Cost-effectiveness analysis of ecosystem management with ecosystem services : from theory to practice

Reference:
Boerema Annelies, Van Passel Steven, Meire Patrick.- Cost-effectiveness analysis of ecosystem management with ecosystem services : from theory to practice
Ecological economics - ISSN 0921-8009 - 152(2018), p. 207-218
Full text (Publisher's DOI): https://doi.org/10.1016/J.ECOLECON.2018.06.005
To cite this reference: https://hdl.handle.net/10067/1517540151162165141

Cost-effectiveness analysis of ecosystem management with ecosystem services: from theory to practice

Boerema, A.[1 *], S. Van Passel[2] and P. Meire[1]

[1] Ecosystem Management research group (ECOBE), Department of Biology, University of Antwerp, Antwerp, Belgium

[2] Departement of Engineering Management, Faculty of Applied Economics, University of Antwerp, Antwerp, Belgium

*corresponding author: annelies.boerema@uantwerpen.be

Address corresponding author: University of Antwerp, Universiteitsplein 1 C.131, 2610 Wilrijk, Belgium

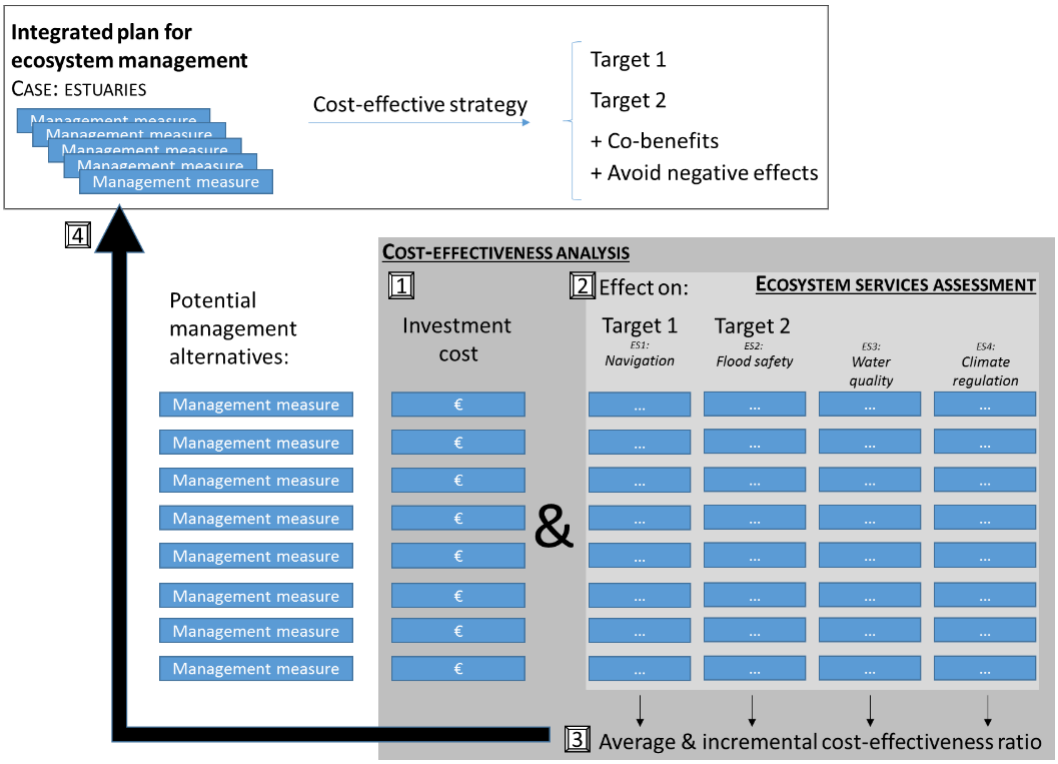
Boerema, A., S. Van Passel, and P. Meire. 2018. Cost-Effectiveness Analysis of Ecosystem Management With Ecosystem Services: From Theory to Practice. Ecological Economics 152:207-218.

Abstract

Integrated ecosystem management is challenging due to many, often conflicting, targets and limited resources to allocate. A valuable and straightforward approach is to integrate an ecosystem services assessment in a cost-effectiveness analysis as method to evaluate and compare the cost-effectiveness of several management scenarios to reach one or more objectives and take into account the potential effects on other ecosystem functions and services. Nevertheless, this method is not commonly used in ecosystem management evaluation but can provide an alternative for the frequently used but often contested cost-benefit analysis (which requires the step of assigning a monetary value to each benefit). The aim of this study is to apply the cost-effectiveness analysis in combination with an ecosystem services assessment on a real case-study (comparing alternative management strategies for estuaries) to derive lessons learned to go from theory to practice. The application of this method for the case-study reveals many remaining challenges such as data availability and knowledge to assess ecosystem effects of management measures. Nevertheless, the analysis demonstrates that this method can be used for making a more integrated evaluation and supporting better-informed management decisions.

Keywords: average and incremental cost-effectiveness ratio, estuary, multiple management targets, integrated management decision

Graphical abstract



Highlights

- An ecosystem services assessment is integrated in a cost-effectiveness analysis.
- The method is applied on 4 management alternatives in estuaries.
- Despite data limitations, the cost-effectiveness is calculated for four ecosystem services.
- The main challenge of assessing management effects is finding a common indicator.
- Flood control areas are more cost-effective than traditional dikes.

1 Introduction

Optimal and cost-effective management of an ecosystem involves many stakeholders such as management agencies, cities, local residents, economic sectors, acting at different levels from local to global, with often contrasting interests and objectives (Turner et al. 2003, Sanon et al. 2012, Labiosa et al. 2013). This results in different environmental problems to be addressed simultaneously, with potential conflicts and trade-offs between economic, ecological, environmental and social interests (Halpern et al. 2011, Zhang et al. 2014). Indeed, managing an ecosystem to improve a single function can also have an impact on other ecosystem functions with positive (co-benefits) or negative (trade-offs) implications for the ecosystem functioning and society (Seppelt et al. 2013, Smith et al. 2013). Furthermore, policy and legislation is often disconnected in separate departments such as air, water and soil (Smith et al. 2013). The ecosystem services (ES) concept proved to be a useful framework to assess different ecosystem functions and links ecological and socio-economic interests (Boerema et al. 2017). Case-studies aim at optimising ecosystem management for several ES, e.g. for agricultural management (Crossman and Bryan 2009), forest management (Macmillan et al. 1998), integrated coastal and river management (Pouwels et al. 1995, Breen and Hynes 2014). A common method for project evaluation is a cost-benefit analysis. If combined with an ecosystem services analysis, a variety of economic, social and ecological benefits and costs of a project can be balanced (Boerema et al. 2016). However, this requires that all costs and benefits are expressed in monetary values which is strongly discussed and contested (Gómez-Baggethun and Ruiz-Pérez 2011). An alternative economic approach that can be used for environmental management evaluation is a cost-effectiveness analysis. This method compares the investment cost with management effects and this could be expressed in any unit (Balana et al. 2011). Hence data requirements are lower and the step towards the monetary valuation of benefits can be avoided. As a disadvantage, the analysis treats each ES separately and integrating over different ES is not straightforward because they are expressed in different units. Nevertheless, management of ecosystems is oriented towards different targets and therefore it is important to make an integrated assessment of the impact of management alternatives for the different targets. This research paper tests the practical application of the cost-effectiveness analysis to evaluate and compare ecosystem management measures for multiple management targets and additional side-effects.

When managing environmental problems, the effectiveness of planned actions to solve the problem is an important criterion. In combination with general budget restrictions, this raises the questions to find the most cost-effective strategy to solve environmental problems at the least possible cost (Interwies et al. 2004). A cost-effectiveness analysis (CEA) is applied in very diverse cases to find an optimal and cost-effective solution, e.g regarding site selection (e.g. Adame et al. 2014), or forest management options (e.g. Tóth et al. 2013). Another application is to solve specific problems such as water quality improvement, pollution and eutrophication reduction (Comello et al. 2014), biodiversity conservation (Drechsler et al. 2007, Helm and Hepburn 2012), greenhouse gas emission reduction (MacLeod et al. 2010), water provision improvement (Yang 2011), and flood prevention (Dawson et al. 2011). Limited examples study the optimal strategy for integrated ecosystem management when taking into account several stakeholders and objectives (e.g. Pouwels et al. 1995). Most CEA studies related to environmental policy evaluation are theoretical and conceptual (Prato 2007, Wainger et al. 2010). Practical examples use mostly scores to estimate benefits of management options, either given by experts and/or stakeholders (e.g. Macmillan et al. 1998, Bryan 2010) or translated from biophysical data and models (e.g. Pouwels et al. 1995, Crossman and Bryan 2009). In these studies, a weighting factor, given by experts and/or stakeholders or randomly chosen (e.g. all with equal weight), is applied to make an integrated evaluation over several benefits.

