Tidal River Development

Manual for the valuation of ecosystem services in estuaries

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The Interreg IVB North Sea Region Programme

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1 Introduction

1.1 Aim

The objective of this guidance document is to describe a practical methodology for economic valuation of ecosystem services in estuaries for estuarine managers. Based on the best available data and current insights we explain how and when a (monetary) valuation of ecosystem services in estuaries can be performed. A schematic road map (page 21) sets out the different steps to perform ecosystem service valuation in any given estuary. The proposed methods are demonstrated by Illustrations mostly from the Scheldt estuary.

The guidance document contains:

- An introduction to the concept of ecosystem services
- An introduction to monetary and non-monetary valuation methods with their strengths and weaknesses
- A list of ecosystem services relevant for estuaries, a description of the ecosystem service and the processes supporting the delivery of the service
- Methodologies to value services for a selection of ecosystem services. We follow the pyramid-like approach of Gantioler et al. (2010) existing of three steps :
    1. Identification = providing a qualitative score per habitat
    2. Quantification = describing the delivery of the ecosystem service in bio-physical terms.
    3. Valuation = estimating the value in monetary terms
- Information on how the necessary data to perform a valuation can be collected

We also provide guidelines on how to integrate the different estuarine ecosystem services and estimate the socio-economic value delivered by estuaries.

1.2 TIDE

This assignment is part of the European Interreg IV-B NSR project TIDE (Tidal River Development), in which partners with experiences in 4 estuaries, work jointly to design an integrated management of estuaries by exchange of experiences and knowledge. TIDE considers in the North Sea Region (NSR) tidally influenced estuaries, in which important fairways to seaports are located and which are exposed to dynamic sediment processes. Within these estuaries important ecosystem services are provided by intertidal and shallow estuarine habitats. These ecosystem services which have direct and indirect economical benefits, are under pressure if sustainable maintenance of ecologically important areas is not considered. At the same time decision-makers dealing with the management of these estuaries are faced with an increasingly challenging legal and global economic framework.

TIDE aims to lead the path towards a more sustainable and effective use of large scale investments made into mitigation and compensation measures in NSR estuaries by applying for the first time a unified ecosystem approach to guide the process of integrated participatory management planning. TIDE aims to improve the effectiveness of European, national and regional policy, to provide instruments for regional development and to make an essential contribution towards a more sustainable and effective use of investments into estuaries - since the planning of policies will be based on a unified assessment concept and integrated management planning procedures.

2 Theoretical basis

2.1 The ecosystem services approach

Our economy, health and survival depend entirely, albeit often indirectly, upon natural resources (MEA 2005). Humankind benefits from a multitude of resources and processes that are supplied by natural ecosystems. Collectively, these benefits are known as ecosystem goods and services, further in the document shortened to ecosystem services. Together with population growth and growing per capita consumption rates, the demand for those resources increased, and the impact of this consumption pattern became more and more clear: natural resources, supposed to be infinitely available and freely available, are becoming scarce or degraded. Health problems, natural disasters and high costs for technical replacement of natural regulating functions have boosted the need to adopt a broader view and strategy on resource use.

While scientists and environmentalists have discussed ecosystem services for decades, these services were popularized and their definitions formalized by the United Nations 2005 Millennium Ecosystem Assessment (MEA), a four-year study involving more than 1,300 scientists worldwide. Ecosystem services are typically categorized in provisioning, regulating, cultural and supporting services (Figure 1). Provisioning services are the products obtained from ecosystems such as food, fresh water, wood, fiber, genetic resources and medicines. Regulating services are defined as the benefits obtained from the regulation of ecosystem processes such as climate regulation, natural hazard regulation, water purification and waste management, pollination or pest control. Cultural services include non-material benefits that people obtain from ecosystems such as spiritual enrichment, intellectual development, recreation and aesthetic values. These services are generated, supported and ensured by ecosystems in all their diversity (supporting services). Later classifications add sometimes another category: habitat services (e.g. TEEB 2010). This category was added to highlight the importance of ecosystems to provide habitat to migratory species (e.g. as nurseries) and gene-pool ‘protectors’.

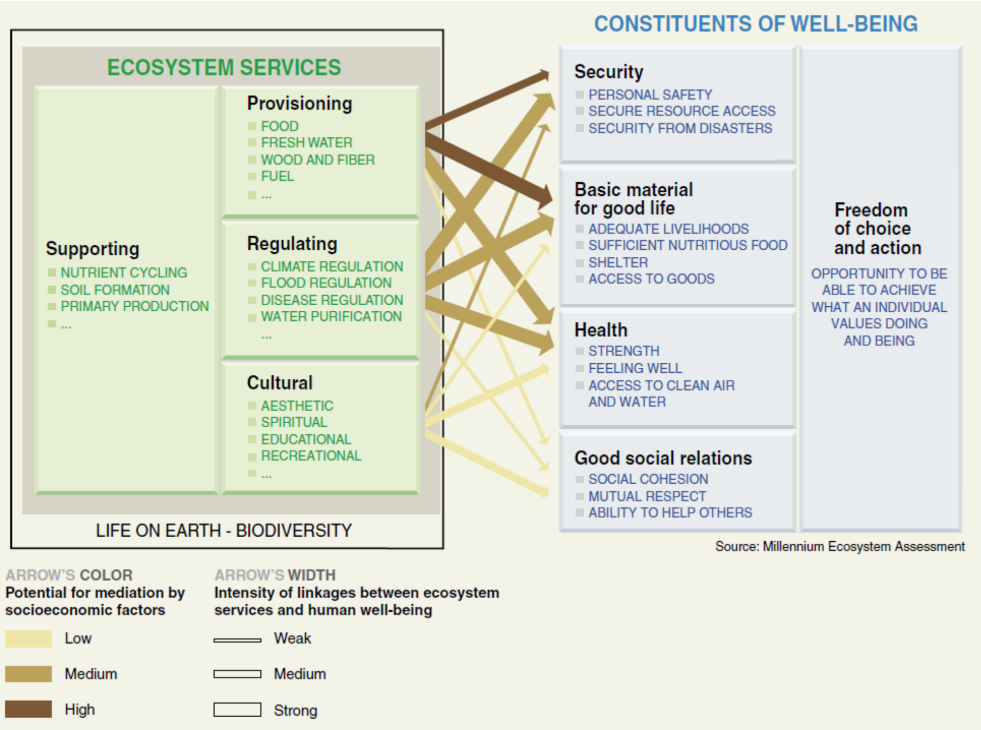

Figure 1: Link between ecosystem services and human well-being (MEA, 2005)

