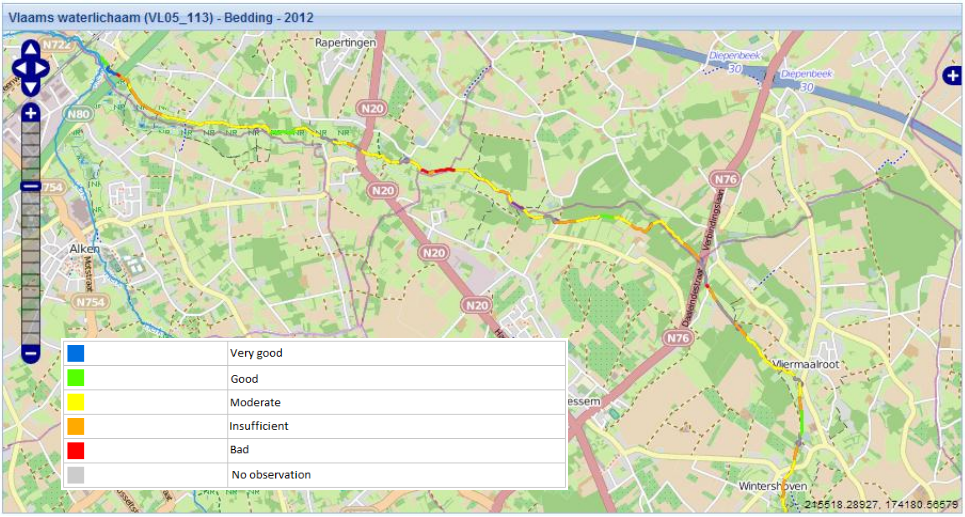

Figure 25: Visualisation of hydromorphological quality (river bed structure) of the Mombeek

Table 34: Qualitative effect assessment of typical hydromorphological restoration measures (+: effect; 0: no effect)

Measure	**P**	**O**	**S**	**B**	**LoC**	**AP**	**LaC**
Removal of invasive waterplants	0	0	+	+	0	0	0
Improving mowing regime of waterplants	0	+	+	+	0	+	0
Improving dredging regime sediments	+	0	+	+	0	+	0
Construction of fish spawning areas	+	+	0	+	0	0	+
Removal fish migration barrier	0	0	0	0	+	0	0
Bufferstrips	+	+	0	0	+	0	+
Removal migration barriers on river banks for terrestrial species	0	+	0	0	+	0	0
Nature friendly river bank restoration	+	+	0	0	+	0	0
Repair or reconnect river meanders	+	0	+	+	0	+	+
Reconstruct winter/summer bed	+	+	+	+	+	+	+
Reconstruct natural flow regimes	+	+	+	0	+	+	0

(P: Profile; O: River bed structure; S: Flow and flow variation; B:River bank structure; LoC - LaC: Longitudinal and lateral River continuity; AP: Alluvial processes)

7.6.3. Sediments

For sediments, both quantity and quality are important water management issues in the Flemish context. Due to erosion losses and emissions from urbanized areas large amounts of (polluted) sediments enter the river system creating issues for shipping, water quantity and quality. Typical measures include erosion prevention, sediment capping and dredging.

To evaluate the status of sediments, both quality and quantity issues are relevant. Monitoring data on quality (triad methodology as described in Chapman (1996) and De Deckere et al. (2011) combining biological, eco-toxicological and chemical quality) and quantity are available for individual observation points within the selected waterbody (Table 35). One out all out evaluations for the entire waterbody have less added value. Evaluations on the individual trajectory or observation point are more informative. For sediment quantity, target levels are not defined.

A pressure analysis can be performed for sediment quantity (Figure 26). The valley of the Mombeek is an area sensitive to erosion. The large majorities of sediments entering the river system are coming from agricultural areas. Non-connected households are another important source. Combined sewer overflows might also be an important source. Though several overflows are present in this waterbody (Figure 24), no methodologies are available to predict the contribution of these overflows.

Sediment quality issues are mostly caused by historical pollution. An important policy question is not where the pollution was coming from, but how pollution levels in sediments influence pollution levels in surface water quality and the potential consequences removal of polluted sediments might have on physico chemical and ecological quality. This is not included in the application, but is a subject of future research.

The cost-effectiveness analysis compares erosion prevention measures as bufferstrips, winter cover crops and reduced tillage with other measures treating point sources (households or industry) (Table 36). The estimated effectiveness of extending the sewage network is derived from the effectiveness of the existing WWTPs for suspended solids. The effectiveness of bufferstrips, winter cover crops and reduced tillage on erosion losses (% removal) is derived from ILVO, 2007 (percentage removal) and applied on the estimated erosion losses for each waterbody from WATEM-SEDEM.

Due to the large contribution of agriculture to sediment losses, erosion prevention measures as reduced tillage, winter cover crops and buffer strips are by far more cost-effective compared to extending the sewage network.

Table 35: Status data on sediments for some observation points in the Mombeek

Parameter	Unit	Observation	Evaluation	Year
Observation point: 45100				
Biological quality	Index 1-4	1	Good	2008
Ecotoxicological quality	Index 1-4	2	Slightly acute impact	2008
Physico chemical quality	Index 1-4	3	Polluted	2008
Global quality	Index 1-4	3	Polluted	2008
Thickness sediment	m	0.3	no target	2006
Observation point: E001452				
Biological quality	Index 1-4	2	Average	1997
Ecotoxicological quality	Index 1-4	2	Slightly acute impact	1997
Physico chemical quality	Index 1-4	2	Slightly polluted	1997
Global quality	Index 1-4	3	Polluted	1997
Thickness sediment	m	0.4	no target	2006
Observation point: E001453				
Biological quality	Index 1-4	3	Bad	1997
Ecotoxicological quality	Index 1-4	1	No acute impact	1997
Physico chemical quality	Index 1-4	1	Not polluted	1997
Global quality	Index 1-4	2	Slightly polluted	1997
Thickness sediment	m	0.5	no target	2006

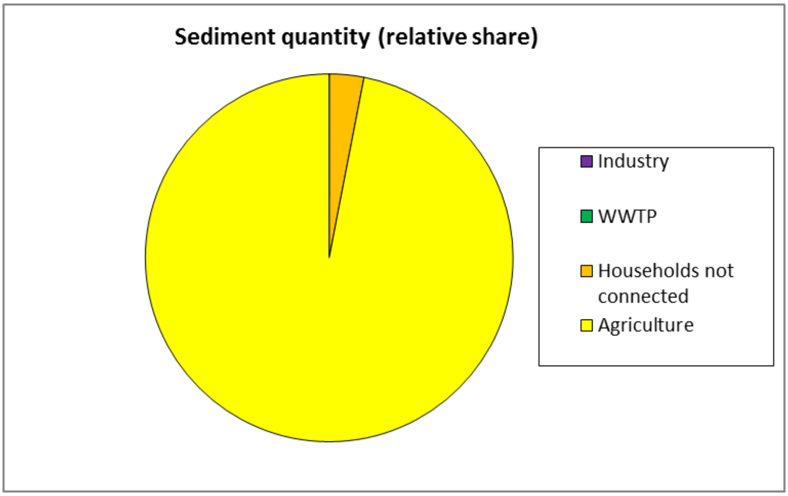

Figure 26: Pressures on sediment quantity of the Mombeek

Table 36: Cost effect ratios for measures to reduce sediment losses in the Mombeek

Description	Annual cost €/year	Annual effect kg/year	Cost effect €/kg
Reduced tillage	7,280	1,067,235	0.007
Winter cover crop	3,873	251,887	0.02
Bufferstrips	15,954	79,243	0.2
Extend municipal sewage network - priority 2 projects	46,671	1,529	31
Extend municipal sewage network - priority 1 projects	259,601	7,783	33
Extend municipal sewage network - priority 3 projects	1,836,046	53,602	34

7.6.4. Disproportionate costs

As stated previously, the disproportionate cost analysis includes an affordability assessment and a cost-benefit analysis. This is demonstrated for a randomly selected scenario of measures that can be taken in the waterbody of the Mombeek and the maximum scenario including the implementation of all measures in the tool database for this waterbody. The first step in the affordability assessment is to translate costs into expenditures for each individual sector (Figure 27). The largest investments in this waterbody are performed by the government and households. In a next step, these expenditures can be compared to benchmarking indicators which are derived from affordability criteria on a regional level (Table 37). The specific values for all indicators show that the costs for selected amount of measures are below the thresholds of the affordability criteria and that the high benefit estimates for realizing the good status exceed the costs (Table 38). For the lowest estimates, benefits are below the estimated costs. However, attention should be given to what is compared. Benefit estimations correspond with reaching a good surface water quality, whereas the selected scenario does not guarantee to realize this goal.