Assessing the effectiveness of management measures is an important but challenging step in the CEA study. The impact on the targeted objective and also additional positive and negative effects should be investigated to take into account the impact on all the functions and services of the ecosystem (Balana et al. 2011). A common indicator should be found for each service to be able to compare the impact of different management measures (Convertino et al. 2013). This is challenging for services consisting of various components such as water quality regulation (e.g. different nutrients, oxygen, acidity etc.) (Smith et al. 2013). It is important to fully understand and be able to quantify the impact of management measures to the entire coupled ecologic and socio-economic system. Selecting the most effective management strategy for several objectives adds another degree of complexity since the impact on the different objectives could not be added up because the benefits are expressed in different units. The objective of this paper is to test and examine the approach to evaluate and select the most cost-effective ecosystem management strategy for optimising one or more specific ES objectives. First, the basic steps of the cost-effectiveness analysis are shortly described in Section 2. Second, the method is applied on a test case (management of the Scheldt estuary for two different management targets) in section 3. Lastly, section 4 discusses the challenges and lessons learned from the application of the CEA method on a real ecosystem management case.

2 Method

For a cost-effectiveness analysis, four steps are followed: (1) collecting data on the cost of the management measures; (2) quantifying the effect of each management measure on the different ES (targets, co-benefits, negative side effects); (3) calculating the average cost of each management measure for each ES (= cost/effect on ES); (4) selecting the most cost-effective management strategy.

2.1 Step 1: Cost of management measures

A good estimation of the investment cost is the first crucial step to calculate the average cost of management measures. Accounting for all costs, including social costs and indirect costs, is important (Interwies et al. 2004, Duke et al. 2013). Furthermore, also accounting for maintenance costs after construction is important to take into account. This requires to set a time period for the analysis.

2.2 Step 2: Effect of management measures on ES

Evaluating the effect of management measures on different ecosystem services enables the comparison of the effectiveness of the different management measures regarding a specific target but also the additional contribution to delivering ecosystem services and generating societal benefits. This requires a common indicator for each ecosystem service to be able to quantify and compare the effect of each management measure. A review on methods used to quantify ecosystem services revealed the diversity of indicators demonstrating the challenge of finding one common indicator (Boerema et al. 2017).

2.3 Step 3: Average cost to invest in ES

After collecting cost data and estimating the effect of each management measure on each ES, the average cost is calculated by dividing the cost by the ES impact (e.g. €/m^3, €/cm, €/kgN, €/tonC).

2.4 Step 4: Cost-effectiveness analysis

After calculating the average cost of management measures to invest in several ES, the next step is to select an optimal set of management measures to reach one or a variety of objectives in the most cost-efficient way. In an integrated cost-effectiveness analysis (for different ES), all ES are

treated separately since each ES is assessed in different units. Options for integration over several ES effects are via ranking, visual integration or numerical integration. Visual integration based on the average (ACER) and incremental cost effectiveness ratio (ICER) method is applied in this study and both methods are shortly explained below (explained in detail in Compernolle et al. 2012). ACER and ICER are the two most commonly used methods for a cost-effectiveness analysis and give complementary results. The ACER approach results in an investment cost curve with details about all measures in each investment strategy (scenario) per ES, while the ICER approach compares the total outcome of each investment strategy (scenario) against each other per ES.

i. Average cost effectiveness ratio (ACER)

The ACER approach uses the average cost as selection criteria, i.e. ratio of the cost of a scenario relative to the benefit (cost-benefit ratio, €/unit-benefit). Measures with a low average cost are preferred over measures with a high average cost. After calculating the average cost of each management measure, an investment curve is compiled by ranking the measures according to the average cost (Figure 1 left panel). The X-axis gives the benefit of the measures for a specific ES (e.g. kg nitrogen removed) and the Y-axis gives the average cost (e.g. €/kg nitrogen removal). The surface under the investment curve equals the total cost of the measures (e.g. €/kg nitrogen × kg nitrogen removed with the measure). The curve can be used to select a set of most cost-effective measures given one of more criteria such as achieving a certain benefit (y-axis) or within a certain budget (total surface).

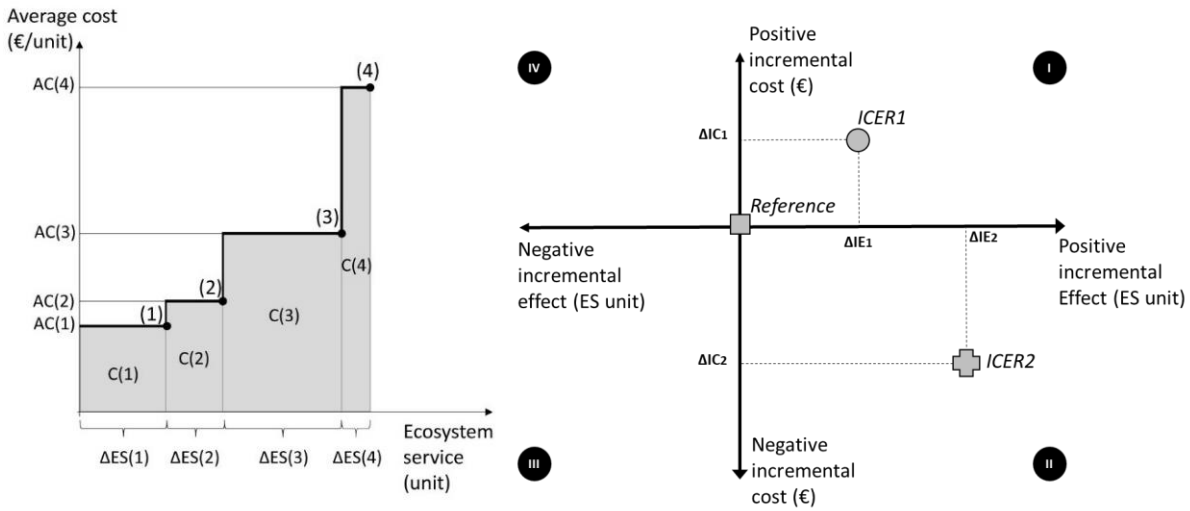


Figure 1: Left: Average cost-effectiveness plane. Each management measure is represented as a rectangular, with the width indicating the improvement for an ecosystem service (ΔES) and the height indicating the average cost to improve the ecosystem service on the y-axis (AC). The surface of the rectangular gives the total cost of the management measure (C). Right: Incremental cost-effectiveness plane. Each management measure is represented as a point in the plane showing a positive or negative incremental effect (x-axis) and a positive or negative incremental cost (y-axis) compared to a reference management measure (0-point in the plane). This results in a plane with four quadrants I-II-III-IV.

ii. Incremental cost effectiveness ratio (ICER)

For the ICER approach, measures are compared relative to one reference measure (0-point) in a cost-effectiveness plane for both the cost (y-axis) and the effect on the ES (x-axis). This results in four options (Figure 1 right panel): measure can have higher costs and higher effect on ES compared to the reference measure (quadrant I), lower cost and higher effect on ES (quadrant II), lower cost and lower effect on ES (quadrant III), or higher cost and lower effect on ES (quadrant IV). This leads to following conclusions: measures that are preferred over the reference measure

(quadrant II, e.g. ICER2), measures that are not preferred over the reference (quadrant IV), or measures for which it is not clear whether they are preferable over the reference (quadrant I and III, e.g. ICER1).

3 Case-study and results

3.1 Scenario building

The methodology is illustrated for the management of the Scheldt estuary, a representative estuary for Northwest Europe. The Scheldt estuary, located in Belgium and the Netherlands, is a large and dynamic ecosystem with many challenges and policy objectives related to increasing harbour activities, increasing flood risk, poor water quality, reduced habitat and biodiversity, etc.. Hence, developing an integrated and cost-effective management strategy requires integrated analysis of different management strategies for different targets and potential co-benefits and negative side effects. For the case study, the ACER and ICER method are applied on five scenarios that are built based on traditional and alternative management strategies for two targets navigability and flood protection (table 1). For the navigability target traditional dredging is compared with alternatives for dredging such as the nautical depth concept (or fluid mud concept) and a sediment trap. For the flood control target traditional dike heightening is compared with alternatives such as flood control areas and flood control areas with controlled reduced tide.

Table 1: Overview of five scenarios (Reference, S1, S2a, S2b, S3): set of traditional and alternative management measures for two targets (navigability target and flood protection target)

Target	Strategy	Management measure(s)	Scenario				
			Reference	S1	S2a	S2b	S3
Navigability target	(i) Traditional	Dredging, land disposal	X		X	X	
	(ii) Alternatives	Fluid mud concept, sediment trap, beneficial use of dredged material		X			X
Flood protection target	(iii) Traditional	Dike heightening	X	X			
	(iv) Alternatives	Flood control area			X		
		Flood control area with controlled reduced tide				X	X

3.2 Step 1: Cost of management measures

Data on the investment cost of the management measures is taken from projects in the Scheldt estuary (management plan of the Upper Sea Scheldt, Lower Sea Scheldt and Western Scheldt), projects studied in the EU Interreg IV B project TIDE on managing estuaries (www.tide-toolbox.eu) and projects described and studied in international peer reviewed papers (Boerema and Meire 2017). Data on the cost of all measures in the scenarios is summarised in table 2 and details can be found in Appendix 1.