A benefit to human well-being, generated by an ecosystem service, often requires a(n) (technological) investment, for instance drinking water requires a pumping installation and a distribution system. The service itself originates from an ecosystem function. A function is the set of physical, chemical and biological structures and processes which eventually produce the service. Sometimes, several more or less separated functions are appropriate to describe the supply. Often, intermediate services underlying the final service are therefore distinguished. Structures and processes are not exclusively linked to one single service, they contribute to several services, sometimes exhibiting trade-offs. As such, every single service is connected to an intertwined web of structures and processes, finally supported/insured by the resilience of the entire ecosystem. Essential functions in the ecosystem, such as natural population dynamics, nutrient cycling, are therefore called 'supporting services', covering all of the diversity within the ecosystem.

Sustaining the different flows of ecosystem services requires a good understanding of how ecosystems function, provide services and contribute to well-being. A way of representing the logic that underlies the delivery of ecosystem services is shown in Figure 2. A distinction is made between ecological structures and processes created or generated by living organism and the benefits that people eventually derive. In the real world the links are not as simple and linear as this. There is a kind of cascade linking the two ends of a ‘production chain’ (TEEB, 2010).

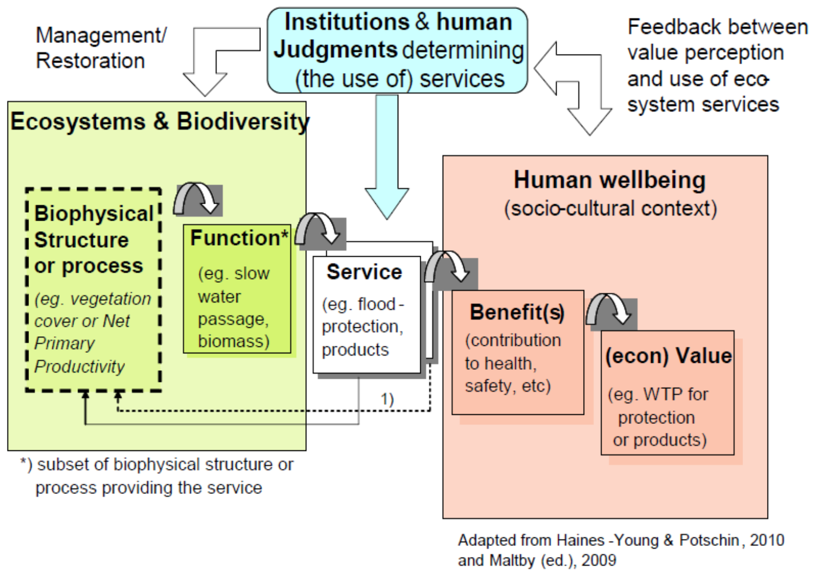

Figure 2: From ecosystem to human well-being (TEEB, 2010)

2.2 Valuing ecosystem services

2.2.1 What is valuation?

Apart from describing and understanding the functions behind ecosystem services, ecosystem services are often valued in order to make choices. De Groot et al. (2010) equate ‘value’ to ‘importance’. This value or importance is not easy to determine. As Maris and Bechet (2010) point out, values are contextual, relative to a certain place, a certain time, and a certain group of people facing a problem and engaged in collective action. According to Costanza (2000), value ultimately originates in the set of goals to which a society aspires. Valuation can be defined as the act of assessing value, or an appraisal of the value. Valuation can thus refer to assessing a monetary value or a price but also an estimation or appreciation of ‘worth’, in the broad meaning of the latter word. Moreover, Costanza (2000) recognizes that, in order to conduct appropriate valuation of ecosystem services, we need to consider a broader set of goals that include ecological sustainability and social fairness, along with the traditional economic goal of efficiency. According to Costanza and Folke (1997), valuation of ecosystem services occurs in three ways: ecological sustainability (S-value), economic efficiency (E-value) and social fairness (F-value). Although robust methods are still under development, analysis of policies using the ES-concept has the strength of at least visualizing and demonstrating the sustainability and fairness issues/problems, which is often sufficient to inform and improve current resource management policies.

In economics the concept of “value” is always associated with scarcity and trade-offs i.e. something only has (economic) value if we are required and willing to give up something to get or enjoy it. This concept of valuation is thus anthropocentric in nature. Economic valuation usually attempts to measure the value of ecosystem services in monetary terms, in order to provide a common metric in which to express the benefits of the variety of services provided by ecosystems. This explicitly does not mean that only monetary sacrifices, or only services that generate monetary benefits, are taken into consideration. What matters is that people are willing to make tradeoffs.

Economic valuation means demonstrating the value of ecosystem services in economic terms being either money (monetary valuation) or another metric (non-monetary valuation). This guidance document focuses on the monetary valuation of ecosystem services.

2.2.2 Why valuation?

Ecosystems are still degrading. One of the reasons is the failure to account for the full value of ecosystems and biodiversity for human societies in decision making. The economic valuation of ecosystem services presents a promising approach to highlight the relevance of ESS to society and the economy, to serve as an element in the development of cost-effective policy instruments for nature restoration and management and use in impact assessments in cost-benefit analysis. Economic valuation may also be useful in developing payments for ecosystem services (Markandya 2011)

Economic valuation can be particularly effective in enabling informed trade-offs in cost-benefit analyses, where the focus lies on assessing the marginal change in the provision of an ecosystem service relative to the provision of the same service in an alternative scenario.

2.2.3 Total economic value, use and non use values

The goal of economic valuation is to value the so called total economic value (TEV) of an ecosystem to provide information on changes in the value of ecosystem services that result from policy decisions or other human activities. In other words, economic valuation should be set within the context of contrasting scenarios recognizing that both the values of ecosystem services and the costs of actions can be measured as a function of changes between alternative options.