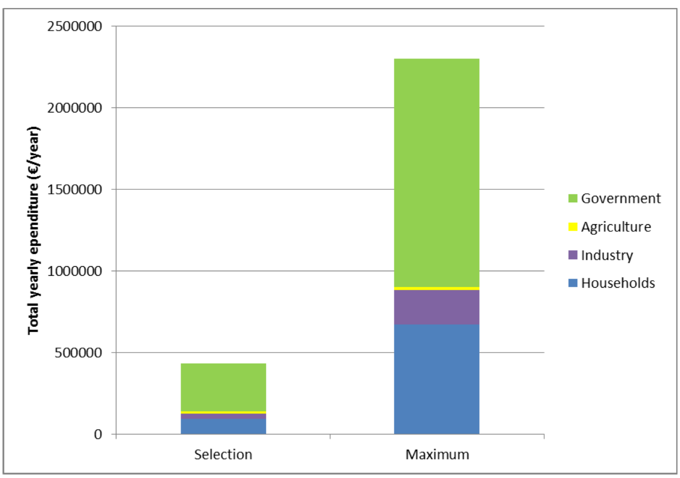

Figure 27: From costs to expenditures per sector for a random selection of measures and the maximum scenario in the Mombeek

Table 37: Evaluation of affordability of programs of measures on the basis of benchmark indicators for a random selection of measures and the maximum scenario in the Mombeek (green is affordable, orange is potentially not affordable, red is not affordable)

Indicator	Selection	Maximum
Total expenditures in € per ha per year	51	272
Total expenditures in € per km of water course per year	4,447	23,709
Total expenditures for agricultural sector in € per ha per year	4	4
Total expenditures for households in € per resident per year	5	37
Average expenditures for industry in € per company	NA	NA

NA: not applicable for case study

Table 38: Estimation annual benefits of realization good surface water status and benefit-cost ratio for random scenario for the Mombeek

	Yearly benefits in k€/year	
Benefit category	**Low estimate**	**High estimate**
a) Top-down estimation (survey willingness to pay)		
- nature friendly banks		894
- water quality		548
- biodiversity		129
Total for good status	*664*	*1,571*
Benefit - cost ratio selection	***1.5***	***3.6***
b) Bottom-up via individual ecosystem services		
- walking/cycling	56	112
- jogging	11	21
- swimming	85	103
- angling	35	35
- nature related activities (e.g. bird watching)	14	27
- boating recreational – non-motorized	5	6
- motorized water sports	0	0
- other activities (e.g. meditation, ...)	30	60
- amenity - housing values		173
Total known benefits	*235*	*537*
Benefit - cost ratio selection	***0.5***	***1.2***

7.7. Conclusions and recommendations

The set-up of a web-based tool to support the development of cost-effective programs of measures in Flanders was discussed. The tool is currently being used to perform a desktop screening of waterbodies, to prioritize which waterbodies to target first and to perform the disproportionate costs analysis for the second generation river basin management plans.

Based on the end user feedback, it can be concluded that creating this application is of added value for decision making in Flemish water policy. The majority of the users consult the application for getting a quick overview on the status and pressures for specific waterbodies. Previous online applications to distribute data on water quality and emissions are considered less user-friendly and require a lot of post-processing activities to derive the necessary information as required for integrated water management. Also how water quality largely depends on activities performed upstream, which was roughly represented by the estimation of load reduction targets with and without the realization of good status targets in the waterbodies upstream, created additional insights. It is apparent that still large steps need to be taken to transfer data into useful information, which can support local decision making.

For many users, it was also the first time to get confronted with specific data issues and the limitations of the hydrological models used by the environment agency. This helped to identify major knowledge gaps and to define additional research activities. Important improvements are still required on monitoring systems. Especially the monitoring frequency and locations are questionable to estimate peak concentrations and calibrate water quality models. For a large number of pressures it is hardly known how important their contribution is to specific types of pollution. The emissions from agriculture are estimated with the SENTWA model which is limited to predicting the short-term losses of nutrients. Methodologies to assess the impact of combined sewer overflows are not available. For sediments, losses coming from urban settlements in upstream areas are not accounted for. Ongoing research activities to tackle some of these issues include the development of an updated emission model for agriculture (ArcNEMO), a water quality model which is better able to predict dynamic behaviour of the river system and a sediment balance and transport model to better explain the origin and behaviour of sediments.

Also to assess the cost-effectiveness of measures, many data lacks were identified. Especially predicting the effectiveness of measures proved to be difficult. Whereas the effectiveness can be predicted reasonably well for wastewater treatment, this is much less the case for agricultural and natural water retention measures. Natural water retention measures as wetland restoration and remeandering are not included in the assessments but have a potential impact for water quality but the experiences are limited to single case studies. Bridging the gap between the experimental field scale and the regional modelling scale as required for the water framework directive will be crucial to better demonstrate the cost-effectiveness of this type of measures. This will also be an important step towards cost-effectiveness analyses across water aspects.

Still large methodological challenges remain for the disproportionate cost analysis. Though the water framework directive requires member states to perform this type of analysis on the level of waterbodies, the methodologies to do so on this scale are limited. Affordability assessments often do not make sense on a local scale as contributions from local sectors mostly do not correspond with investments made in that same area. There is also no scientifically sound basis to define threshold values for affordability. This remains a political decision. A more scientifically sound methodology is to apply a cost-benefit analysis. Applying a cost-benefit analysis for water management on a local scale still involves large challenges. Ideally benefit assessments can be made for every individual measure and for every marginal improvement of the water status. In practice, benefit assessments are much less detailed and are mostly limited to estimating the impact of achieving more generic quality improvements as reaching a good or very good status. Even for these more generic benefit estimations, simplified Benefits Transfer techniques based on number of inhabitants or length of the river system are required. Consequently, composing a limited set of scenarios that are sufficient to reach a good or very good status and comparing the costs for implementing these scenarios with the benefits of reaching a good status is the most obvious procedure to perform.

At this moment, uncertainty is not well represented in this tool and the major weakness. For the disproportionate cost analysis wide ranges are applied on the affordability assessment and low and high estimates are given for the benefit assessment. For the cost-effectiveness analysis this is not the case. Though users have the option to consult all basic data and information on the methodologies, no indication is given on the uncertainty of the cost-effectiveness of measures. As the tool is mainly focused on synthesizing information, uncertainty is mostly caused by the presented monitoring data and model results and less by the calculation procedures incorporated in the tool. The tool is foreseen to include at least low and high estimates for both costs and effects, but the data to include these estimates for all measures are lacking. Qualitative or quantitative uncertainty estimation by experts (Bernd et al., 2012) or sensitivity analysis on cost and effect data, lifespan and discount rate as performed in chapters 4 and 5 are possible options for future development. Another option is performing ex post audits, where predictions on both costs and effects of measures implemented in pilot basins are compared with the observed results.

Acknowledgements

This chapter builds on data that was collected in the framework of studies commissioned by the Flemish Environment Administration (LNE) and the Flemish Environment Agency (VMM).

CHAPTER 8. Conclusions

8.1. Introduction

The overall objective of this research is to develop and apply modelling tools that can assist policy makers to compose cost-effective programmes of measures for water management. Important for these tools is that they are on the one hand applicable for decision making on a national or regional level (macro-scale), but on the other hand also sufficiently detailed to correspond with the choices policy makers are faced with on the local project level (micro-scale).

More specific research objectives are:

- Objective 1: Develop a multi-objective economic optimization model to account for the differences in spatial scales within a cost-effectiveness analysis.

- Objective 2: Integrate an economic optimization model and a hydrological model to assess the cost-effectiveness of measures to reach specific concentration targets

- Objective 3: Develop a risk based approach to select cost-effective programmes of measures instead of optimizing towards predefined environmental targets.

- Objective 4: Develop an ecosystem service valuation framework to take into account the impact of measures on different water aspects simultaneously for the selection of cost-effective programmes of measures

- Objective 5: Develop a web based decision support system that supports policy makers in selecting cost-effective programmes of measures both at micro- and macro-scale.

8.2. Results of this work

8.2.1. Economic optimization model

An economic optimization model (Environmental Costing Model for Flanders, Milieukostenmodel or MKM in Dutch) is developed to select the most cost-effective abatement measures to obtain a given surface water quality target. By means of linear programming techniques the model identifies the least-cost combination of abatement measures to satisfy reduction targets for multiple pollutants at multiple locations. Chapter 3 indicates that it is feasible to construct economic optimization models at the same level of detail as hydrological models. Compared to most existing applications, this set up has the advantage to allow a balanced exchange of scenarios between hydrological models and economic models. The difference in results for individual subbasins also indicates the relevance of dealing with sufficiently detailed data to allow for optimization at local scales (waterbody, subbasin).

The model was specifically built to demonstrate the importance of scale and potential cost savings due to a spatial diversification of the programme of measures. To deal with the differences in scale, typical optimization algorithms as described in Hanley et al., 1998 and van der Veeren and Tol, 2001 are extended with the possibility to include individual measures aimed at reducing individual sources and collective measures that can reduce multiple sources simultaneously. For individual measures decisions are made independently on an individual source level. For collective measures, investment decisions are made on a higher scale level. Costs cannot be divided across all individual sources, influenced by a collective measure, but need to be considered in total for the optimization. Also, emission points can shift between waterbodies due to connection to a wastewater treatment plant. This causes emissions to reduce in one waterbody but to increase in another waterbody, which might cause the need for additional investments in other areas. Results demonstrate that optimization at the local scale to achieve similar effects of programmes of measures defined at the regional scale can provide large cost savings. One third of all costs can be reduced if targets can be realized by locally diversified programmes of measures and if targets can also be achieved in upstream areas.