3.3 Step 2: Effect of management measures on ES

To test and examine the cost-effectiveness methodology, the different scenarios are evaluated for its impact on the two targets navigability and flood protection. Although not typically categorised as ecosystem services, abiotic services such as navigability are provided by the carrier function of biophysical river and marine structures. Therefore, water availability and water regulation that allows for navigation (shipping) is added in several ecosystem services studies for rivers and marine systems (Atkins et al. 2011, Jacobs et al. 2015). Furthermore, also the impact regarding two additional ecosystem services are assessed (water quality regulation and climate regulation). These four ES are considered to be important in estuaries with a harbour, large cities and pollution problems. In addition, some pragmatic aspects also determined the selection of these services: a common unit

to quantify changes in the ES could be found and sufficient data was available to quantify the impact of at least several different management measures. Overall, quantitative data on the effects of management measures on the selected ES is limited (Boerema and Meire 2017) and the methods to express the effects of different measures vary a lot which complicates the comparison of effects between measures. A review on management for estuarine ecosystem services provides a database of studies that made a quantitative assessment of estuarine management measures on one or more ecosystem services (Boerema and Meire 2017). Relevant studies for the four selected ES were selected and reviewed to describe how the effect of management measures in estuaries are studied regarding navigation, flood safety, water quality regulation and climate regulation. For the purpose of the cost-effectiveness analysis, a common indicator was selected which enabled the comparison of most measures. A description of the variety of indicators and the selection of a common indicator used in this study is described below. Data on the impact on the four ES for all measures in the scenarios, is summarised in table 2 (details in Appendix 1). Regarding effects that have an annual character, a period of 10 years is considered.

For the **ES water quantity regulation for transportation (navigability)** we focus on the maintenance of navigation channels and not on the maintenance of harbours and docks for which other specific measures could be applied. Related to the navigability target, management measures can be grouped in three categories. First, measures to adapt the dimensions of the navigation channel by increasing the navigable depth and straightening meanders channel. Second, measures to reduce the need for dredging such as stabilising navigation channels (e.g. training wall), the nautical depth concept to increase the navigable depth without dredging (Manson and Pinnington 2012), and prevent shoreline erosion. Lastly, measures to dispose dredged material that allow to mitigate effects of dredging (e.g. habitat restoration and creation with dredged material).

The most common type of data that was found to express the effect on navigation is the volume of sediment that is removed or prevented to be deposited in the navigation channel (m^3 sediment removed or avoided). The latter effect is usually in an indirect way by sediment burial elsewhere. This is expressed in annual m^3 sediment buried per hectare. We assume that sediment that is buried outside the navigation channel will not be deposited in the navigation channel and hence results in a reduction of the need to dredge.

Related to the **ES flood protection**, management measures can contribute (or negatively influence) to this service in various ways. First, measures can change the tidal regime and high and low water levels by changing water to flow on land (e.g. dikes or intertidal nature) and by changing water to flow further upstream or downstream (e.g. barrier). Second, measures can provide water storage capacity in the system that changes discharge and high and low water levels during peak flows. Third, measures can contribute to energy dissipation and wave attenuation in the system by hard structures or vegetation (e.g. wetland creation). Fourth, measures can improve river drainage and water retention to mitigating the effects of floods and droughts.

The effect of management measures on flood safety is expressed in various ways such as reduction in mean high water level (cm), wave attenuation and reduction in wave height (cm, %), water storage capacity (ha, m^3), river drainage and manning coefficient (%), flood risk level for the estuary (protected against storm surges with a risk of 1/xx years), and flooded area with damage (ha, €). The common unit for flood protection that is applicable to assess the impact of most management measures is a reduction in the mean high water level (MHWL), expressed in cm. However, many of the measures have an indirect effect: either by an increase of dike height (cm) or avoiding that higher dikes are required (cm avoided dike increase). To be able to compare the effect of the different measures on flood safety, we made a strong assumption that a cm MHWL reduction gives a similar effect on flood safety as cm dike increase. This follows from the simple

idea that improving flood safety requires either a lower high water level or a higher dike to block the water. However, in reality both effects are not directly linked due to the dynamic characteristics of flood risk and water levels in an estuary. This is further discussed in section 4.

The **ES water quality regulation** consists of many components (sub-services) that should be studied separately such as nitrogen, phosphate, oxygen, and pollutants. Because nitrogen is one of the important components in the water quality regulation, and, as a secondary and pragmatic reason, because most available data relates to the impact on nitrogen, only nitrogen removal is considered in this analysis as indicator of changes in water quality (kg nitrogen per hectare). There are different ways how measures affect nitrogen concentrations: buried with sedimentation (e.g. intertidal areas along the river), avoid new nitrogen input by reducing nitrogen surplus that flows to the river (e.g. agro-measures), nitrogen removal with vegetation removal (nitrogen content in the vegetation), and remobilising buried nitrogen (e.g. dredging). The first two are considered to be directly comparable but the latter two are excluded because the effect could not be expressed in the same unit.

Lastly, the **ES climate regulation** consists of several components (sub-services) such as carbon sequestration, greenhouse gas emissions and local temperature effect. Following a similar reasoning as for water quality, only carbon burial due to sedimentation is considered in this analysis as indicator of changes in climate regulation (ton carbon per hectare per year). It is assumed that this is permanent burial. This assumption is only valid in case of net sedimentation.

Table 2: Overview of management measures analysed for the different scenarios for the two targets navigability and flood protection.

Targets	Traditional (T) or alternative (A) approach	Scenario					Management measure	Cost	Ecosystem services (ES)			
									Navigability	Flood protection	Water quality regulation (nitrogen)	Climate regulation (carbon)
		Ref.	S1	S2a	S2b	S3			million m^3	cm MHWL(i) reduction	kgN	tonC
Navigability: 3th deepening Scheldt: guarantee 13.1 meter depth [a]	T	X		X	X		Capital dredging	1-20 €/m^3 [b]	14	-5.5 à -10	Negative; no data available	0
							Maintenance dredging	1-20 €/m^3 [b]	100		Negative; no data available	0
							Land disposal	20-70 €/m^3 [b]	114	0	0	0
	A		X			X	Fluid mud	0 €/m^3	57 [f]	0	0	0
							Sediment trap	1-20 €/m^3 for initial dredging; 1-5 €/m^3 for annual maintenance	112,500	0	0	0
							Remaining capital dredging	1-20 €/m^3 [b]	7	-2.25 à -5	Negative; no data available	0
							Remaining maintenance dredging	1-20 €/m^3 [b]	50		Negative; no data available	0
							Beneficial disposal	0.20 and 24 €/m^3 [c]	57	Positive; no data available	Positive or negative; no data available	Positive or negative; no data available
Flood protection: safety level T4000 (Sigmaplan)	T	X	X				Dikes	255 million € [d]	0	115	0	0
	A(a)			X			Flood control area	263 million € [e]	0	Equivalent of 115	0	0
	A(b)				X	X	Flood control area with controlled reduced tide	277 million € [e]	270,000 [g]	Equivalent of 115	7,85,000 [h]	4,000 [h]

References and abbreviation:
(a) (Arcadis-Technum 2004, De Wit and Sas 2007)
(b) (OVAM 2003)
(c) (Comoss et al. 2002, Yozzo et al. 2004, Boerema and Roose 2012, HPA 2012, Wasserman et al. 2013)
(d) (Broekx et al. 2011)
(e) (Smets et al. 2005)

(f) (Kirby 2013)
(g) (IMDC et al. 2004, Temmerman et al. 2004)
(h) (Tijdelijke Vereniging Resource Analysis et al. 2004)
(i) MHWL: mean high water level

3.4 Step 3 & 4: Average and incremental cost to invest in ES and the cost-effectiveness analysis of estuarine management for 4 ES

i. ACER approach: average cost effectiveness ratio

The results from the ACER approach show the investment cost curves of the 5 scenarios with details about the average cost of each measure in the scenarios to invest in navigability (Figure 2), flood protection (Figure 3A), water quality regulation (Figure 3B), and climate regulation (Figure 3C). This enables the identification of measures with high or low impact on the ES (width of horizontal lines) and measures that have a high or low average cost to invest in the ES (y-axis). Similarities and differences between the investment cost curve of different scenarios gives insight in the scenarios (both regarding the investment cost and regarding the potential to invest in the ES). This is further explained below for each of the ES.

For the **ES navigability** (Figure 2), there are mainly two types of investment cost curves for the five scenarios: with a lower investment cost curve for the scenarios with alternative measures for dredging (scenarios S1 and S3), and higher investment cost curves for scenarios with traditional dredging (scenarios Ref., S2a, S2b). The financial benefits of the alternative measures are clearly shown in these graphs (lower average costs): first in scenarios S1 and S3 half of the dredging requirement is avoided by applying the fluid mud concept that is for free, and second the average cost of the beneficial sediment disposal in scenarios S1 and S3 is much lower than the average cost of land disposal applied in the other scenarios. Furthermore, the dimension of the sediment trap in our analysis is not beneficial in light of the sediment volumes that should be removed for the 3th deepening of the Scheldt and the required maintenance. Lastly, the average cost of sediment burial in a flood control area or flood control area with controlled reduced tide (FCA-CRT) is 100 times higher than sediment removal with dredging. Therefore, scenario S1 results in the most cost-effective outcome when optimising for the navigation target (because there is no investment in the measure FCA-CRT with an extreme high average cost for navigability).