The Total Economic Value consists of use value and non-use value (Figure 3). By definition, **use values** are derived from the actual use of the environment. They are sometimes further divided into two categories: (a) *Direct use value*, related to the benefits obtained from direct use of ecosystem services. Such use may be extractive, which entails consumption (for instance of food and raw materials), or non-extractive use (e.g., aesthetic benefits from landscapes). (b) *Indirect use values* are usually associated with regulating services, such as air quality regulation or erosion prevention, which can be seen as public services which are generally not reflected in market transactions. The **option value** is defined as the value of future use of known and unknown ecosystem services. **Non-use values** on the other hand are non-instrumental. They reflect satisfaction that individuals derive from the knowledge that biodiversity and ecosystem services are maintained and that other people have or will have access to them (Kolstad, 2000). In the first case, non-use values are usually referred to as *existence values*, while in the latter they are associated with *altruist values* (in relation to intra-generational equity concerns) or *bequest values* (when concerned with inter-generational equity) (TEEB 2010).

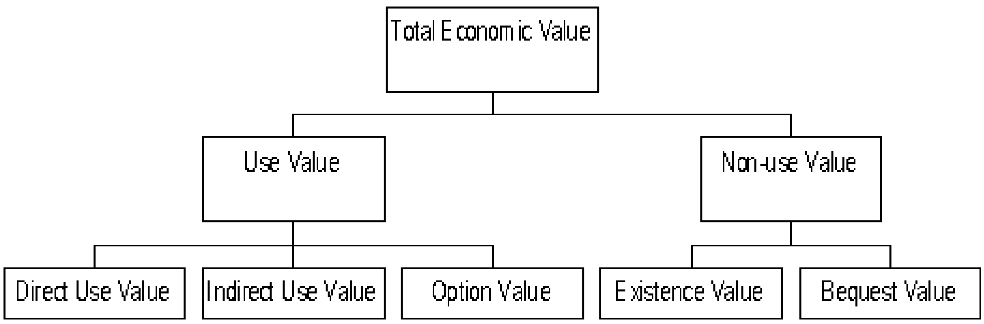

Figure 3: Total Economic Value Framework

Economic valuation cannot value everything: not all benefits provided by ecosystem services are fully translatable into economic terms. E.g. some ecological values such as the value of one species to the survival of another species. (Farber et al 2002). Therefore, it should be used to complement and not substitute other legitimate reasoning to biodiversity conservation.

2.2.4 Valuation methods

A variety of approaches can be used to estimate values of ecosystem services. They fall in two main categories: monetary and non-monetary methods. Monetary methods try to express all values in monetary terms (€).The non-monetary approaches are more aimed towards ranking of the importance of services based on group decisions and consensus.

The monetary methods exist of two groups: techniques that estimate economic values - valuation approaches, and techniques that produce estimates equivalent to prices - pricing approaches. It is important to know that the price of a good or service and its economic value are distinct and can differ greatly: pricing approaches are never able to capture the total value or consumer surplus.

→ *Monetary valuation methods*

Economists have a toolbox to monetize goods and services that ecosystems can deliver, and the appropriate tools depend on the characteristics of the goods or services (see Brouwer 2000; and overviews made in e.g. Freeman 2003; Champ et al. 2003; Hanley and Barbier 2009).

An overview is given in table 1.

Table 1: Valuation methods applied to ecosystem services

Valuation method	Value types	Overview of method	Common types of applications	Examples of ecosystem services valued	Example studies
Adjusted market prices	Use	Market prices adjusted for distortions such as taxes, subsidies and non-competitive practices.	Food, forest products, R&D benefits.	Crops, livestock, multi-purpose woodland, etc.	Bateman et al., (2003); Godoy et al., (1993)
Production function methods	Use	Estimation of production functions to isolate the effect of ecosystem services as inputs to the production process.	Environmental impacts on economic activities and livelihoods, including damage costs avoided, due to ecological regulatory and habitat functions	Maintenance of beneficial species; maintenance of arable land and agricultural productivity; support for aquaculture; prevention of damage from erosion and siltation; groundwater recharge; drainage and natural irrigation; storm protection; flood mitigation	Ellis and Fisher (1987); Barbier (2007).
Damage cost avoided	Use	Calculates the costs which are avoided by not allowing ecosystem services to degrade.	Storm damage; supplies of clean water; climate change.	Drainage and natural irrigation; storm protection; flood mitigation	Badola and Hussain (2005); Kim and Dixon (1986).
Averting behaviour	Use	Examination of expenditures to avoid damage	Environmental impacts on human health	Pollution control and detoxification	Rosado, et al., (2000).
Revealed preference methods	Use	Examine the expenditure made on ecosystem related goods (e.g. travel costs; property prices in low pollution areas).	Recreation; environmental impacts on residential property and human health.	Maintenance of beneficial species, productive ecosystems and biodiversity; storm protection; flood mitigation; air quality, peace and quiet, workplace risk.	See Bockstael and McConnell (2006) for the travel cost method and Day et al., (2007) for hedonic pricing.
Stated preference methods	Use and non-use	Uses surveys to ask individuals to make choices between different levels of environmental goods at different prices to reveal their willingness to pay for those goods	Recreation; environmental quality, impacts on human health, conservation benefits.	Water quality, species conservation, flood prevention, air quality, peace and quiet.	See Carson et al., (2003) for contingent valuation and Adamowicz et al., (1994) for discrete choice experiment approach.

From Bateman et al, 2010: adapted from De Groot et al., 2002; Heal et al., 2005; Barbier, 2007; Bateman 2009 and Kaval, 2010.

Estimating economic values for provisioning services (as the production of food, materials...) (so called direct use values), would seem fairly straightforward. These services are largely traded on markets and have a **market price**. This is somewhat deceptive as there are a number of limitations to market prices. Markets are often distorted (monopolies, subsidies...). If possible we need to take market distortions into account and correct the existing market prices.

Methods to value regulating and cultural services that are not sold on a market often require a number of assumptions to hold as well as copious amounts of data and intensive statistical analysis. Probably the most serious problem facing robust valuation of ecosystem services are gaps in our understanding of the underpinning science relating those services to the production of human well-being.

Regulating services are mostly valued through avoided (damage) costs (costs that we would have incurred if the service was absent or costs of replacing a service with man-made systems) e.g. avoided damage costs for flooding or avoided investment costs in wastewater treatment to estimate the value of water quality. The major underlying assumptions of these approaches are that the nature and extent of physical damage expected is predictable (there is an accurate damage function available) and that the costs to replace or restore damaged assets can be estimated with a reasonable degree of accuracy.