8.2.2. Integration of economic optimization and surface water quality model

Chapter 4 focuses on the integration of the economic optimization model and a hydrological model. This allows determining the most cost-effective set of reduction measures to reach an in-stream concentration target instead of an emission load reduction target. The framework is based on the coupling of two models: the hydrological water quality model SWAT and an economic optimization model (Environmental Costing Model, ECM). SWAT is used to determine the relationship between the modelled surface water concentration at the river basin outlet and the associated emission reduction. The ECM is used to set up marginal abatement cost curves for nutrients and oxygen demanding substances.

This set-up allows dealing with smaller time scales and daily variations in hydrological conditions in a cost effectiveness analysis. This makes it possible to optimize towards other types of quality parameters (e.g. nitrates instead of total nitrogen) and other types of objectives as summer average or 90 percentile, instead of yearly averages. The impact of reducing point or diffuse source emissions on quality parameters can be very different and this clearly influences the cost-effectiveness of measures. Summer average concentrations are more influenced by reducing point sources (WWTP, households and industry) and peak concentrations, often experienced during winter periods, are more influenced by reducing diffuse sources (agriculture). Consequently, rankings of measures based on cost-effectiveness are very different for both types of objectives.

8.2.3. A cost-benefit analysis for flood risk management

Chapter 5 describes how a cost-benefit analysis instead of a cost-effectiveness analysis can be used to determine cost-effective programmes of measures. CBA addresses the question of whether the objective (or action) is economically worthwhile and finding the efficient level of emissions. This means programmes are no longer optimized towards reaching predefined target levels but risk based approaches are applied whereby specifically for different locations costs of measures are balanced with the benefits they achieve.

This risk based approach is applied for flood risk management on the Scheldt river. Potential measures include a large storm surge barrier, dyke heightening and floodplain restoration. Scenario development (local optimization of measures) is performed by iterative feedback loops between flood risk simulation models and economic analysis. The results indicated the added value of modelling tools, both economic appraisal techniques and hydrological models, for integrated water management. Especially a localized optimization of measures based on actual flood risks instead of fixed safety targets can realize much higher net benefits.

Besides effects on flood protection, the impact on other ecosystem services as are also valued as part of the cost-benefit analysis. The valuation of these additional services demonstrates the potential added value of constructing reduced tidal areas instead of traditional flood control areas. This can have a positive impact on water quality, sediment management, recreation and climate change regulation.

8.2.4. Project specific valuation of ecosystem services

The ecosystem services approach as applied in the previous chapter provides an interesting framework to list and value different types of benefits. However, still large methodological challenges remain to apply this framework for the selection of a cost-effective programme of measures. Whereas most applications on ecosystem service valuation are aimed towards large scale ecosystem service accounting exercises (value of ecosystem services in an entire country or region), new exploratory methodologies were developed that can be linked to the decisions we are faced with in integrated land and water management. This allows assessing and valuing the impact of individual measures as floodplain restoration on regulating services (carbon sequestration, water quality) and cultural services.

To use this framework for location specific optimization of measures, it is crucial to take into account the major characteristics of project sites influencing the value of the services they deliver. In the previous chapter unit value Benefits Transfer techniques are applied. More specifically the value of an ecosystem service at the project site is estimated by multiplying a mean unit value estimated at another study site with the size of the area. This is sufficiently detailed to demonstrate the added value of reduced tidal areas compared to traditional flood control areas, but is insufficiently detailed to determine where the construction of a reduced tidal area would generate the highest benefits.

In chapter 6 more characteristics of the project site (e.g. soil, surrounding land use, population living nearby, sociodemographic factors of the beneficiaries) are taken into account to transfer values from other study sites. The results of three case studies demonstrate the importance of taking site specific characteristics (other than size and land use type) into account for the valuation of ecosystem services.

8.2.5. Decision support with a web-based application

Chapter 7 describes how data and results from hydrological models and economic appraisal techniques as discussed in previous chapters can be made available for end users in a web based decision support system. The objective is to provide synthesized information from monitoring, field studies and models on different scales and water aspects. It allows end users to consult data from water monitoring campaigns and results from modelling tools as developed in the previous chapters.

The results indicate that a more extensive analysis of multiple water aspects simultaneously, such as surface water quality, hydromorphology and sediments is of added value to policy makers. The web-based decision support tool includes all the necessary data to assess costs, effects, benefits and affordability of packages of measures. Information about status, pressures, costs and effects of measures can be retrieved and simulation results can be generated on different scales, from individual waterbodies to regional level.

This tool is currently being used to perform a desktop screening of waterbodies, to prioritize which waterbodies to target first and to perform a disproportionate costs analysis for the second generation river basin management plans.

8.3. Lessons learned

8.3.1. Added value of economic appraisal in integrated water management

The European Water Framework Directive was an important stepping stone in realizing more integrated water management in the European Union. Since its adoption in 2000, an enormous amount of policy effort and research is spent to find out how we can realize a good water status. These efforts also made clear the big challenges European Member States are facing. The WFD established the objective to achieve good water status by 2015. This deadline is nearby. Based on the first version of river basin management plans published in 2009 it can be concluded that this objective will not be achieved in many European Member States. In Flanders, Belgium the situation is even more difficult. It was projected that in none of the approximately 250 waterbodies a good water status will be achieved by 2015.

The WFD also explicitly integrated economic appraisal techniques as an important management tool and stepping stone towards a good ecological status. The amount of research and scientific papers in this field increased significantly during the last decade. A general consensus exists on the potential added value of these tools now and in the future. Policy makers today are simultaneously confronted with decreasing budgets on the one hand and increasing demands for environmental quality on the other hand. Because a lot of efforts to improve our water status were already made in the past, additional measures to go further are becoming increasingly expensive and increasingly less effective (the “quick wins” are already implemented). Therefore, systematic approaches to assess cost-effectiveness supported by hydrological models have the potential to provide additional insights.

Creating win-win situations will be crucial in the future. Implementing measures having an impact on multiple water aspects simultaneously and other environmental domains as climate change and air quality will be a likely course of action. The recent blueprint performed by the

European Commission to provide a state of play on integrated water management confirms this line of thinking (European Commission, 2012). Whereas previous investments mainly focused on technical, engineering type of measures as wastewater treatment, sewage and dyke heightening, future focus is put more on natural water retention measures as buffer strips, river and wetland restoration. It is only by performing integrated land and water management that we are able to reach a good ecological status.

This poses however additional challenges to economic appraisal techniques. A more holistic approach is required taking into account the complexity of the water system (e.g. upstream-downstream effects, surface water – groundwater interactions, impact of climate change, land and water interactions, short term vs. long term impacts). This requires more comprehensive and complex modelling exercises, large datasets and more extensive economic appraisal techniques. However, models and appraisal techniques are only of use if they are transparent, easy to understand and are able to provide new insights for decision making (see also Claassen, 2007). This is not straightforward if they become too complex. Finding the right balance between accuracy, comprehensiveness and transparency is difficult to achieve. The tools and decision support software presented in this PhD contribute to this purpose.

8.3.2. Choosing between economic appraisal techniques

In this research we looked at two types of economic appraisal techniques: the cost-effectiveness analysis and the cost-benefit analysis. Both techniques are applied in the different chapters.

Cost-effectiveness analysis (CEA) is a technique for identifying the least-cost option for meeting a specific physical objective/outcome (Balana et al., 2011). In the context of water management, the purpose of a cost-effectiveness analysis is to find out how predetermined targets, for example pollutant loads in a waterbody, river basin or estuary, can be achieved at minimal costs (Lise and van der Veeren, 2002). It can also be used as an appraisal technique for assessing and ranking the relative performance of different measures or combination of measures on the basis of their costs and their effectiveness. A cost-effectiveness analysis allows for a ranking of large sets of potential alternative solutions. By calculating and ranking marginal abatement costs, it can easily be determined which measures are the most cost-effective. However, when multiple effect indicators are considered simultaneously as is the case for integrated water management, either expert based or stakeholder based weighting procedure can be applied to translate different effect indicators to a single aggregated effect indicator (see also chapter 7) or make use of multi-objective optimization algorithms that are able to minimize costs to reach different targets simultaneously (see also chapter 3). An important boundary condition for this last option is that targets need to exist for all impacts considered. Optimized solutions can only be calculated when specific targets are defined and a sufficient amount of measures is available to reach these targets. As shown previously in chapter 3, both conditions are not always fulfilled in the case of integrated water management. For water aspects as water quantity (floods, droughts), sediment management and hydromorphology specific targets are mostly not available. Typical measures as public and individual wastewater treatment in combination with reducing manure application levels in agriculture are often not sufficient to reach the water quality targets in all areas.