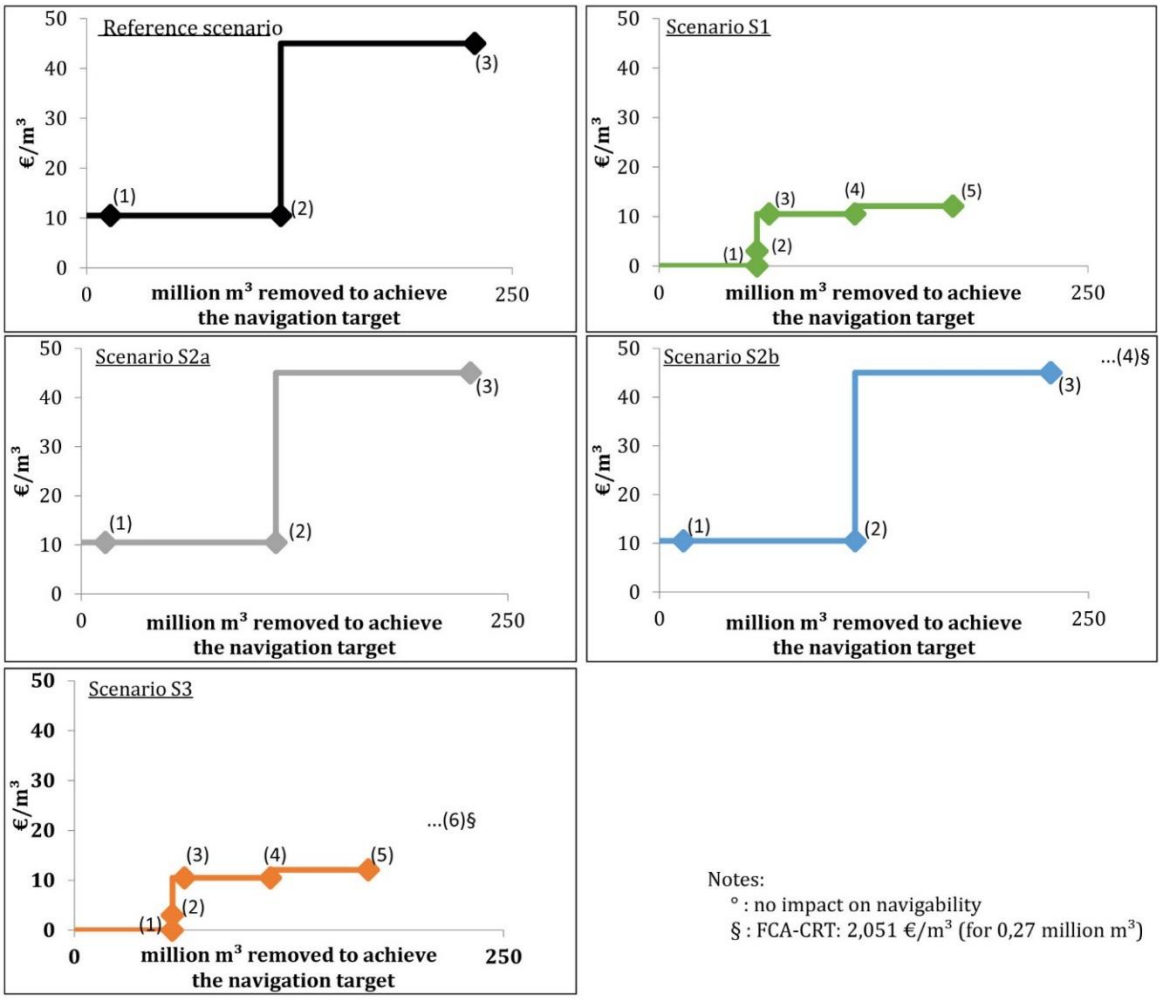
For the **ES flood protection** (Figure 3A), 3 different investment curves are visible with differences in the average cost but an equal effect on the ES. The traditional dike option gives the lowest average cost (scenarios Ref., S1), followed by FCA (scenario S2a) and the FCA-CRT (scenarios S2b, S3). The negative effect of traditional dredging (smaller in the scenarios with alternatives for dredging, S1 and S3), can easily be compensated by both the traditional dike strategy and the alternatives for dikes. Data on the impact of beneficial use of dredged material (for scenario S1 and S3) is missing, but a potential positive effect is assumed (Table 2). Therefore, scenario S1 results in the most cost-effective outcome when optimising for the flood protection target.

For the **ES water quality regulation** (Figure 3B) and **ES climate regulation** (Figure 3C), data is only available for the project FCA-CRT which is part of the scenarios S2b and S3. Related to water quality regulation, dredging is assumed to generate negative effects (by remobilising nutrients) but data to estimate this effect is missing. Hence, scenario S3 is the referenced scenario when focussing on water quality regulation (nitrogen removal).

ii. ICER approach: incremental cost effectiveness ratio

In contrast to the ACER approach, the results from the ICER approach (Figure 4) show clearly the differences between the total outcomes of the different management scenarios (but not details for the different measures in each scenario). For the **ES navigability**, scenarios S1 and S3 (both with

alternatives for dredging) give the same outcome with less sediment volume to manage (170 million m^3 instead of 215 million m^3) and at the lowest total cost. Both scenarios also give the most cost-effective result for the **ES flood protection** because the total cost is lower for the same safety level (T4000) and a slightly better result for the MHWL since the negative effect of dredging is partially avoided. For the **ES water quality regulation** (nitrogen) and **ES climate regulation** (carbon), scenarios S2b and S3 (both with alternatives for dikes) give the most cost-effective result because the other scenarios have no effect on these ES. To conclude, the most cost-effective scenarios to invest in navigability and/or flood protection are S1 and S3 but when also water quality regulation and/or climate regulation are added to the decision exercise only S3 is overall most ideal (lowest investment cost to invest in all four ES).



Notes:
° : no impact on navigability
§ : FCA-CRT: 2,051 €/m³ (for 0,27 million m³)

Ref.: traditional dredging + traditional dikes
(1) Capital dredging
(2) Maintenance dredging
(3) Land disposal dredged material
(4) Dikes (°)

S1: alternatives for dredging + traditional dikes
(1) Fluid mud concept
(2) Sediment trap
(3) Capital dredging
(4) Maintenance dredging
(5) Disposal of dredged material in the system
(6) Dikes (°)

S2a: traditional dredging + alternative for dikes (FCA)
(1) Capital dredging
(2) Maitnenance dredging
(3) Land disposal dredged material
(4) Flood control area (°)

S2b: traditional dredging + alternative for dikes (FCA-CRT)
(1) Capital dredging
(2) Maintenance dredging
(3) Land disposal dredged material
(4) Flood control area with controlled reduced tide (§)

S3: alternatives for dredging + alternative for dikes (FCA-CRT)
(1) Fluid mud concept
(2) Sediment trap
(3) Capital dredging
(4) Maintenance dredging
(5) Disposal of dredged material in the system
(6) Flood control area with controlled reduced tide (§)

Figure 2: Average cost effectiveness ratio (ACER): Investment curve per scenario for the ES **navigability**, based on the efficiency of each management measure to achieve the navigation target (m^3 sediment removed to achieve the target of the 3th deepening of the Scheldt) and its average cost. Notes: dikes and flood control areas (FCA) have no impact on navigability (°) and the average cost of a flood control area with controlled reduced tide (FCA-CRT) is too high to show in the graphs (§).

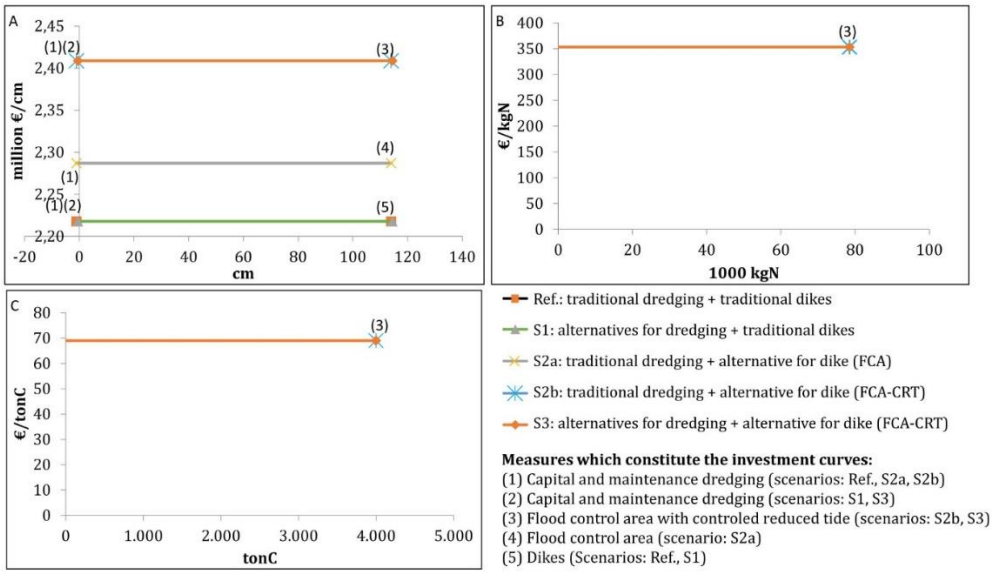


Figure 3: Average cost effectiveness ratio (ACER): Investment curve per scenario (Ref., S1, S2a, S2b, S3) for the ES **flood protection** in cm (A), **water quality regulation** in kgN (B) and **climate regulation** tonC (C), based on the contribution of each management measure to the ES (X-axis) and its average cost (Y-axis). Related to the ES flood protection (A): all scenarios start with a negative value due to the dredging measures: (1) being larger in case of the traditional dredging strategy (scenarios Ref., S2a and S2b) and (2) smaller in case of the alternatives for dredging with less traditional dredging.