Another method that may be used is the averting behaviour method. This approach is similar to the travel cost method and hedonic pricing, but it differs as it uses individual behaviour to avoid negative intangible impacts as a conceptual base. For example, people buy goods such as safety helmets to reduce accident risk, and double-glazing to reduce traffic noise, and in doing so reveal their valuation of these bads. However, the situation is complicated (again) by the fact that these market goods might have more benefits than simply that of reducing an intangible bad. This method is not a widely used methodology and is limited to cases where households spend money to offset environmental hazards/nuisances. Appropriate data may be difficult to obtain (Hadley et al. 2011).

Cultural services such as amenity values, recreation values etc. are mostly valued through revealed preferences or stated preferences techniques.

Revealed preferences methods are the hedonic pricing method and the travel cost method.

Hedonic pricing is based on the fact that the prices paid for goods or services that have certain environmental attributes differ depending on those attributes. Thus, a house in a clean environment will have a higher market value than an otherwise identical house in a polluted neighbourhood. Hedonic price analysis compares the prices of similar goods to extract the implicit value ("shadow price") that buyers place on the environmental attributes. This method assumes that markets are transparent and work reasonably well. It would not be applicable where markets are distorted by policy or market failures. Moreover, this method requires a very large number of observations, is very data intensive and statistically complex to analyse. Its applicability is also limited to environmental attributes. The advantage of this method is that it is a well established technique and is based on actual observed behaviour, which makes it less controversial.

The **travel cost** method enables the economic value of recreational use (an element of direct use value) for a specific site to be estimated. The method requires that the costs incurred by individuals travelling to recreation sites - in terms of both travel expenses (fuel, fares etc.) and time (e.g. foregone earnings) - is collected. The basic assumption is that these costs of travel serve as a proxy for the recreational value of visiting a particular site. The advantage of the method is that it is a well established technique and is based on actual observed behaviour. Disadvantages are that it is only applicable to recreational sites, is difficult to account for the possible benefits derived from travel and multipurpose trips, is very resource intensive and statistically complex to analyse.

Contingent valuation is an example of a stated preference technique. It is carried out by asking consumers directly about their WTP to obtain an environmental service (or, in some circumstances, their willingness-to-accept). A detailed description of the service and how it will be delivered is provided. The valuation can be obtained in a number of ways, such as asking respondents to name an amount (classical CV), asking them whether they would pay a specific amount (dichotomous or polychotomous choice) or having them choose from several options (choice modelling). By phrasing the question appropriately, CV can be used to value any environmental benefit. Moreover, since it is not limited to deducing preferences from available data, it can be targeted to address specific changes in benefits that a particular change in ecosystem condition might cause.

Because of the need to describe the service in detail, interviews in CV surveys are time-consuming. In designing CV surveys it is important to identify the relevant population to ensure that the sample is representative, and to pre-test the questionnaire to avoid bias. A potentially important limitation when applying these methods to ecosystem services is that respondents cannot make informed choices if they have a limited understanding of the issue in question. Choosing the right approach to improve the sample group's understanding of biological complexity and the question at hand without biasing respondents, is a challenge for stated preference methods.

Choice modelling consists of asking respondents to choose their preferred option from a set of alternatives where the alternatives are defined by a set of attributes (including price). The alternatives are designed so that the respondent's choice reveals the marginal rate of substitution between the attributes and the item that is trade off (for example money). Choice modelling has several advantages. One advantage is that the control of the stimuli is in the experimenter's hand, as opposed to the low level of control generated by real market data. Second, the control of the design yields greater statistical efficiency. Third, the attribute range can be wider than found in market data. The method also minimizes some of the technical problems (such as strategic behaviour of respondents) that are associated with CV. The disadvantages associated with the technique are that the responses are hypothetical and therefore suffer from problems of hypothetical bias (similar to CV) and the choices can be complex when there are many attributes and alternatives. The econometric analysis of the data generated by choice modelling is also relatively complex.

A final category of approach is **benefits transfer** (BT), which refers to applying results of previous environmental valuation studies to new decision-making contexts. Benefits transfer is commonly defined as the transposition of monetary environmental values estimated at one site (study site) to another site (policy site). The study site refers to the site where the original study took place, while the policy site is a new site where information is needed about the monetary value of similar benefits. The most important reason for using previous research results in new policy contexts is that it saves a lot of time and money. BT 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 benefit transfer studies is that most existing stated preference 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. Instead of transferring a single value, another approach is to perform a regression-based value function transfer based on either a single or multiple studies (Brouwer 2000). In the latter case a meta-regression model is estimated. Meta-analysis of existing valuation studies estimates a function that relates the economic value to site, sample and study characteristics, and uses this function in benefit transfer to estimate economic values of non-surveyed areas. A difficulty in using this method is the multitude of original studies: Differences in range (changes from reference to target levels), spatial and temporal scale, and the number of explanatory variables (Brouwer and Spaninks 1999) may affect the suitability of processing valuation studies into the meta-analysis.

The value function based on a single study is estimated from the data of one survey using characteristics of the site, population characteristics and should also include spatial characteristics.

→ *Non-monetary valuation methods*

A critique often given on the monetary valuation methods by sociologists and psychologists is the fact that traditional economics starts from the paradigm of the rational human being, optimizing its behaviour and preferring to maximise benefits. This is not always correct. Empirical research in behavioural economics, anthropology, psychology and moral philosophy have rejected the standard economic assumptions with respect to people‘s preferences and behaviours. Consequently, expressing values in monetary terms is not always suited to express preferences. Several non-monetary valuation methods are considered an alternative approach.

Deliberative and inclusionary approaches (DIPs) including participatory appraisal, focus groups, the Delphi approach, consensus conferences and citizen's juries, can help to overcome the critique. These methods are aimed at creating better informed decisions that are owned by, and have the broad consent of, all relevant actors and stakeholders. They stand in contrast to the more 'technocratic' approaches such as Cost-Benefit Analysis, Cost Effectiveness Analysis, or Multi-Criteria Analysis. DIPs seek to build a process of defining and redefining interests that stakeholders introduce as the collective experience of participation evolves. As participants become more empowered (i.e., more respected and more self-confident) it is assumed they may become more able to adjust to, listen to, and learn from others, and accommodate to a greater consensus.