Also, it is important that sufficient measures are available for all impacts considered. If other environmental impacts as climate change and carbon sequestration are considered, targets are available (national CO_2 emission reduction targets). However, to use these targets directly in optimization procedures, this also means that other types of measures (e.g. reducing emissions from energy production or transportation) need to be included to reach these targets. This would make the optimization extremely extensive and complex. Another disadvantage for the multi-objective optimization is that it is not possible to set up a ranking of all measures. The optimal solution is generated without giving details on the ranking within this solution. In general, policy makers want to know what to do first, second, etc. in the upcoming years to reach in the end the good ecological status. This can be determined by comparing solutions from a series of optimization runs while gradually setting targets more strict between model runs. Another option is performing more sophisticated combinations of model runs, where the model is constrained for the targets for which quantified minima exist and optimizes for the other targets. Also performing sensitivity analyses where for different target levels of one parameter, rankings are set up for other parameters could provide additional insights for this type of complex rankings. However, in the end it is still up to the expert to determine how different targets vary between model runs and which combination of target levels are used for the final ranking. This is implicitly equal to performing a kind of weighting procedure.

Cost-benefit analysis (CBA) is a technique that is used to estimate and sum up (in present value terms) the future flows of benefits and costs of society's resource allocation decisions or policy alternatives to establish the worthiness of undertaking the stipulated activity or alternative, and inform the decision maker about economic efficiency (Balana et al., 2011). A CBA goes further than a CEA, in a sense that we do not start from a pre-defined target level of environmental pollution. Instead, the objective is to estimate the efficient level of pollution or emissions. In a CEA the objective is fixed exogenously whereas in a CBA it is endogenously determined in the optimization. The advantage is that targets do not need to be pre-defined, which often causes difficulties in a CEA. The disadvantage of a CBA is that all environmental effects (reduced damages) or benefits need to be valued in monetary terms to allow for a comparison with the costs. A lot of critique is given on monetary valuation of environmental goods and services. A wide range of scientific papers are available disputing the robustness of the methodologies such as value transfer methodologies where monetary value estimates are taken to hold for other times, places, and ecosystems. Another weakness often discussed is the fact that environmental economics only works for marginal changes and not for “once-and-for-all” circumstances (non-linear ecosystem responses, tipping points). A third disadvantage often stated is the fact that monetisation is insufficiently capable in prioritising human needs, in particular those of the poor and this can result in serious social and environmental inequity (Vatn, 2010; Cornell, 2011; Spangenberg, 2012). Most of these arguments are indeed valid, but the suggested alternatives such as a multi-criteria analysis (which is often suggested by these same authors as alternative) suffer from the same difficulties. Outcomes of a multi-criteria analysis are equally influenced by choices made by the researchers (e.g. criteria selection and weighting individual criteria) or at best a limited group of stakeholders. Monetary valuation of impacts certainly has its downsides but it allows broadening the scope of the analysis and assessing the efficiency of measures across water aspects and other environmental domains. Additionally, incorporating the ecosystem services concept in a cost-benefit analysis, as applied in chapters 5 and 6, provides a systematic framework that helps to decide which impacts are relevant to include and how these services can be quantified and valued. Importantly, it also helps to identify which impacts are not included in a cost-benefit analysis (if valuation methods are not available) but are nevertheless important to consider in decision making processes.

8.3.3. A combination of economic appraisal techniques and hydrological models

For integrated water management it is not straightforward to determine the most cost-effective courses of action, as multiple causes of problems can be solved by multiple possible solutions. Furthermore, the effectiveness of solutions can be dynamic in time and space. Models can support decision making by providing information on the current state (what is the issue, how important are specific pressures), estimating the potential effectiveness of additional measures (how can the state be influenced, now and in the future) and comparing/optimizing the cost-effectiveness between scenarios (how can we reach targets at the lowest cost achievable or with the highest net benefits). Ideally, a combination of both economic appraisal techniques and hydrological models is applied.

However, specific issues related to water such as upstream-downstream effects, time dynamics and the large difference in local circumstances makes the integration of economic and hydrological modelling for water specifically challenging. Whereas economic models usually build on administrative boundaries as regions or countries, hydrological models apply waterbodies, watersheds and basins as the geographical unit (Brouwer and Hofkes, 2008). To be able to interact and perform a balanced integration, economic models need to be more detailed compared to typical economic models to assess cost-effectiveness for climate change or energy. Preferably they build on the same datasets on pressures (emissions) as used by the hydrological models. Making use of detailed datasets on pressures also allows incorporating differences in local circumstances that influence both costs and effectiveness of individual measures.

In this research different methodologies are presented on the integration of economic appraisal techniques and hydrological models. Throughout the thesis, a **modular approach**, where different types of models are developed and run separately has been advocated. The actual integration of both types of models into one single model or the so called holistic approach is not performed because of its complexity to calculate and interpret results and because this research was mostly performed for government agencies where existing hydrological models were readily available.

Throughout the research, different approaches are applied to deal with the complexities of water. Upstream-downstream effects are considered by two approaches. A first approach is multi-objective optimization where cost-effective solutions are determined with linear programming techniques to reach multiple objectives simultaneously in up- and downstream areas. A second approach is stepwise scenario building where the optimal solution for areas upstream is the starting point for optimization downstream (or vice versa for tidal areas). Both approaches work, however from a pragmatic and policy oriented point of view stepwise scenario building is more obvious to perform, especially if decision makers have the intention to perform this themselves in online, easy-to-use web applications.

8.3.4. The role of the ecosystem services concept

Ecosystem services are a multitude of services (resources and processes) that are supplied by natural ecosystems towards the society. The economic valuation of ecosystem services presents a promising tool to highlight the relevance of ecosystem services to the society and the economy, and to serve as an element in policy development. The concept is particularly useful to assess the impact of multi-purpose and more natural water retention measures. By quantifying and valuing the different services that these measures deliver, a better view can be obtained on the wider impact of measures instead of the impact on a single environmental issue. As integrated water management is moving away from typical engineering type of measures towards more natural water retention measures, including this concept in economic appraisal techniques is becoming more important to make more balanced comparisons between scenarios.

Research on ecosystem services is rapidly developing. In chapter 6 we present the methodologies and a tool that can be used to quantify and value the impact of land and water management on ecosystem services. The framework proves to be interesting and raises a lot of interest for potential end users. However, most ecosystem service studies focus on large scale land use changes and this will be insufficient to be of added value for water management. More targeted methodologies for both monitoring and modelling are required to estimate the impact of natural water retention measures on ecosystem services. Examples are the multiple impacts of buffer strips, wetlands, river restoration on water quality, floods, droughts-water retention, climate change (C sequestration) and air quality.

8.3.5. Which measures are cost-effective?

Throughout the chapters several rankings of measures are presented for specific cases (Flanders, Grote Nete, Scheldt estuary) and specific water aspects (surface water quality – nutrients, COD; flood prevention). Though some chapters are based on outdated model results, the general findings on cost-effectiveness analysis are still valid today. In general, land based measures as floodplain restoration, winter cover crops, tuned fertilisation and bufferstrips in combination with some technical measures (WWTP, targeted dyke heightening) are considered the best choices to make. Engineering type of measures as dykes, a storm surge barrier, sewage for more remote houses and individual wastewater treatment for households are not appearing to be the most cost-effective solution. This confirms the idea that the further we need to go, the more important more land-based and natural water retention measures are becoming.

This does not necessarily mean that the Flemish Region has made the wrong decisions in the past. Driven by specific legislation as the Nitrates Directive and the Urban Wastewater Directive, both adopted in 1991, obvious choices were made since then as building collective sewage and wastewater treatment and restricting nutrient losses on agricultural land by performing manure processing, reducing livestock, reducing nutrient content in fodder and increasing crop productivity. However, more and more the feasible reduction potential we can expect from these measures is being reached. The further the Flemish Region has to go, the less cost-effective these measures are becoming. An obvious example is treating household emissions. Building individual wastewater treatment or extending sewage networks for more remote houses is very costly and can potentially be avoided by applying other types of measures.

8.3.6. End user involvement

This research was largely based on policy oriented projects and financed by different environment and water agencies unlike many other doctoral researches. This means that this research was less fundamental but concentrated more on the construction of practically applicable tools for end users.

The involvement of end users was important during this research and also influenced the scope. The type of modelling tools applied during this research evolved quite drastically during this research. In the beginning, the focus was more oriented towards more complex economic optimization models. In the end, the focus went more on building less complex but more transparent and easy-to-use web applications. Providing accessibility of end users to information and hands-on simulation tools became more the focus of this research. This was largely inspired by end user consultations. During different projects it became more and more clear that the most powerful aspects of the integration of economic appraisal techniques and hydrological models is not the modelling part as such, but the methodologies applied to structure and integrate data on environmental quality, emissions, model simulations and measures. How data is brought together and translated into easy-to-use information, which suits the needs of end users, is one of the major achievements across this PhD.