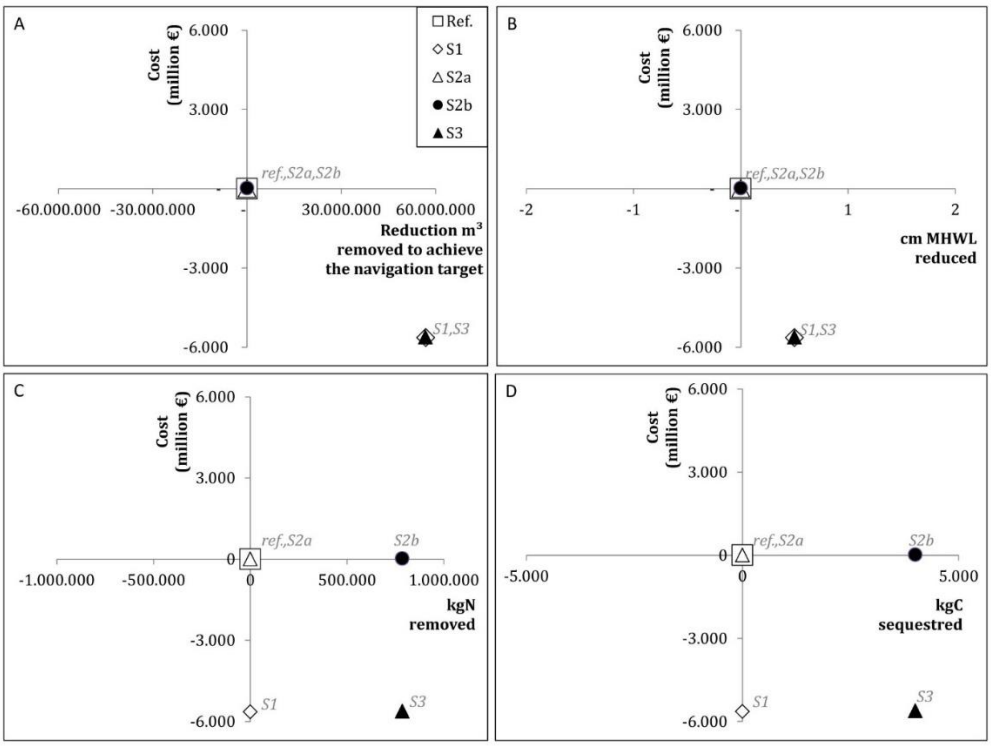


Figure 4: Incremental cost effectiveness ratio (ICER): comparing 5 scenarios for the 4 ES: A: navigability, B: flood protection, C: water quality regulation (nitrogen), D: Climate regulation (carbon). Five scenarios are compared: reference scenario with traditional dredging and traditional dikes; S1 with alternatives for dredging and traditional dikes; S2 with traditional dredging and alternatives for dikes (S2a with flood control area and A2b with flood control area with controlled reduced tide) and S3 with alternatives for dredging and alternatives for dikes (with flood control area with controlled reduced tide).

4 Challenges and lessons for practical applications

Integrated management of an ecosystem is challenging when taking into account different targets and additional ES. The cost-effectiveness method combined with an ecosystem services assessment is tested in this study as method to select the most cost-effective management strategy when taking into account two main targets (navigability and flood protection) and two additional relevant services in the estuary (water quality regulation and climate regulation). The two mostly methods for a cost-effectiveness analysis are tested to select the most optimal set of management measures to optimise one or more ES in our case-study: average cost-effectiveness ratio (ACER) and incremental cost-effectiveness ratio (ICER). To calculate the cost-effectiveness of the management measures, several assumptions had to be made (e.g. effect of dredging on the mean high water level and related flood risk). As a result, the outcome of the exercise presented in this study is not to be seen as representative for the real world situation. Nevertheless, lessons are learned from the use of the available biophysical data in this exercise. A hypothetical exercise or the use of scores based on stakeholder preferences and expert judgement to estimate the impact of management projects (Pouwels et al. 1995, Bryan 2010) would have been easier and a way to avoid the lack of data and the need to make simplified assumptions. However, by using biophysical data as much as possible this study also includes a test of how far we can get with available data. At this stage, the available knowledge on the effects of management measures is still too limited to be able to make an accurate calculation of the cost-effectiveness. When more accurate data is available in the future, the method tested in this study can be used to select the most optimal management measures.

4.1 Step 1: Cost of management measures

The cost per management measure is retrieved from individual real-life projects (Scheldt estuary, TIDE projects and international literature). Regarding the cost of measures it is often not specified which part of the costs is included (e.g. construction cost, maintenance cost, labour cost, etc.). Data found on the investment cost of projects is assumed to account for all costs. For many measures, a large range in the cost data was found. This can be linked to the number of publications from which data is taken, but also the variability of costs depending on local conditions of the measures. An accurate estimate of the cost of the investigated management measures will be crucial for a realistic application of the cost-effectiveness analysis.

4.2 Step 2: Effect of management measures on ES

Finding a common indicator per ES to evaluate the impact of the different management measures on the same basis remains challenging. Many ES are complex with different components (Smith et al. 2013). By selecting only one indicator per ES only one aspect of the service is taken into account (Boerema et al. 2017). For example for flood protection, the most common unit is cm reduction in the mean high water level (MHWL) but then the storage capacity of the system is not taken into account and neither is energy dissipation and wave attenuation. Other examples are that only the impact for nitrogen is considered as indicator for water quality regulation and carbon sequestration for climate regulation. When different aspects of a service cannot be combined in one indicator, different indicators should be selected and considered as separate sub-services for which a separate effectiveness assessment should be undertaken.

Furthermore, available data for certain management measures can be expressed in an indicator other than the selected indicator. For that reason, these management measures had to be excluded from the analysis. Based on the available data and the diversity of units that are in use to estimate the impact of management measures, it appears impossible to compare all management practices with only one metrics per ES.

Overall, we were able to find an indicator for each ES to compare the impact of a diverse set of management measures. However, to do so we had to make assumptions about the direct link between several effects. Related to navigability, it was assumed that the actual m^3 sediment removed from the navigation channel has the same impact as m^3 sediment buried outside the navigation channel with the reasoning that it will not end in the navigation channel and dredging of this volume is avoided. For flood protection, a direct link is assumed between a reduction in the mean high water level (cm) and increase in dike height (cm) or avoided need to increase the dike height (cm). For nitrogen removal, linked to water quality regulation, it was assumed that the kg nitrogen buried with sedimentation (e.g. in intertidal areas along the river) has the same impact as the kg nitrogen that is avoided by reducing nitrogen surplus that flows to the river (e.g. agro-measures to reduce nutrient leaching). However, in reality it is not guaranteed that these effects are directly linked (e.g. that the volume of sediment buried outside the navigation channel is avoided from the navigation channel and would otherwise have been dredged). As argued before, the main focus of this study was on the methodology of combining an ES assessment with a cost-effectiveness analysis to evaluate and compare management measures. When more knowledge on the impact of management measures is available in the future and when the effects are expressed in a consistent unit, the outcome of the illustrated method will be improved.

4.3 Step 3 & 4: Average and incremental cost to invest in ES and the cost-effectiveness analysis of estuarine management for 4 ES

i. Visual integration for several ES with ACER and ICER

The ACER approach gives details about all projects in each investment strategy (scenario). By looking at the average cost to realise an improvement for a certain ES, information about the total cost is however lacking. The total cost is simply the surface under the investment curve (Figure 2-3), but with complicated investment curves it can be hard to compare visually. The ICER approach gives a better insight in the total outcome, but without details about the investment strategy and individual projects in the scenarios. Therefore, we conclude that both methods provide supplementary information. For both approaches hold that visual integration for several ES is only possible with a limited amount of decision options (scenarios) and/or for a limited amount of ES.

By comparing scenarios with the ACER and ICER approach, synergies and trade-offs for integrated management of different ES can be revealed. For scenarios that are more expensive with a higher ES impact, or less expensive with less impact, it is not possible to make a conclusive decision if it is better or worse than other scenarios. If a scenario is good for one ES but generate a negative effect for another ES, this scenario should not be excluded necessarily when the negative effect can be compensated. However, when this negative effect cannot be compensated this can be used as a conclusive reason to exclude the scenario.