'Focus groups, in-depth groups aim to discover the positions of participants regarding, and/or explore how participants interact when discussing, a pre-defined issue or set of related issues. In-depth groups are similar in some respects, but they may meet on several occasions, and are much less closely facilitated, with the greater emphasis being on how the group creates discourse on the topic.

Citizens' juries are designed to obtain carefully considered public opinion on a particular issue or set of social choices. A sample of citizens is given the opportunity to consider evidence from experts and other stakeholders and they then hold group discussion on the issue at hand.

Disadvantage of the above non-monetary methods is that you can only perform these methods in small groups of stakeholders, in comparison with e.g. CV and choice modelling. A correct selection of stakeholders is essential.

The intention of **Delphi surveys and systematic reviews** is to produce summaries of expert opinion or scientific evidence relating to particular questions. However, they both represent very different ways of achieving this. Delphi relies largely on expert opinion, while systematic reviews attempt to maximise reliance on objective data. Delphi and systematic review are not methods of valuation but rather means of summarising knowledge (which may be an important stage of other valuation methods). Note that these approaches can be applied to valuation directly, that is as a survey or review conducted to ascertain what is known about values for a given type of good.' (Hadley et al. 2011)

Health-based valuation approaches measure health-related outcomes in terms of the combined impact on the length and quality of life. For example, a quality-adjusted life year combines two key dimensions of health outcomes: the degree of improvement/deterioration in health and the time interval over which this occurs, including any increase/decrease in the duration of life itself.

→ *Conclusion*

All methodologies have their strengths as well as their shortcomings. They are affected by uncertainty, stemming from incomplete knowledge of ecosystem dynamics, human preferences and technical issues in the valuation process. The choice of the valuation method(s) 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.

When deciding which valuation tools to use, one should consider its shortcomings. A combination of valuation techniques is required to comprehensibly value ecosystem goods and services. 'Improved understanding of the application of both economic valuation approaches and deliberative or participatory methods to valuing ecosystem services will be important. The latter obviously have a part to play in understanding people's preferences and the process of decision-making and may therefore influence policy choices. However they do not easily fit into the more formal process of economic appraisal' (DEFRA 2007).

3 Ecosystem services in estuaries

3.1 The importance of estuaries

Estuaries - as a transitional zone between land-based ecosystems and the world ocean - are vital to the biosphere's functioning. This is expressed by their complex geology, hydrology and morphology, their prominent role in the historical and actual support of economies and ecosystems, their manifest dynamics that both sustain and put at risk all inhabiting organisms and their prominent role as biogeochemical filter for land-ocean exchanges.

Estuaries and coastal marine ecosystems are cited among the most productive biomes of the world, and serve important life-support systems also for human beings (Day et al. 1989, Costanza et al. 1997). Estuaries support many important ecosystem functions: biogeochemical cycling and movement of nutrients, purification of water, mitigation of floods, maintenance of biodiversity, biological production (nursery grounds for commercial fish and crustacean species) etc. (Daily et al. 1997, De Groot 2002, de Deckere and Meire 2000, Meire et al. 1998).

Many estuaries, as is the case with the four TIDE estuaries Schelde, Humber, Weser and Elbe, are of tremendous economic and social importance as they are the main trade hub for international shipping, attracting industrial production and transport companies, providing labour and significant added economic value. Human activities have led to polluted water and land conversion. Consequently, estuarine and coastal ecosystems are some of the most heavily used and threatened natural systems globally (Lotze et al. 2006, Worm et al. 2006, Halpern et al. 2008, Barbier et al. 2011), and their deterioration due to human activities is intense and increasing (Barbier et al. 2011). This degradation has a direct impact on the services delivered by estuaries, and thus threatens the well-being of people as well as the economic activities itself.

Given the rate and scale at which estuaries and coastal ecosystems are disappearing worldwide, assessing and valuing the ecological services of these systems is critically important for improving their management and for designing better policies (Barbier et al. 2011). Yet, as the review by Barbier et al. (2011) has shown, many of these values are non-marketed, and efficient management of such ecosystem services requires explicit methods to measure this social value. Translating this value into economic incentives, management plans, project evaluations and legislation will safeguard the many benefits from estuarine ecosystem services in the long run.

3.2 Ecosystem services in estuaries

As the previous paragraph indicates, estuaries are able to deliver multiple ecosystem services simultaneously. To identify which services are relevant and how we can value these services, we start from the internationally accepted TEEB classification of ecosystem services. TEEB (The Economics of Ecosystems and Biodiversity) is a European study to evaluate the costs of the loss of biodiversity and the associated decline in ecosystem services worldwide (Balmford et al. 2008). This manual focusses on the 20 services that were considered most relevant in the TIDE-estuaries, plus the provision and use of sand (ranked 24th in TIDE) because of its assumed economic importance (Jacobs et al. 2013)

Table 2: Ecosystem services in estuaries considered in this manual

TEEB classification	TIDE	Short description
PROVISIONING		
Food	1. Animals / Crops	presence and use of edible animals, including livestock growth and fodder production
Water	2. Water for industrial use	provision and use of water for e.g. cooling water, rinsing water, water for chemical reactions
	3. Water for navigation	presence and use of water for shipping purposes
Raw materials	4. Sand	Provision and use of sand from dynamic environments which are renewed within a few generations (100 y)
Genetic resources		
Medicinal resources		
Ornamental resources		
REGULATING		
Air quality regulation		
Climate regulation	5. Carbon sequestration and burial	buffering carbon stock in living vegetation, burial of organic matter in soils
Disturbance prevention or moderation	6. Flood water storage	storage of storm or extreme spring tides in natural or flood control habitats
	7. Water current reduction	reduction of water current by physical features or vegetation
	8. Wave reduction	reduction of wave height by physical features or vegetation
Regulation of water flows	9. Water quantity: drainage of river water	drainage of the catchment by the river
	10. Water quantity: dissipation of tidal and river energy	buffering of average flood and discharge variations in the river bed
	11. Water quantity: landscape maintenance	formation and maintenance of typical landscapes and hydrology