Making data publically available through web applications has added value compared to typical desktop modelling. It makes it possible for a larger amount of people to check data quality and possibly improve it. This does not only improve the trustworthiness of models but also can lead to drastic improvements and efficiency gains. On the other hand, a danger appears in reducing information in easy accessible, single numbers, creating a loss of information. Sufficient care should be given to representation and documentation of data and calculation procedures, acknowledging and communicating implicit decisions and assumptions behind the tool to end users (Voinov et al., 2014).

Building web applications requires additional time investments. To be sure that these investments have an added value, it needs to be clear from the start that a sufficiently large amount of potential users exist and can be identified. Developing web applications instead of desktop models also adds restrictions on complexity, especially if end users need to perform model simulations. Models need to be sufficiently simplified and easy to operate to allow end users to perform so called “on the fly” calculations. Limitations on calculation time and model complexity are much more extensive compared to desktop modelling, e.g. the use of optimization algorithms in web based applications is difficult. If models are too complex or require too high calculation times, possible solutions are to limit applications to consultation of pre-calculated model results or to offer the possibility to schedule model simulations, where end users request simulations and receive results in a later stage.

8.4. Future research needs

In this research different challenges in applying economic appraisal techniques for integrated water management were covered. Specific challenges include the integration of hydrological and economic models, the consideration of different spatial and temporal scales which are simultaneously relevant for integrated water management and taking into account the effectiveness for multiple water aspects simultaneously in assessing the cost-effectiveness.

Though several methodologies to tackle these challenges were provided, not all issues were tackled. Differences in temporal scales have hardly been dealt with in this research. It is clear that reaching good water status will be very difficult and may require decades to realize. Some aspects as groundwater quality and the interaction between sediment quality, groundwater quality and surface water quality take a lot of years or even decades to improve. Consequently, for some measures that are implemented now, it can take several decades to see their full impact. Dynamic hydrological models that are able to predict long term impacts are required to answer this question. How cost-effectiveness is influenced when also long term impacts are considered is an important future research question and might change the rankings of measures derived from static models.

Also the integration of multiple water aspects in a cost-effectiveness analysis requires future research. This research focused mainly on improving surface water quality and preventing flood risks. These water aspects are also dealt with the most in scientific literature. Also hydromorphology and sediments are handled in chapter 7, but the knowledge available on status, pressures and impact of measures (hydrological models) on these aspects is much less advanced compared to floods and surface water quality, which makes it difficult to apply economic appraisal techniques. Other aspects requiring further attention are groundwater quality, ecological quality and water scarcity issues, which are possibly becoming more relevant for Flanders due to climate change.

Another issue which is less focused on is uncertainty and model evaluation of the tools presented in this thesis. Some potential techniques listed in Laniak et al. (2013) to perform model evaluation are sensitivity analysis, the use of alternative models, uncertainty quantification and ex post-audits. Sensitivity analysis is performed in chapters 4 and 5. The impact of different assumptions in lifespan, discounting rates, long term economic growth scenarios and sea level rise on the cost-effectiveness of measures did not significantly influence the ranking of measures. The literature review on ecosystem service valuation results clearly demonstrated that especially in this type of research uncertainties are very large. The use of alternative modelling techniques as Bayesian belief Networks (Barton et al., 2008; Landuyt et al., 2013), which are able to provide more insight on uncertainties and how uncertainties propagate through conceptual model chains, will create more insights on the robustness of the models and the variables which are mostly influencing this uncertainty. Performing expert reviews and comparing results of conceptual ecosystem service modelling with the outcomes of more complex process based models for specific services (e.g. air quality model, water quality model) is another potential step to test the robustness of the presented decision support systems. Ex post audits, whereby model predictions are compared to the observed results after implementation requires investments in monitoring and can also be very difficult because of long delays in observing impacts. However, the potential expenditures related to water management require very large budgets (several billions of €) and take long timespans to implement (decades). A systematic monitoring of both costs and effects of pilot experiments, comparing these results with model predictions and improving datasets and models if required are important research activities which will earn back its investment.

A large part of the uncertainty discussed above is related to the availability of data on costs and effects of measures. One of the major drawbacks experienced during this research is the availability of data, especially for less conventional measures. By now, the effectiveness and costs of typical measures as public wastewater treatment, dykes and storm surge basins can be reasonably predicted but it is becoming more and more apparent that these measures are not sufficient to reach the good water status or are not necessarily the most cost-effective courses of future action. More land-based and natural water retention measures will very likely be required in the future. However, knowledge on costs but especially effectiveness of this type of measures is scarce or often limited to very small scale field experiments. Consequently, this type of measures is often not included as potential options in economic appraisal techniques or hydrological models. This causes existing tools to re-confirm the typical conventional choices made in the past. Bridging the gap between technology development, field testing and economic appraisal techniques/hydrological models is an important future research need. The ambition should be to develop methodologies or guidelines on how knowledge from field experiments can be captured better in modelling environments and economic appraisal techniques or vice versa, to design field experiments based on data requirements posed by modelling exercises aimed to answer specific end user questions. Cooperation between administrations, universities, research institutes and other stakeholders will be crucial to realize this ambition.

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Annex 1. Detailed description of data economic optimization model chapter 3

Table 39: Data sources emissions

Source	Data	Geographical scale in model
Industry	Measurements performed by Flemish Environment Agency for 1479 individual companies in 2006.	Individual company
WWTP	Measurements incoming and outgoing loads for 246 individual stations performed by Flemish Environment Agency in 2006	Individual WWTP
Households not connected	Inhabitants not connected to a WWTP as inventoried by the Flemish Environment Agency in 2006. Emission factors per inhabitant as applied by Flemish Environment Agency (N: 9.7 g/day/IE, P: 1.4 g/day/IE, COD: 89 g/day/IE.	Households are grouped according to specific sewage construction projects or at the municipality level (7,350 sources)
Agriculture diffuse	Diffuse emissions estimated by SENTWA model. (System for the Evaluation of Nutrient Transport to Water, developed by Nolte et al. (1991) and adapted by Pauwelyn et al. (1997) and Verlinden et al. (2002))	Waterbody level (265 sources)

Table 40: Total emissions per sector (Flanders)

Sector	Emissions (tonnes/year)			Emissions (% of total)		
	COD	N	P	COD	N	P
Households	113,982	14,155	1,867	65%	37%	49%
Industry	33,769	5,024	577	19%	13%	15%
Agriculture	27,805	18,785	1,367	16%	49%	36%

Table 41: Data sources costs and effects of measures

Source	Measure	Data sources for costs and effects
Industry	Maximum concentration targets BAT	VITO and Resource Analysis (2006); Table 4
	Maximum concentration targets Urban Wastewater Directive	VITO and Resource Analysis (2006); Table 4
WWTP	Construction new or renovate existing WWTP > 2,000 IE to reach efficiency targets Urban Wastewater Directive	Project specific cost estimates from investment program Flanders collective sewage and treatment. Aquafin, 2006
	Construction new or renovate existing WWTP < 2,000 IE to reach efficiency targets Urban Wastewater Directive	Project specific cost estimates from investment program Flanders collective sewage and treatment. Aquafin, 2006; effectiveness based on effectiveness existing wastewater treatment plants and legal targets; Table 5
Households not treated by a WWTP	Connecting sewage to existing treatment with new collectors (projects planned before or during 2006)	Project specific cost estimates from investment program Flanders collective sewage and treatment. Aquafin, 2006; effectiveness based on effectiveness existing wastewater treatment plants and legal targets; Table 5
	Connecting sewage to existing treatment with new collectors (projects planned after 2006)	Project specific cost estimates from investment program Flanders collective sewage and treatment. Aquafin, 2006; effectiveness based on effectiveness existing wastewater treatment plants and legal targets; Table 5
	Extending the sewage system	Cost estimates based on project specific estimates on length of sewage (VMM zoneringsplannen; VITO and Resource Analysis, 2006) and unit costs Aquafin (2005)
	Small scale individual treatment for remote houses	VITO and Resource Analysis (2006)
Agriculture	Existing nutrient legislation ((including EU Nitrates Directive derogation)	Cost and effects from ILVO (2007); reduced nutrient losses simulated by SENTWA.
	Livestock reduction (poultry and other livestock)	
	Increased dairy cattle efficiency	
	Increased feed efficiency (pigs and poultry)	
	More strict nutrient legislation (exclusion of Nitrates Directive derogation)	
	Tuned fertilisation (only up to crop requirements)	
	Buffer strips along watercourses	
	Reduced tillage	
	Winter cover crops	

Table 42: Costs and effects of supplementary measures (Flanders)