In case no conclusive decision can be made between scenarios (e.g. higher effect but also higher cost), a cost-benefit ratio can be determined to compare the net benefit of the alternatives even if the benefits are not expressed in monetary terms (figure 5). A lower ($\lambda 1$) or higher ($\lambda 2$) monetary value (€/ES unit) will result in a lower or higher monetary value of the incremental effect ($\lambda 2 \Delta ES1 > \lambda 2 \Delta ES1$). The difference between the monetary value of the incremental effect and the incremental cost gives the net benefit which can be positive ($\lambda 2 \Delta ES1 > \Delta IC1$) or negative ($\lambda 2 \Delta ES1 < \Delta IC1$). ICER1 is only preferred over the reference in case of a positive net benefit (with $\lambda 2$). However, this requires that a monetary value of the effect on an ES can be determined. It can be discussed if this is always necessary. The visual integration is easier because you avoid the step of monetary valuation and this can be sufficient for its main purpose: informing decision makers by giving a better insight in the different effects of the management options (not only the intended effects).

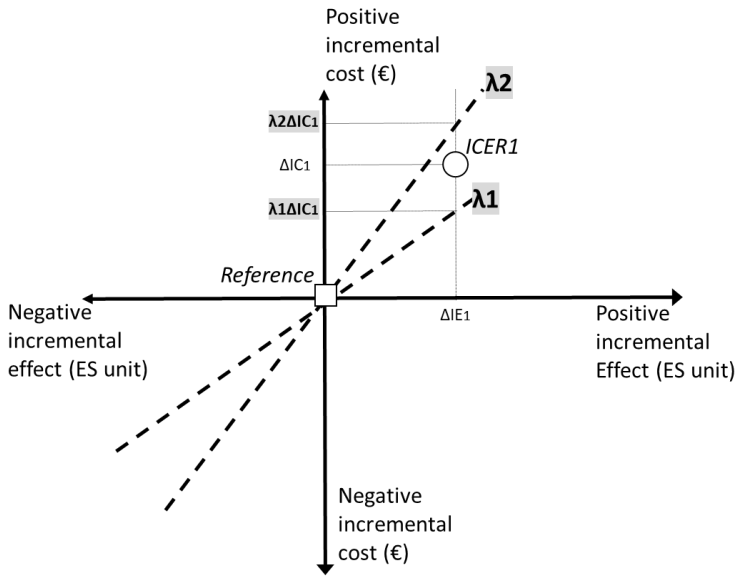


Figure 5: The relationship between the incremental cost effectiveness ratio (ICER) and net benefit can be studied with a willingness to pay line (λ). A different λ-line can determine the preference for a certain management alternative. In case of λ_1 (lower €/unit), the monetary value of the incremental effect ($\lambda_1 \Delta IE_1$) is lower than the incremental cost (ΔIC_1) and the reference is preferred over ICER1. In case of λ_2 (higher €/unit), the monetary value of the incremental effect ($\lambda_2 \Delta IE_1$) is larger than the incremental cost (ΔIC_1) and ICER1 is preferred over the reference.

ii. Selecting the most optimal management strategy

The most optimal strategy can differ depending on the ES that are targeted. When optimising for the ES navigability or flood protection, scenarios 1 and 3 are most cost-effective. This means that the alternative strategy for dredging is dominant both for navigability (much cheaper) and flood protection (reduction of the negative effect of traditional dredging). When optimising for water quality regulation or climate regulation, scenarios 2b and 3 are optimal because only these scenarios contain the FCA-CRT management measure which is the only measure in this analysis with a positive effect on these ES. Overall, this means that scenario 3 is the most optimal strategy for integrated management of the 4 considered ES. When including more ES in the analysis, finding one alternative that is most cost-effective for all ES will become more difficult with only a visual integration. The next step will be to determine the pareto options with an indifferent preference when all ES are considered equal or the ES can be weighted and added together in a multi-criteria analysis (Bryan 2010).

Our approach followed a cost minimization strategy to reach two targets at the least possible cost without taking into account possible budget restrictions. The most optimal scenario (S3) is one of the cheapest scenarios (together with S1, Figure 4). An alternative is a benefit maximization strategy per ES to generate the most benefits for a given budget (Balana et al. 2011). In the latter approach, the results will largely depend on the budget size (Duke et al. 2013). In our case, when budget restrictions would be lower than the total cost of scenario 3, other more cost-efficient management measures would be required or the objectives (that were now stated to detect the cost minimization strategy) could not be met. For each ES, the best benefit maximization strategy can be selected and the outcome can be visually compared for different ES. Treating each ES separately is inherent for the CEA method as each ES is assessed in a different unit. This allows to avoid the, often contested step, of assessing all natural, social and economic benefits in a monetary value which can imply that some benefits are excluded from the analysis when it cannot be expressed in a monetary value. Furthermore, the occurrence of both positive and negative effects

on different ES and for different stakeholders are not masked as is the case when summing all benefits in a cost-benefit analysis (Suzuki and Iwasa 2009).

4.4 Remaining challenges and way forward

A limited set of 4 ES was selected to be able to focus on the methodology and not on the complexity of adding more ES. Nevertheless, it is possible to extend this method for many other functions and ES. An important advantage of the presented method (in comparison to the mostly used but also strongly contested cost-benefits analysis) is that monetary valuation of the effects is not necessary and there is no risk of double counting or masking inequality since the effects are not added together in one net sum. Therefore, this method can also account for the impact of management measures on, for example, habitat and biodiversity. Being able to account for all environmental, social and economic effects is crucial when designing an integrated management plan for an ecosystem. Nevertheless, testing this method on a case-study revealed several remaining challenges.

Aspects related to the spatial and temporal scale of ecosystem management are not addressed in this analysis. A spatial analysis is advised to take into account geographical specifications of ecosystem management and its impact for ES benefits (Crossman and Bryan 2009, Wainger et al. 2010). It can be assumed that management measures implemented at different locations of the ecosystem can result in different effects for ES (e.g. depending on gradients such as salinity in an estuary). Furthermore, when specific targets are defined for different regions in the ecosystem this will add another level of complexity since certain management measures can be effective only for certain regions. Another aspect linked to the spatial scale, is the size of management measures. We assumed linearity with size in this assessment, but it is expected that one large project or several smaller projects can result in different effects for ES.

A time scale of 10 years is considered in the scenario analysis without discounting costs for this short period. Choosing the time span of the analysis will depend on for example whether short term or long term targets are considered or whether an ultimate objective or a milestone towards a long term objective is considered (Vogt-Schilb and Hallegatte 2014). A longer time period can be necessary because the effects of management measures can be delayed (e.g. due to processes such as ecological succession). However, making an analysis for a long time period can cause intertemporal complications such as changing targets or changing budgets over time (Duke et al. 2013).

Developing an integrated management strategy requires the assessment of relationships between management measures and between the impact for different ES (Sanon et al. 2012, Smith et al. 2013). However, in most cases the impact of a management measure is estimated independently of its impact for other ES and independently of the impact of other management measures. An intertidal areas along an estuary river, for example, generates several positive effects such as sediment burial (linked to habitat development and nutrient burial) and water storage (linked to flood protection). However, in case of high sedimentation, large volumes of sediment are stored which is good for nutrient burial but which decreases the water storage capacity. The latter is a disadvantage for flood protection. Therefore, the impact of intertidal areas could not be high for both functions. Furthermore, different investments that are implemented together could interact and strengthen or lessen the effect of other investments.

The case study in this paper was intended as exercise to test and examine different techniques for a cost-effectiveness analysis in combination with an ES assessment and by using available cost data and biophysical data. Due to data limitations, many assumptions and hypothetical considerations were necessary. This study shows that the available knowledge is still too limited

to make realistic estimates of the cost-effectiveness of management measures for different important ecosystem functions and services. In addition, the effects that are studied are often expressed in different units which makes comparison impossible. Despite some remaining challenges, this study demonstrates the added value of the approach to inform decision making in the development of an integrated management plan while taking into account the effects on both ecology and society.

5 Appendix 1

i. Cost data for the scenarios

- Traditional dredging: The cost for traditional dredging is 6270 million € which is the total cost of the 3th widening and deepening of the Scheldt estuary (1140 million € for capital and maintenance dredging and 5130 million € for disposal of dredged material). Different studies on beneficial disposal of dredged material, e.g. for habitat restoration and creation, reveal an average cost of 0.20-24 €/m^3 (Comoss et al. 2002, Yozzo et al. 2004, Boerema and Roose 2012, HPA 2012, Wasserman et al. 2013).
- Alternatives for dredging:
    - Nautical depth concept: no cost, but savings in dredging. When assuming that the nautical depth concept applies to all locations, this would mean that about half of the dredging can be avoided.
    - A sediment trap can be built in various sizes to trap incoming sediment and reduce sediment supply in the estuary. For the calculation, a sediment trap from which annually 10,000 m^3 can be dredged is assumed. To build the sediment trap, an initial dredging of 12,500 m^3 is required (trap efficiency is about 70-90%). By reducing the dredging activity to one location, the morphology of the river is much less affected. Therefore we assume that the mean high water level will not be influenced by a sediment trap. The cost to maintain a sediment trap is estimated based on the lower end of the cost range for dredging (1-5 €/m^3) because it is easier and more efficient to dredge at one location. The total cost for the sediment trap is 0.4 million €.
- Traditional dikes: This requires a heightening of the former safety dikes with 115 cm to the new Sigma height of 9.15, 9.5 or 10.35 m TAW depending on the location in the estuary, at a cost of 255 million € (IMDC et al. 2004, Broekx et al. 2011).
- Alternatives for dikes: The FCA is less expensive compared to the FCA-CRT but generates no additional benefits on the other ES.

ii. Impact of the scenarios on the navigation target

- **Target**: target from the 3th deepening of the Scheldt estuary
- Traditional dredging: The management strategy traditional dredging consists of capital and maintenance dredging to realise the target of the 3th deepening of Scheldt estuary and land disposal of the dredged material. To realise the target for navigability, 114 million m^3 should be dredged and disposed.
- Alternatives for dredging: For the management strategy alternatives for dredging, following management measures are included: fluid mud concept, sediment trap, traditional dredging for the remaining requirements and beneficial disposal of dredged material (e.g. underwater disposal, habitat nourishment).
    - The fluid mud concept (or nautical depth concept) takes into account that mud with a density up to 1250 kg/m^3 is still navigable and can be considered as part of

the required depth including the safety margin of 12.5% (Fettweis et al. 2011, Manson and Pinnington 2012, Kirby 2013). This provides an additional 2 to 3 m of nautical depth that should not be dredged (Kirby 2013). During the 3th Scheldt deepening about 3 to 4 m is removed to guarantee a depth of 14.7 m to reach the harbour of Antwerp. When assuming that the nautical depth concept applies to all locations, this would mean that about half of the dredging can be avoided.