TEEB classification	TIDE	Short description
	12. Water quantity: transportation	discharge and tidal input for shipping, including water use for canals and docks
Waste treatment	13. Water quality: transport of pollutants and excess nutrients	transport of pollutants from source, dilution
	14. Water quality: reduction of excess loads coming from the catchment	binding of N, P in sediments and pelagic food web
Erosion prevention	15. Erosion and sedimentation prevention by water bodies	sediment trapping and gully erosion by variable water currents and topography
	16. Erosion and sedimentation prevention by biological mediation	sediment trapping and erosion prevention by vegetation, effects of bioturbation
Maintaining soil fertility		
Pollination		
Biological control		
HABITAT & Supporting**		
Lifecycle maintenance	17. Biodiversity	total amount of abiotic and biotic diversity at all levels (gene-landscape), regardless of rarity or vulnerability
Gene pool protection		
CULTURAL & Amenity ***		
	18. Aesthetic information	appreciation of beauty of organisms, landscapes,…
	19. Recreation & tourism	opportunities and exploitation for recreation & tourism
	20. Inspiration for culture, art and design	appreciation of organisms, landscapes,… as inspiration for culture, art and design
	21. Information for cognitive development	use of organisms, landscapes for (self-) educational purposes

*: Based; TEEB; Gantioler et al. 2010
**: These are the insurance/condition for all ES. Biodiversity in the broad sense.
***: Classification follows UK National Ecosystem Assessment, Church 2011

3.3 Approach

When assessing the impact on ecosystem services it is essential to assess the whole bundle of relevant services. However, there is not the same amount of information available for all ecosystem services. State-of-the-art data and insights were gathered and used to develop the best possible valuation methodology. The total value of ecosystem services can be represented by a combination of monetary values, quantitative numbers and qualitative insights (and unknowns), with generally less information and insight being available at the monetary level, and a broader view at the qualitative level. This is also referred to as the pyramid-approach as described in Gantioler et al. (2010).

In this guidance you will find for each ecosystem service the minimal elements to take into account, the assumptions made and where the needed information can be found or how they can be collected (monitoring, literature...).

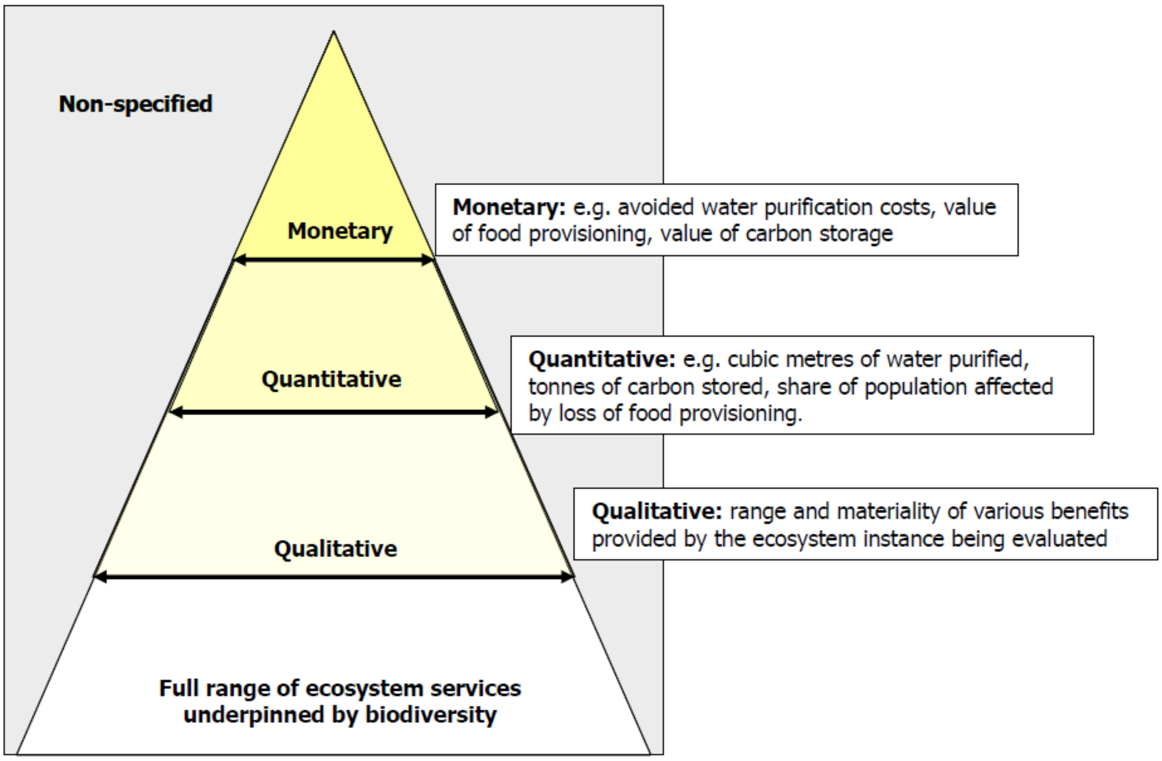

Figure 4: Pyramid-approach in ecosystem services assessment

Identification means giving a qualitative score of the significance of a certain ecosystem service in estuaries, and a qualitative description of the way different restoration measures have an influence on the service (factors influencing the functioning of the ecosystem service e.g. flooding frequency, nutrient content of the water, salinity...). This is based on expert judgement. Data on relevance and importance of certain ecosystems for a specific ecosystem service are collected in a TIDE survey (Jacobs et al. 2013). That report also explains the scores in detail.

Quantification means giving a magnitude in bio-physical terms of the flow of services e.g. number of visitors, kg nutrients, tonnes biomass....

Valuation means expressing the value of the quantified ecosystem services. This can be done in monetary as well as non-monetary terms. For this guidance we focus on monetary valuation methodologies.

4 Use of this guidance document

4.1 Who is the guidance intended for?

The guidance document can help water managers and other people working in an estuarine environment (local authorities, city regions, local enterprise partnerships, port authorities and non-governmental organisations) to estimate the ecosystem services delivered in estuaries and how this can be influenced for instance by infrastructure works or restoration projects.

4.2 Why was it developed?

"Biodiversity policy is not a new field. In recent decades, nearly all countries have adopted targets and rules to conserve species and habitats. Despite this progress, the scale of the biodiversity crisis shows that current policies are simply not enough" (TEEB 2011).