Source	Measure	Annual cost (5% discount rate)	Reduction emissions (kg/year)		
			COD	N	P
Industry	Maximum concentration targets Urban Wastewater Directive	15,889,915	1,796,338	255,968	31,560
WWTP	Construction new or renovate existing WWTP < 2000 IE to reach efficiency targets Urban Wastewater Directive	20,367,939	2,949,339	1,213,284	65,793
Households not treated by a WWTP	Connecting sewage to existing treatment with new collectors (projects planned after 2006)	22,792,737	3,356,399	308,812	46,410
	Extending the sewage system (grouped according to cost compared with small scale individual treatment households: cost sewage < cost individual treatment (low-cost sewage)	101,954,332	11,125,176	1,057,884	158,953
	Extending the sewage system (grouped according to cost compared with small scale individual treatment households: ccost sewage < 2 x cost individual treatment (medium-cost sewage)	135,193,568	8,995,269	869,291	132,021
	Extending the sewage system (grouped according to cost compared with small scale individual treatment households: cost sewage > 2 x cost individual treatment (high-cost sewage)	67,769,851	1,962,220	188,121	28,719
	Small scale individual treatment for remote houses	52,235,279	3,636,763	88,081	6,356
Agriculture	Livestock reduction (pigs)	43,357,549		243,500	23,065
	Livestock reduction (cattle)	9,939,806		59,793	3,754
	Livestock reduction (poultry)	1,216,683		24,998	9,184
	Increased dairy cattle efficiency	-13,972,485		358,437	18,797
	Increased feed efficiency (pigs)	6,643,626		474,816	43,298
	Increased feed efficiency (poultry)	23,979,428		125,544	2,987
	More strict nutrient legislation (exclusion of Nitrates Directive derogation)	62,463,505		244,968	0
	Tuned fertilisation (only up to crop requirements)	29,656,574		866,154	11,349
	Buffer strips along watercourses	9,071,836		0	12,344
	Reduced tillage	2,341,020		0	11,091
	Winter cover crops	5,258,494		1,821,978	11,573

** Effects are compared with the baseline scenario and do not include interaction effects between individual measures*

Annex 2. River network and calibration results Pégase water quality model (Aquapole, 2008) chapter 3

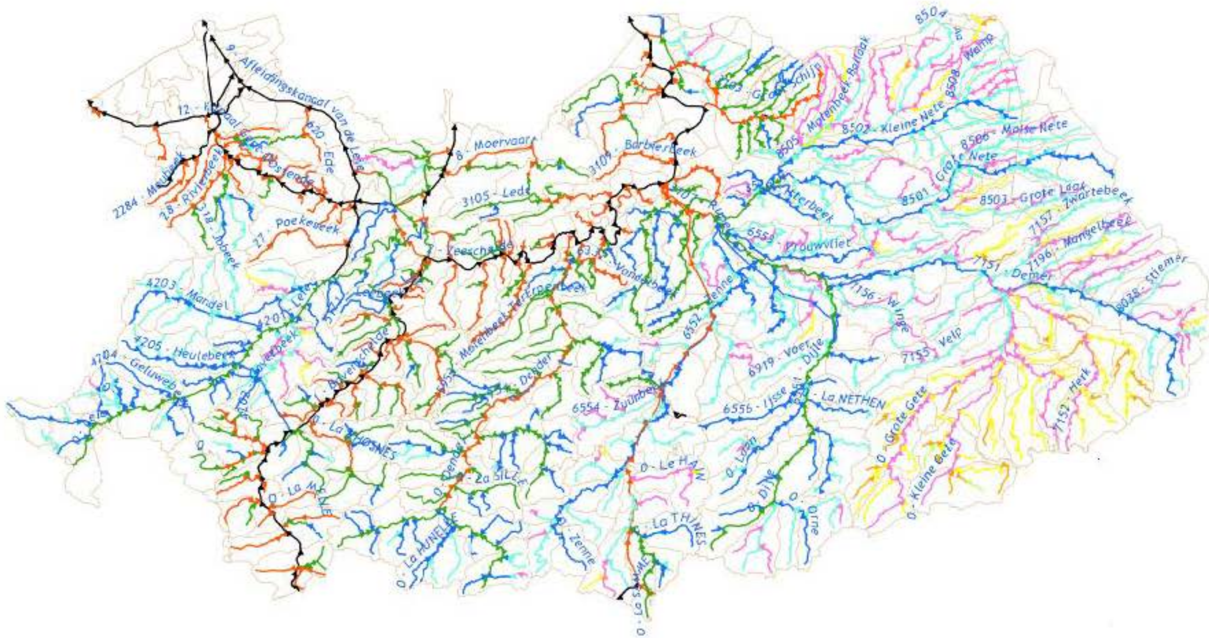

Figure 28: River network modelled with Pégase (Aquapole, 2008)

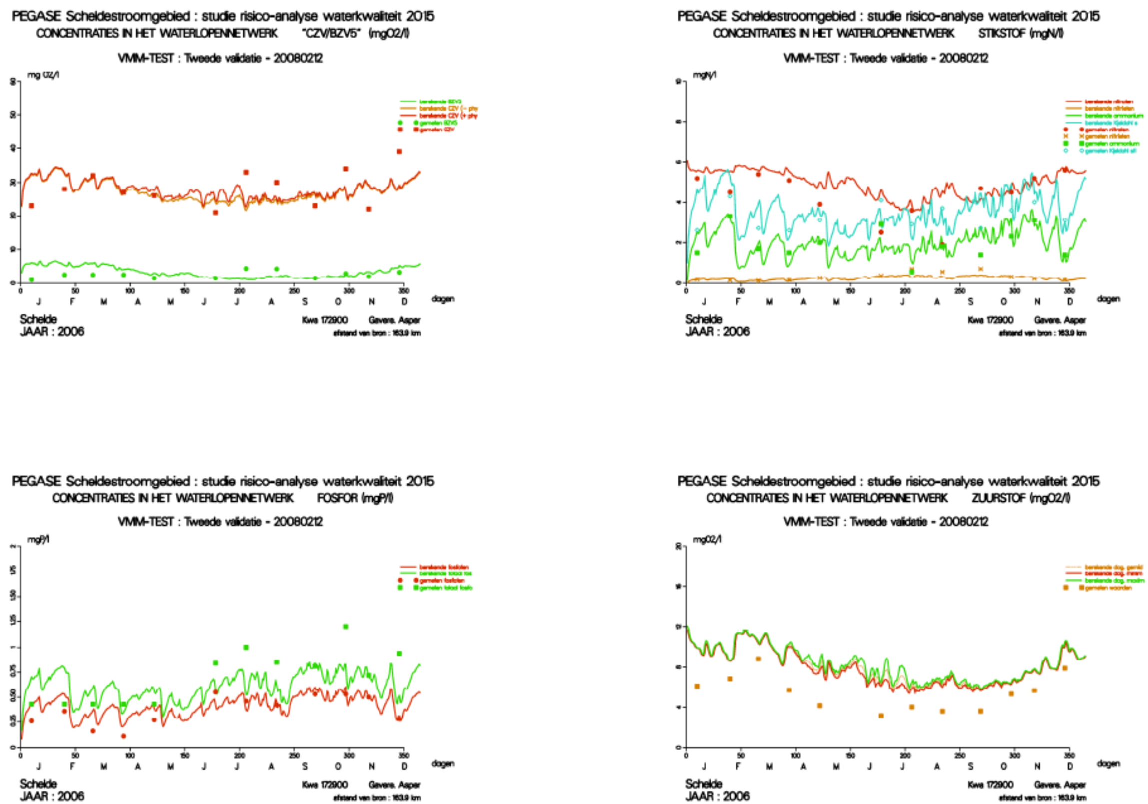

Figure 29: Outputs Pegase vs. observed concentrations for the Scheldt at Asper (top left COD (red) - BOD (green), top right NO_3 (red) - NO_2 (orange) - NH_4 green) - KjN (blue), bottom left P (green) - PO_4 (red), bottom right dissolved oxygen (Aquapole, 2008)