  - A sediment trap can be built in various sizes to trap incoming sediment and reduce sediment supply in the estuary. For the calculation, a sediment trap from which annually 10,000 m^3 can be dredged is assumed. To build the sediment trap, an initial dredging of 12,500 m^3 is required (trap efficiency is about 70-90%).
  - Since the alternatives fluid mud concept and sediment trap are not sufficient to replace the entire dredging target, 50% of the traditional dredging is still required to have the same result as with the traditional dredging scenario. A small amount, 10,000 m^3, will be dredged annually from the sediment trap at a lower cost. By reducing traditional dredging with 50%, we assume correspondingly a reduction of 50% in the effect on the ES. For the period of 10 years, this means that only 57,012,500 m^3 dredged material should be disposed.
  - As alternative for the traditional land disposal, beneficial strategies such as underwater disposal to fill deep areas in the channel or habitat nourishment are assumed.

- Traditional dikes: no impact on navigability
- Alternatives for dikes: The FCA-CRT will generate additional effects for navigability (annual burial of 135,000 ton in the new area (IMDC et al. 2004), or 270,000 m^3 of sediment calculated based on an average bulk density of 500 kg m^{-3} in the Scheldt (Temmerman et al. 2004)).

iii. Impact of the scenarios on the flood safety target

- **Target**: target from the Sigmaplan (flood protection management plan for the Flemish part of the Scheldt estuary)
- Traditional dikes: The management strategy traditional dikes include the heightening and strengthening of dikes to a safety level of T4000 which means a protection against a storm with an occurrence of once in 4,000 years. This requires a heightening of the former safety dikes with 115 cm to the new Sigma height of 9.15, 9.5 or 10.35 m TAW depending on the location in the estuary (IMDC et al. 2004, Broekx et al. 2011).
- Alternatives for dikes: The management strategy alternatives for traditional dikes consist of two sub-alternatives: a) flood control area (FCA) and b) flood control area with controlled reduced tide (FCA-CRT). For both alternatives an area of 2800 ha of flood control area can also realise a safety level of T4000 and is hence equivalent to 115 cm increase in dike height (IMDC et al. 2004).
- Traditional dredging: Historic data from the Scheldt shows an increase in the mean high water level of 55 mm between 1966 and 1982 after a deepening by 4 m and 100 mm between 1997 and 1999 after a deepening by 1.4 m (Cox et al. 2003). The order of magnitude is comparable with the annual sea level rise (0.05 m/y), an extrapolation of the locally observed average rate of MHWL rise since 1930 (Temmerman et al. 2004). Hence, it is expected that dredging causes a substantial negative effect on the service flood protection. However, this is a very simplistic assumption and should be interpreted only as indication of a potential negative effect. The direct cause-effect relationship between the increase in MHWL and the deepening is not generic and several other causes of the MHWL increase are involved.
- Alternatives for dredging:

- With the beneficial use of dredged material, the morphology of the river can be adapted with potential positive effects on flood protection.

iv. Impact of the scenarios on water quality regulation

- Traditional dredging: For the service water quality regulation, dredging has a negative effect by remobilising nutrients and contaminants and causing higher turbidity. However, dredging can be used to remove highly contaminated sediment and prevent contamination of the river water (AKWA 2004, Zhong et al. 2010). Since data is lacking to estimate the negative effect of dredging on water quality, this effect is not taken into consideration. However, when discussing the results a qualitative assessment is added for the effects that are not quantified. Land disposal has normally no effect on water quality, although there is a risk for leaching contaminants.
- Alternatives for dredging: Relocating the material in the system might re-suspend stored carbon and nitrogen which is negative for the service water quality regulation (nitrogen removal).
- Traditional dikes: no impact on water quality
- Alternatives for dikes: The FCA-CRT will generate additional effects for water quality regulation (a total of 785 ton nitrogen removal per year in the new area).

v. Impact of the scenarios on climate regulation

- Traditional dredging: Dredging and land disposal have no direct effect on the service climate regulation, except that it generates emissions from machinery and fuel use (Bates et al. 2015). This does however not affect service delivery but rather enhances the demand for the service.
- Alternatives for dredging: Relocating the material in the system might re-suspend stored carbon and nitrogen which is negative for the service climate regulation (carbon burial).
- Traditional dikes: no impact on climate regulation
- Alternatives for dikes: The FCA-CRT will generate additional effects for climate regulation (annual sequestration of 1.5 ton C/ha/y or a total of 4,000 ton carbon per year in the new area) (IMDC et al. 2004).

646 # 6 References

Adame, M. F., V. Hermoso, K. Perhans, C. E. Lovelock, and J. A. Herrera-Silveira. 2014. Selecting Cost-Effective Areas for Restoration of Ecosystem Services. Conservation Biology:n/a-n/a.

AKWA. 2004. Bagger: het onzichtbare goud? Hoofdnota Maatschappelijke Kosten-Batenanalyse Waterbodems. Advies en Kenniscentrum Waterbodems.

Arcadis-Technum. 2004. Strategisch Milieueffectenrapport, Hoofdrapport, Ontwikkelingsschets 2010 Schelde-estuarium. Page 259.

Atkins, J. P., D. Burdon, M. Elliott, and A. J. Gregory. 2011. Management of the marine environment: Integrating ecosystem services and societal benefits with the DPSIR framework in a systems approach. Marine Pollution Bulletin **62**:215-226.

Balana, B. B., A. Vinten, and B. Slee. 2011. A review on cost-effectiveness analysis of agri-environmental measures related to the EU WFD: Key issues, methods, and applications. Ecological Economics **70**:1021-1031.

Bates, M. E., C. Fox-Lent, L. Seymour, B. A. Wender, and I. Linkov. 2015. Life cycle assessment for dredged sediment placement strategies. Science of the Total Environment **511**:309-318.

Boerema, A., L. Geerts, L. Oosterlee, S. Temmerman, and P. Meire. 2016. Ecosystem service delivery in restoration projects: the effect of ecological succession on the benefits of tidal marsh restoration. Ecology and Society **21**:10.

Boerema, A., and P. Meire. 2017. Management for estuarine ecosystem services: A review. Ecological Engineering **98**:172-182.

Boerema, A., A. J. Rebelo, M. B. Bodi, K. J. Esler, and P. Meire. 2017. Are ecosystem services adequately quantified? Journal of applied ecology **54**:358-370.

Boerema, A., and F. Roose. 2012. 'TIDE pilot 1: Large scale execution "plaatrandstortingen" (2010), relocation of dredged sediment to shallow water areas near sandbars' (Schelde estuary). Measure analysis in the framework of the Interreg IVB project TIDE. Measure 22. Antwerp Port Authority (APA), Antwerp, Belgium.

Breen, B., and S. Hynes. 2014. Shortcomings in the European principles of Integrated Coastal Zone Management (ICZM): Assessing the implications for locally orientated coastal management using Biome Portfolio Analysis (BPA). Marine Policy **44**:406-418.

Broekx, S., S. Smets, I. Liekens, D. Bulckaen, and L. De Nocker. 2011. Designing a long-term flood risk management plan for the Scheldt estuary using a risk-based approach. Natural hazards **57**:245-266.

Bryan, B. A. 2010. Development and application of a model for robust, cost-effective investment in natural capital and ecosystem services. Biological Conservation **143**:1737-1750.

Comello, S. D., G. Maltais-Landry, B. R. Schwegler, and M. D. Lepech. 2014. Firm-level ecosystem service valuation using mechanistic biogeochemical modeling and functional substitutability. Ecological Economics **100**:63-73.

Comoss, E. J., D. A. Kelly, and H. Z. Leslie. 2002. Innovative erosion control involving the beneficial use of dredge material, indigenous vegetation and landscaping along the Lake Erie Shoreline. Ecological Engineering **19**:203-210.

Compernolle, T., S. Van Passel, N. Weyens, J. Vangronsveld, L. Lebbe, and T. Thewys. 2012. Groundwater remediation and the cost effectiveness of phytoremediation. International Journal of Phytoremediation **14**:861-877.