A root cause of this biodiversity crisis is the neglect of the benefits that biodiversity and ecosystems deliver because:

- Their benefits take many forms and are widespread.
- Existing markets and market prices only capture a minor part of these benefits.
- The cost of conservation and restoration has to be paid immediately often at local level, whereas many benefits occur in the future and occur at a different spatial level.

There is a need to make the value of ecosystem services more clear for communication but also to take them into account in decision support tools.

Since economic values are very context dependent (both in time and space), preferably for each decision-making situation original data on ecosystem services and their value should be collected. This is however very time and budget consuming. Often, the only realistic way to estimate the full economic consequences of planned changes in ecosystems is to use proxy data from areas that are ecologically comparable and have a similar socio-economic context through so-called benefit transfer techniques.

This guidance document provides a set of indicators to help assess the impact on ecosystem services delivered by estuaries and to translate this information into policy applications and decision support tools such as a cost-benefit analysis.

4.3 Using the guidance

A number of steps need to be taken when you use this guidance document. These steps are summarized in the table below.

Table 3: Steps in the valuation process

Preparation	Step 1: Identification of the project	User guide 4.3.1
	Step 2: Identification of current land use project area	User guide 4.3.2
	Step 3: Identification of future land use of project area and changes within same land use	User guide 4.3.3
	Step 4: Selection of relevant ecosystem services	User guide 4.3.4 + Table 6
Valuation	Step 5: Gather input data needed for the calculation of relevant ecosystem services	User guide 4.3.5 + x.x.1 of every relevant ecosystem service
	Step 6: Identification: Provides qualitative assessment of effects (scores)	User guide 4.3.6 + x.x.2 of every relevant ecosystem service
	Step 7: Quantification: Provides quantitative assessment of effects (e.g. hectares of habitat, tonnes of carbon).	User guide 4.3.6 + x.x.3 of every relevant ecosystem service
	Step 8: Monetary valuation: Estimate annual environmental cost or benefit in €/year	User guide 4.3.6 + x.x.4 of every relevant ecosystem service
Policy application and reporting	Step 9: Apply results as part of an environmental impact assessment or cost-benefit analysis	User guide 4.3.7.
	Step 10: Make the assessment of economic value available to the wider decision-making process.	User guide 4.3.8

Phase 1: Preparation

4.3.1 Step 1: Identification of the project

Ask yourself the following questions:

My project has:
- a direct positive or negative effect on estuarine ecosystems? Examples include the destruction, fragmentation or creation/restoration of wetlands.
- an indirect (positive or negative) effect on estuarine ecosystems? Example effects include disturbance, drainage and impact on the aesthetic value.

If the answer is yes or unsure to one of these questions, it makes sense to estimate the impact on ecosystem services and to proceed to step 2.

4.3.2 Step 2: Identification of current land use (ecosystems) of the project area

If the project has an effect on the estuarine ecosystem, find information on the different types of land use situated in the study area in hectares for each land use class.

Habitat categories were derived from physical maps of elevation and tidal prisms of the estuaries addressed in the TIDE project. Six habitat types were distinguished and described (see report “Comparison of Hydrodynamics and Salinity of TIDE Estuaries”). Salinity zones were defined in four zones: freshwater zone, oligohaline, mesohaline and polyhaline zone based on the Venice approach as discussed in TIDE (Geerts et al. 2011). The combination of tidal prism and salinity zones leads to a total of 24 estuarine habitats.

Table 4: Estuarine habitats and tidal prisms

Habitat	**Tidal prism**
Marshes	above mean high water (MHW)
Intertidal steep habitat	Between MHW and MLW, slope > 2.5%
Intertidal flat habitat	between MHW and MLW, slope < 2.5%
Subtidal shallow	between MLW and 2m beneath MLW
Subtidal moderately deep	between 2m and 5m beneath MLW

Subtidal deep	>5m beneath MLW

Table 5: Salinity classes

	Chlorinity			**Salinity**		
Fresh water zone		< 300	mg/l		< 0,5	PSU
Oligohaline zone	300	3000	mg/l	0,5	5	PSU
Mesohaline zone	3000	11000	mg/l	5	18	PSU
Polyhaline zone	11000	18500	mg/l	18	30	PSU

In the guidance document we also refer to agricultural land and built-up area next to the estuarine land classes defined in TIDE. This is relevant for case studies as these are mostly subject to replacing estuarine habitats with agricultural or built-up area and vice versa. If an estuarine habitat is restored it may be possible that other nature types (e.g. forest and grassland) will be replaced by the estuarine nature. This potentially causes the loss of some ecosystem services. We refer to other manuals on ecosystem services of terrestrial habitats to take these changes into account. For Flanders, Belgium the “Nature Value explorer” is a webtool to explore the value of ecosystem services (www.natuurwaardeverkenner.be).

4.3.3 Step 3: Identification of future land use of the project area

Identify how the land use, or the area of the different estuarine habitats identified, will change after the project in step 3.

It is also possible that the project does not include a changing land use as such but influences some underlying parameters such as groundwater levels. In this case identify the chemical and biological processes that might be influenced.

4.3.4 Step 4: Selection of relevant ecosystem services

In step 4 we select the relevant ecosystem services which might be affected by the project at hand.

Ask yourself the following questions:

- Which ecosystem services are delivered at present?
- Which ecosystem services are potentially influenced (positive or negative) by the project?

To find an answer on these questions, we advice to discuss with experts which ecosystem services will be affected and are relevant to include. The first question can also be answered by using check lists. Table 6 is an example check list and gives a first indication on the ecosystem services which are relevant for each estuarine land use class. In this overview we do not include supporting services as their impact is also reflected in other services.

To answer question 2, the calculation of the qualitative value in step 6 can be used as a scoping tool to see which ecosystem services need further consideration.

Table 6: Relevant ecosystem services per land use class and salinity zone.