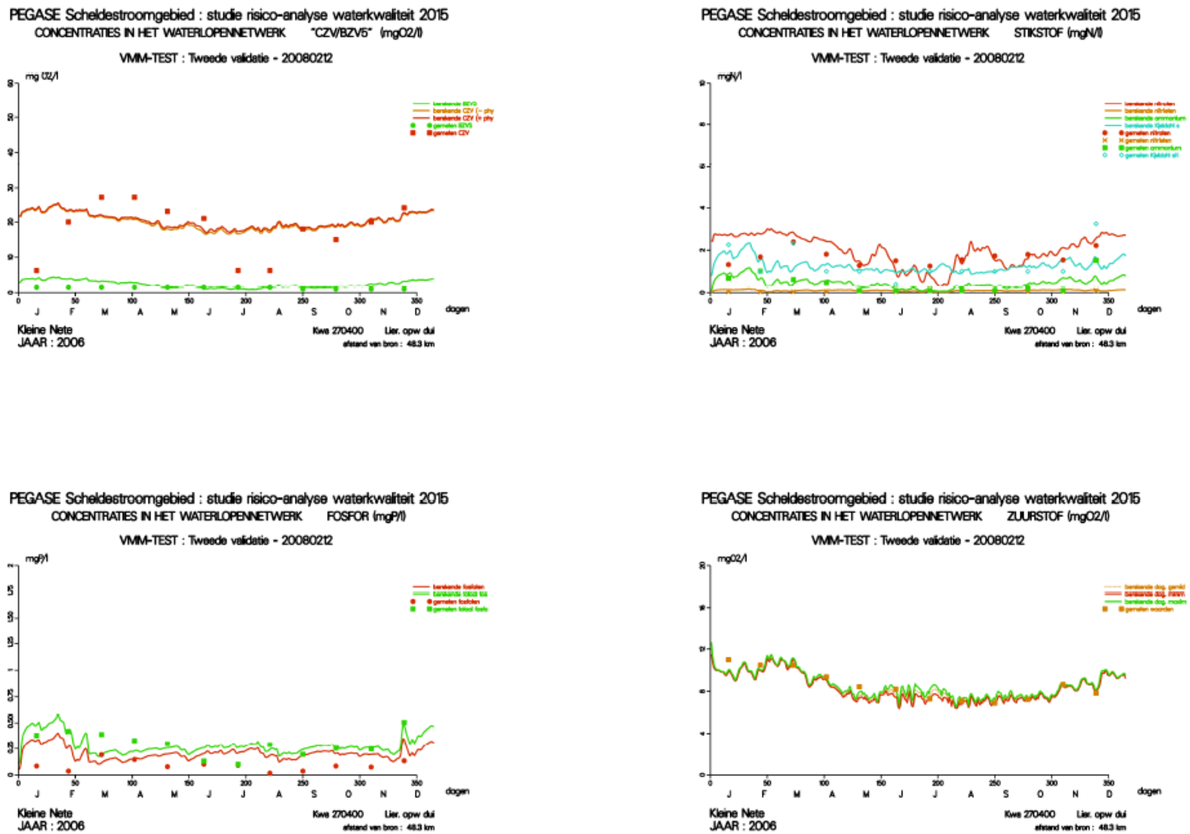

Figure 30: Outputs Pégase vs. observed concentrations for the Kleine Nete at Lier (top left COD (red) - BOD (green), top right NO_3 (red) - NO_2 (orange) - NH_4 green) - KjN (blue), bottom left P (green) - PO_4 (red), bottom right dissolved oxygen (Aquapole, 2008)

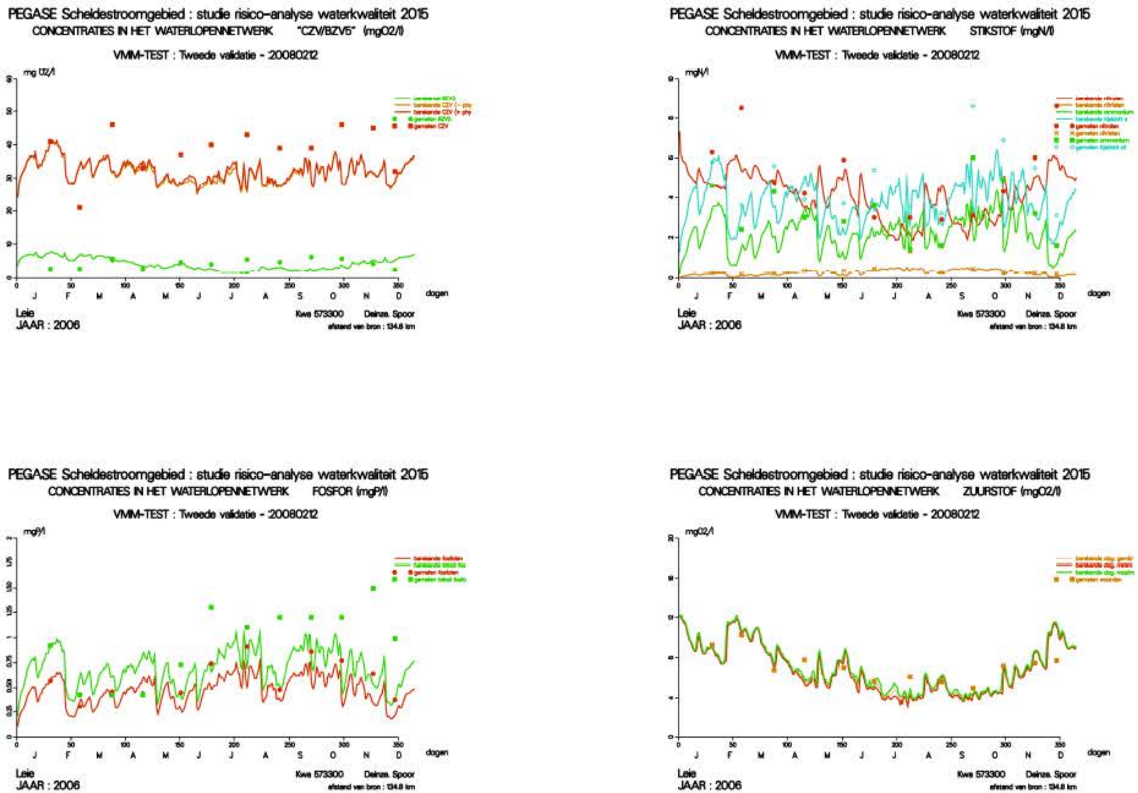

Figure 31: Outputs Pégase vs. observed concentrations for the Leie at Deinze (top left COD (red) - BOD (green), top right NO_3 (red) - NO_2 (orange) - NH_4 green) - KjN (blue), bottom left P (green) - PO_4 (red), bottom right dissolved oxygen (Aquapole, 2008)

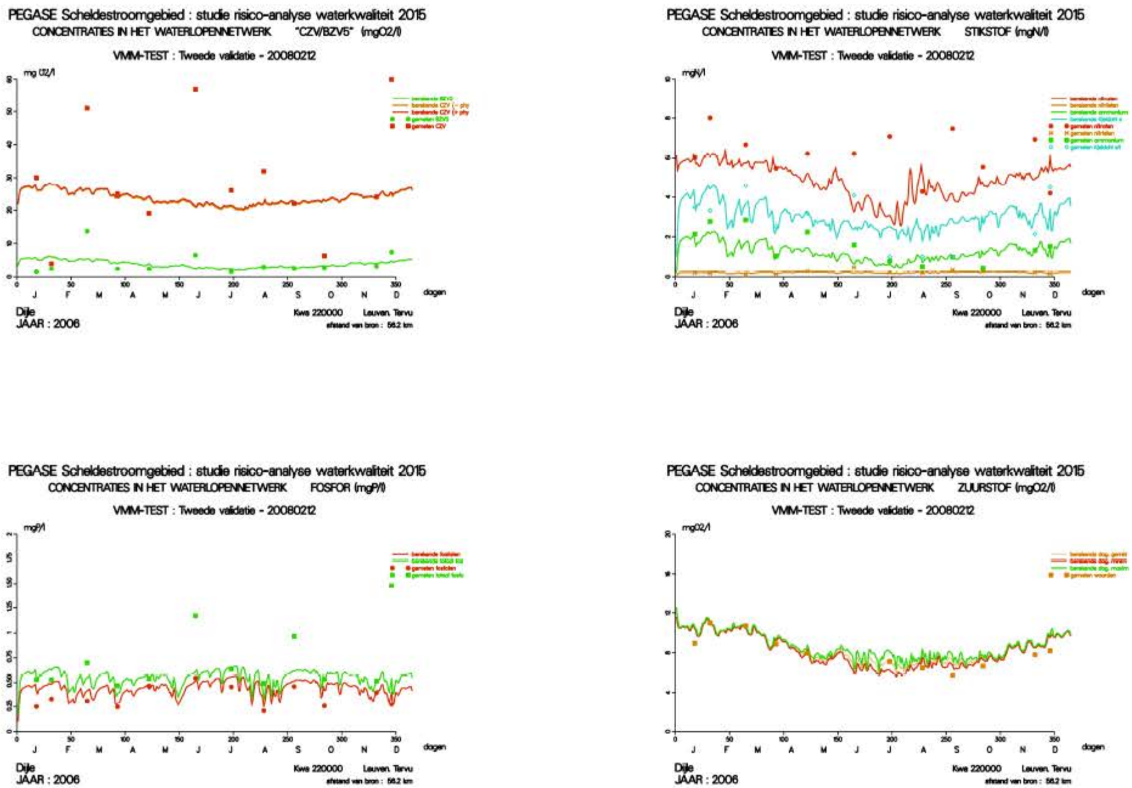

Figure 32: Outputs Pégase vs. observed concentrations for the Dijle at Leuven (top left COD (red) - BOD (green), top right NO_3 (red) - NO_2 (orange) - NH_4 green) - KjN (blue), bottom left P (green) - PO_4 (red), bottom right dissolved oxygen (Aquapole, 2008)

Annex 3. Detailed description data and results Grote Nete chapter 4

Table 43: Data sources SWAT model Grote Nete basin

Available data	Source	Frequency
Surface water quality data	Flemish Environment Agency (VMM)	Monthly
Flow data	Flemish Hydraulic Information Centre (AWZ-HIC)	Daily
Meteorological data	Royal Meteorological Institute of Belgium (KMI)	Daily
Digital Elevation Map and land use (reclassified to corn, pasture, forest, urban, water)	Flemish Agency for Geographical Information (AGIV)	/
Fertilizer data (chemical and animal manure) on municipality level recalculated for 14 subbasins.	Flemish Land Agency (VLM)	Yearly application rates
Emission data: industry, households not connected, wastewater treatment plants, point sources agriculture	Flemish Environment Agency (VMM). See annex 1	Yearly emissions

Geographical spread application fertilizer: 100% of chemical manure is applied on corn; 75% of the animal manure is applied on corn and the remaining 25% is applied on pasture. Manuring for corn is carried out between half of March until half of April. Manure application on pasture starts from half of February until half of August (based on Relaes, 2000; Sels, 2009).