Convertino, M., K. M. Baker, J. T. Vogel, C. Lu, B. Suedel, and I. Linkov. 2013. Multi-criteria decision analysis to select metrics for design and monitoring of sustainable ecosystem restorations. Ecological indicators **26**:76-86.

Cox, R., R. A. Wadsworth, and A. G. Thomson. 2003. Long-term changes in salt marsh extent affected by channel deepening in a modified estuary. Continental Shelf Research **23**:1833-1846.

Crossman, N. D., and B. A. Bryan. 2009. Identifying cost-effective hotspots for restoring natural capital and enhancing landscape multifunctionality. Ecological Economics **68**:654-668.

Dawson, R. J., T. Ball, J. Werritty, A. Werritty, J. W. Hall, and N. Roche. 2011. Assessing the effectiveness of non-structural flood management measures in the Thames Estuary under conditions of socio-economic and environmental change. Global Environmental Change **21**:628-646.

De Wit, K., and M. Sas. 2007. Milieueffectrapport Verruiming vaargeul Beneden-Zeeschelde en Westerschelde. Achtergronddocument Baggeren en storten. Page 169. Technische Scheldecommissie.

Drechsler, M., F. Wätzold, K. Johst, H. Bergmann, and J. Settele. 2007. A model-based approach for designing cost-effective compensation payments for conservation of endangered species in real landscapes. Biological Conservation **140**:174-186.

Duke, J. M., S. J. Dundas, and K. D. Messer. 2013. Cost-effective conservation planning: Lessons from economics. Journal of Environmental Management **125**:126-133.

Fettweis, M., M. Baeye, F. Francken, B. Lauwaert, D. Van den Eynde, V. Van Lancker, C. Martens, and T. Michielsen. 2011. Monitoring the effects of disposal of fine sediments from maintenance dredging on suspended particulate matter concentration in the Belgian nearshore area (southern North Sea). Marine Pollution Bulletin **62**:258-269.

Gómez-Baggethun, E., and M. Ruiz-Pérez. 2011. Economic valuation and the commodification of ecosystem services. Progress in Physical Geography: Earth and Environment **35**:613-628.

Halpern, B. S., C. White, S. E. Lester, C. Costello, and S. D. Gaines. 2011. Using portfolio theory to assess tradeoffs between return from natural capital and social equity across space. Biological Conservation **144**:1499-1507.

Helm, D., and C. Hepburn. 2012. The economic analysis of biodiversity: an assessment. Oxford Review of Economic Policy **28**:1-21.

HPA. 2012. 'Underwater relocation area Medemrinne Ost': Measure analysis 02 in the framework of the Interreg IVB project TIDE. Hamburg Port Authority, Federal Administration for Waterways and Navigation, Germany.

IMDC, Ecolas, Grontmij, and Tijdelijke Vereniging Resource Analysis. 2004. Milieueffectrapportage voor de actualisatie van het Sigmaplan: niet-technische samenvatting. Ministerie van de Vlaamse Gemeenschap. Afdeling Zeeschelde: Antwerpen. 128 pp.

Interwies, E., D. Borchardt, A. Kraemer, N. Kranz, B. Görlach, S. Richter, J. Willecke, and T. Dworak. 2004. Basic principles for selecting the most cost-effective combinations of measures for inclusion in the programme of measures as described in Article 11 of the Water Framework Directive. Federal Environment Agency, Germany.

Jacobs, S., K. Wolfstein, W. Vandenbruwaene, D. Vrebos, O. Beauchard, T. Maris, and P. Meire. 2015. Detecting ecosystem service trade-offs and synergies: A practice-oriented application in four industrialized estuaries. Ecosystem Services **16**:378-389.

Kirby, R. 2013. Managing industrialised coastal fine sediment systems. Ocean & Coastal Management **79**:2-9.

Labiosa, W. B., W. M. Forney, A. M. Esnard, D. Mitsova-Boneva, R. Bernknopf, P. Hearn, D. Hogan, L. Pearlstine, D. Strong, H. Gladwin, and E. Swain. 2013. An integrated multi-criteria scenario evaluation web tool for participatory land-use planning in urbanized areas: The Ecosystem Portfolio Model. Environmental Modelling & Software **41**:210-222.

MacLeod, M., D. Moran, V. Eory, R. M. Rees, A. Barnes, C. F. E. Topp, B. Ball, S. Hoad, E. Wall, A. McVittie, G. Pajot, R. Matthews, P. Smith, and A. Moxey. 2010. Developing greenhouse gas marginal abatement cost curves for agricultural emissions from crops and soils in the UK. Agricultural Systems **103**:198-209.

Macmillan, D. C., D. Harley, and R. Morrison. 1998. Cost-effectiveness analysis of woodland ecosystem restoration. Ecological Economics **27**:313-324.

Manson, S., and N. Pinnington. 2012. 'Deployment of Mudbugs': Measure analysis 32 in the framework of the Interreg IVB project TIDE. Environment Agency, UK.

OVAM. 2003. Overzicht en evaluatie van de verwerkingstechnieken baggerspecie. Openbare Afvalstoffenmaatschappij voor het Vlaamse Gewest, D/2003/5024/08, Mechelen / Belgium.

Pouwels, I. H. M., H. G. Wind, and V. J. Witter. 1995. Multiobjective decision-making in integrated water management. Physics and chemistry of the earth **20**:221-227.

Prato, T. 2007. Selection and evaluation of projects to conserve ecosystem services. Ecological Modelling **203**:290-296.

Sanon, S., T. Hein, W. Douven, and P. Winkler. 2012. Quantifying ecosystem service trade-offs: The case of an urban floodplain in Vienna, Austria. Journal of Environmental Management **111**:159-172.

Seppelt, R., S. Lautenbach, and M. Volk. 2013. Identifying trade-offs between ecosystem services, land use, and biodiversity: a plea for combining scenario analysis and optimization on different spatial scales. Current Opinion in Environmental Sustainability **5**:458-463.

Smets, S., S. Broekx, D. Bulckaen, and L. De Nocker. 2005. Sigmaplan: societal cost-benefit analysis (SCBA) (in Dutch). Projectconsortium SCBA Sigmaplan. Resource Analysis, Antwerp.

Smith, P., M. R. Ashmore, H. I. J. Black, P. J. Burgess, C. D. Evans, T. A. Quine, A. M. Thomson, K. Hicks, and H. G. Orr. 2013. REVIEW: The role of ecosystems and their management in regulating climate, and soil, water and air quality. Journal of applied ecology **50**:812-829.

Suzuki, Y., and Y. Iwasa. 2009. Conflict between groups of players in coupled socio-economic and ecological dynamics. Ecological Economics **68**:1106-1115.

Temmerman, S., G. Govers, S. Wartel, and P. Meire. 2004. Modelling estuarine variations in tidal marsh sedimentation: response to changing sea level and suspended sediment concentrations. Marine Geology **212**:1-19.

Tijdelijke Vereniging Resource Analysis, IMDC, Grontmij, Ecolas, and VITO. 2004. Milieueffectrapportage voor de actualisatie van het Sigmaplan, niet-technische samenvatting.

Tóth, S. F., G. J. Ettl, N. Könnyű, S. S. Rabotyagov, L. W. Rogers, and J. M. Comnick. 2013. ECOSEL: Multi-objective optimization to sell forest ecosystem services. Forest Policy and Economics **35**:73-82.

Turner, R. K., J. Paavola, P. Cooper, S. Farber, V. Jessamy, and S. Georgiou. 2003. Valuing nature: lessons learned and future research directions. Ecological Economics **46**:493-510.

Vogt-Schilb, A., and S. Hallegatte. 2014. Marginal abatement cost curves and the optimal timing of mitigation measures. Energy policy **66**:645-653.

Wainger, L. A., D. M. King, R. N. Mack, E. W. Price, and T. Maslin. 2010. Can the concept of ecosystem services be practically applied to improve natural resource management decisions? Ecological Economics **69**:978-987.

Wasserman, J. C., S. R. Barros, and G. B. A. Lima. 2013. Planning dredging services in contaminated sediments for balanced environmental and investment costs. Journal of Environmental Management **121**:48-56.

www.tide-toolbox.eu. The TIDE toolbox is a result of the INTERREG IVB-Project "Tidal River Development" TIDE. The section management measures (http://www.tide-toolbox.eu/measures/) gives you an overview of 42 measures which have been carried out or are planned in the TIDE estuaries.

Yang, W. 2011. A multi-objective optimization approach to allocate environmental flows to the artificially restored wetlands of China's Yellow River Delta. Ecological Modelling **222**:261-267.

Yozzo, D. J., P. Wilber, and R. J. Will. 2004. Beneficial use of dredged material for habitat creation, enhancement, and restoration in New York–New Jersey Harbor. Journal of Environmental Management **73**:39-52.

Zhang, J., M. Fu, Z. Zhang, J. Tao, and W. Fu. 2014. A trade-off approach of optimal land allocation between socio-economic development and ecological stability. Ecological Modelling **272**:175-187.

Zhong, J., C. Fan, L. Zhang, E. Hall, S. Ding, B. Li, and G. Liu. 2010. Significance of dredging on sediment denitrification in Meiliang Bay, China: A year long simulation study. Journal of Environmental Sciences **22**:68-75.