Data for the economic model are a subset of the data described in annex 1.

Table 44: Detailed description results total nitrogen

Rank	Description	Index	Cost	Reduction emissions		Cost Effect Ratio	
			€/year	kg N/year	%	€/kg	€/kg point
1	Increased dairy cattle productivity	i	-364,377	16,620	1.9%	-21.9	-32.9
2	Basic measures	a	0	13,274	1.5%	0.0	0.0
3	Winter cover crops	j	136,507	47,306	5.4%	2.9	4.3
4	Construction or renovation existing WWTP	b	793,747	86,848	9.9%	9.1	9.1
5	Increased feed efficiency pigs	n	76,425	6,687	0.8%	11.4	17.1
6	UWWD standards for industrial wastewater	h	188,763	3,895	0.4%	48.5	48.5
7	Fertilization without excess	m	566,906	15,908	1.8%	35.6	53.5
8	More strict nutrient legislation + reducing poultry	pq	346,886	9,218	1.1%	37.6	56.4
9	Low cost sewage	d	3,892,738	38,169	4.4%	102.0	102.0
10	New collectors	c	140,681	1,321	0.2%	106.5	106.5
11	Medium cost sewage	e	9,378,551	63,592	7.3%	147.5	147.5
12	Livestock reduction (cattle and pigs)	r	767,526	4,426	0.5%	173.4	260.1
13	High cost sewage	f	3.652,734	10,242	1.2%	356.6	356.6
14	Feed efficiency poultry	o	1,500,335	5,603	0.6%	267.8	401.7
15	Individual wastewater treatment	g	3,440,718	5,802	0.7%	593.0	593.0

Table 45: Detailed description results nitrates (90 percentile)

Rank	Description	Index	Cost	Reduction emissions		Cost Effect Ratio	
			€/year	kg N/year	%	€/kg N	€/kg point NO_3^-
1	Increased dairy cattle productivity	i	-364,377	16,620	1.9%	-21.9	-3.0
2	Basic measures	a	0	13,274	1.5%	0.0	0.0
3	Winter cover crops	j	136,507	47,306	5.4%	2.9	0.4
4	Increased feed efficiency pigs	n	76,425	6,687	0.8%	11.4	1.6
5	Fertilization without excess	m	566,906	15,908	1.8%	35.6	4.9
6	More strict nutrient legislation + reducing poultry	pq	346,886	9,218	1.1%	37.6	5.1
7	Construction or renovation existing WWTP	b	793,747	86,848	9.9%	9.1	9.1
8	Livestock reduction (cattle and pigs)	r	767,526	4,426	0.5%	173.4	23.6
9	Feed efficiency poultry	o	1,500,335	5,603	0.6%	267.8	36.5
10	UWWD standards for industrial wastewater	h	188,763	3,895	0.4%	48.5	48.5
11	Low cost sewage	d	3,892,738	38,169	4.4%	102.0	102.0
12	New collectors	c	140,681	1,321	0.2%	106.5	106.5
13	Medium cost sewage	e	9,378,551	63,592	7.3%	147.5	147.5
14	High cost sewage	f	3,652,734	10,242	1.2%	356.6	356.6
15	Individual wastewater treatment	g	3,440,718	5,802	0.7%	593.0	593.0

Annex 4. Detailed description of data web application chapter 7

Table 46: Data sources status

Source	Data	Geographical scale in model
Water quality	Observations performed by VMM on a monthly basis for period 2006-2012	Observation point (one or more points per waterbody)
Hydromorphology	Observations performed by VMM between 2000 and 2012	Observation point (trajectories of +/- 200 metres)
Sediments	Observations performed by VMM between 1996 and 2012	Observation point (several points per waterbody)

Table 47: Data sources emissions (parameters: BOD, COD, SS, N, P)

Source	Data	Geographical scale in model
Industry	Measurements performed by Flemish Environment Agency for individual companies	Individual company
WWTP	Measurements incoming and outgoing loads for 246 individual stations performed by Flemish Environment Agency in 2006, 2010 and 2012	Individual WWTP
Households not connected	Inhabitants not connected to a WWTP as inventoried by the Flemish Environment Agency (AWIS database 2013 on the sewage network in Flanders). Emission factors per inhabitant as applied by Flemish Environment Agency (N: 9.7 g/day/IE, P: 1.4 g/day/IE, COD: 89 g/day/IE.	Households are grouped according to specific sewage construction projects or at the municipality level if no sewage construction is planned (individual wastewater treatment).
Agriculture diffuse	Diffuse emissions N, P estimated by SENTWA model. (System for the Evaluation of Nutrient Transport to Water, developed by Nolte et al., 1991 and adapted by Pauwelyn et al., 1997 and Verlinden et al., 2002); sediment losses were derived from calculations with the WaTEM/SEDEM model (Van Rompaey et al., 2001; Verstraeten et al., 2002)	Waterbody level

Table 48: Data sources costs and effects of measures

Source	Measure	Data sources for costs and effects
Industry	Individual treatment	VITO and Resource Analysis (2006)
WWTP	Construction new or renovate existing WWTP to reach efficiency targets Urban Wastewater Directive	Project specific cost estimates from investment program Flanders collective sewage and treatment. Effectiveness based on effectiveness existing wastewater treatment plants with comparable technology and capacity
Households not treated by a WWTP	Connecting sewage to existing treatment with new collectors (projects planned before or during 2006)	Project specific cost estimates from investment program Flanders collective sewage and treatment. Effectiveness based on effectiveness existing wastewater treatment plants.
	Connecting sewage to existing treatment with new collectors (projects planned after 2006)	Project specific cost estimates from investment program Flanders collective sewage and treatment. Effectiveness based on effectiveness existing wastewater treatment plants
	Extending the sewage system	Cost estimates based on project specific cost estimates derived from the required length and capacity of the sewage (Gemeentelijke Uitvoeringsplannen)
	Small scale individual treatment for remote houses	VITO and Resource Analysis (2006)
Agriculture	Increased feed efficiency (pigs and poultry)	Cost and effects from ILVO (2007); reduced nutrient losses simulated by SENTWA
	More strict nutrient legislation (exclusion of Nitrates Directive derogation)	Cost and effects from ILVO, 2007; reduced nutrient losses simulated by SENTWA
	Tuned fertilisation (only up to crop requirements)	Cost and effects from ILVO (2007); reduced nutrient losses simulated by SENTWA
	Buffer strips along watercourses	Cost and effects from ILVO (2007); reduced nutrient losses simulated by SENTWA, reduced erosion losses derived from WATEM-SEDEM results
	Reduced tillage	Cost and effects from ILVO (2007); reduced nutrient losses simulated by SENTWA, reduced erosion losses derived from WATEM-SEDEM results
	Winter cover crops	Cost and effects from ILVO (2007); reduced nutrient losses simulated by SENTWA, reduced erosion losses derived from WATEM-SEDEM results

Scientific curriculum vitae

Steven Broekx was born in Bree on Septembre 19, 1978. He completed his secondary education in 1996 at Agnetendal, Peer. In 2001, he obtained his masters degree in applied economic sciences (commercial engineer) at Hasselt University. During his professional career he achieved an additional masters degree in environmental sciences at the University of Antwerp in 2006 and Environmental Coordinator level A in 2008. He also followed courses on environmental and transportation economics at the University of Leuven in 2003.

In 2001 he started working at Linpac Automotive as a Sales Engineer. In April 2003, he started working at VITO as a R&D professional in environmental economics. Since 2009 he is project manager leading a team of environmental economists. He has an extensive knowledge on cost-benefit analysis, cost-effectiveness analysis, environmental cost estimations, ecosystem service valuation and this for topics as transportation, water and air pollution, biodiversity and health. He works on research and consultancy projects for local and European environment administrations and agencies.

Since May 2011 he is trying to achieve a doctoral degree in bioengineering sciences on cost-effective water management.

He participated in many national and international scientific conferences, seminars and workshops with oral and poster contributions. In addition he is author and co-author of various international peer-reviewed publications.

Education

1990-1996	Secondary education (Latin-Sciences) at Agnetendal, Peer
1996-2001	Master in applied economic sciences: commercial engineer at Hasselt University
2002-2003	Environmental and transportation economics at University of Leuven (course)
2004-2006	Master in environmental sciences at University of Antwerp
2008	Environmental coordinator A at University of Antwerp
2011-2014	PhD in Bioscience engineering. PhD thesis title: Modelling tools for cost-effective water management. Ghent University. Faculty of Bioscience engineering, Department of Agricultural Economics.

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