S=supporting service
R= regulating service
P=provisioning service

Legend:

Score	Habitat has…in supply of ES
1	no importance
2	very low importance
3	moderate importance
4	Importance
5	Essential importance

		"Biodiversity"	Erosion and sedimentation regulation by water bodies	Water quality regulation: reduction of excess loads coming from the catchment	Water quality regulation: transport of polutants and excess nutriënts	Water quantity regulation: drainage of river water	Erosion and sedimentation regulation by biological mediation	Water quantity regulation: transportation	Water quantity regulation: landscape maintenance	Climate regulation: Carbon sequestration and burial	Water quantity regulation: dissipation of tidal and river energy	Regulation extreme events or disturbance: Wave reduction	Regulation extreme events or disturbance: Water current reduction	Regulation extreme events or disturbance: Flood water storage	Water for industrial use	Water for navigation	Food: Animals	Materials: sand	Aesthetic information	Inspiration for culture, art and design	Information for cognitive development	Opportunities for recreation & tourism
		S	R1	R2	R3	R4	R5	R6	R7	R8	R9	R10	R11	R12	P1	P2	P3	P4	C1	C2	C3	C4
Freshwater	Marsh	X	X	X	X	X	X	X				X		X			X		X	X	X	X
	Intertidal flat	X	X	X	X	X	X	X		X	X	X		X					X	X	X	X
	Intertidal steep	X				X	X												X	X	X	
	Subtidal shallow	X	X	X	X			X	X	X		X		X					X	X	X	X
	Subtidal moderately deep	X							X	X			X	X	X	X			X	X	X	X
	Subtidal deep	x							X				X	X	X	X			X	X	X	X

		S	R1	R2	R3	R4	R5	R6	R7	R8	R9	R10	R11	R12	P1	P2	P3	P4	C1	C2	C3	C4
	Adjacent land																					
Oligohaline	Marsh	X	X	X	X	X	X	X		X		X		X			X		X	X	X	X
	Intertidal flat	X	X	X	X	X	X	X		X		X		X					X	X	X	X
	Intertidal steep	X				X	X							X					X	X	X	X
	Subtidal shallow	X	X	X	X			X	X	X	X	X		X			X		X	X	X	X
	Subtidal moderately deep	X							X	X	X	X	X	X	X	X	X		X	X	X	X
	Subtidal deep	X							X		X		X	X	X	X			X	X	X	X
	Adjacent land																					
Mesohaline	Marsh	X	X	X	X	X	X	X		X		X		X					X	X	X	X
	Intertidal flat	X	X	X	X		X	X		X		X		X					X	X	X	X
	Intertidal steep	X					X												X	X	X	X
	Subtidal shallow	X	X	X	X			X	X	X	X	X		X			X		X	X	X	X
	Subtidal moderately deep	X							X	X	X	X	X	X	X	X	X		X	X	X	X
	Subtidal deep	X							X		X		X	X	X	X			X	X	X	X
	Adjacent land																					
Polyhaline	Marsh	X	X	X	X	X	X	X	X	X		X		X					X	X	X	X
	Intertidal flat	X	X	X	X		X	X		X		X		X					X	X	X	X
	Intertidal steep	X					X												X	X	X	X
	Subtidal shallow	X	X	X	X			X	X	X	X	X		X			X		X	X	X	X
	Subtidal moderately deep	X							X	X	X	X	X	X			X		X	X	X	X
	Subtidal deep	X							X		X		X	X	X	X			X	X	X	X
	Adjacent land																					

Legend:

Score	Habitat has…in supply of ES
1	no importance
2	very low importance
3	moderate importance
4	Importance
5	Essential importance

Phase 2: Calculation

For steps 5 to 8, each ecosystem service is dealt with in a separate subchapter either in chapter 5 (provisioning services), chapter 6 (regulating services), chapter 7(biodiversity) or chapter 8 (cultural services) of this guidance document.

4.3.5 Step 5: Gather information needed for the calculation of ecosystem services

For each ecosystem service we provide which information the user needs to collect for the quantification and valuation of the respective ecosystem service. This information reflects the underlying bio-physical factors that influence the service delivery.

While collecting this information for the present state, the user should also assess how a project influences these parameters e.g. lower groundwater level, accessibility for recreation.

4.3.6 Step 6-7-8: Calculation of qualitative, quantitative and monetary value

We describe for each ecosystem service specific methodologies to perform the calculation of qualitative, quantitative and monetary values.

The qualitative assessment or identification is based on expert judgement. Data on relevance and importance of certain ecosystems for specific ecosystem services are collected in a TIDE survey (Jacobs et al. 2012). Experts were questionned on the importance of a specific estuarine habitat for the supply of a specific ES. Jacobs et al. analyse these scores in their report.

Score	Habitat has…in supply of ES
1	no importance
2	very low importance
3	moderate importance
4	Importance
5	Essential importance

This score (when two scenarios are compared) gives the importance of the change in the delivery of each ecosystem service due to the project. It can be used as a scoping tool. When the change between the base and future scenario is more than 2 points, it is worthwhile going through the next steps of quantification and valuation. When the change is 4 points it is definitely worthwhile looking in more detail into the value of this service using more sophisticated ecological and economic models instead of using benefit transfer. This is definitely true when the effect on the ecosystem service evokes discussion amongst stakeholders.

The quantitative valuation (also referred to as quantification) and monetary valuation (also referred to as valuation) are based on a literature review for different ecosystem services in estuarine habitats. In this guidance we explain shortly the processes leading to the supply of the ecosystem service. More details are found in the separate TIDE-reports on specific ecosystem services or in the attachments of this guidance document.

We indicate where you may find the correct information to quantify (x.x.3 of each chapter) and monetize (x.x.4 of each chapter) the change in ecosystem services.

As it is not always easy to find the necessary input data on biophysical parameters, we also give values per hectare frequently applied in literature. These values are based on meta-analysis of different valuation studies on estuaries in the world. These values hardly take into account local characteristics and should be used with care. As earlier said, if a change in an ecosystem service appears to be very important it is better to use more spatially specific information.

Phase 3: Policy application and reporting

4.3.7 Step 9: Apply the results in a cost benefit analysis

Cost-benefit analysis (CBA) is an applied economic tool often used to guide economic agents in resource allocation or investment decisions. It is a technique that is used to sum up (in present value terms) and compare the future flows of benefits and costs of different alternatives to establish the worthiness of undertaking the stipulated activity or alternative, and inform the decision maker about economic efficiency. (Balana et al. 2011)

Including the impact on different ecosystem services is particularly useful to assess the impact of so called multi-purpose projects having an impact on different environmental and other issues simultaneously. By quantifying and valuing the different services these projects deliver, a better view can be obtained on their total impact instead of focusing on a single environmental issue.

The calculations in this guidance document are yearly benefits (price level 2010). More information on how to go from yearly benefits to a cost-benefit analysis can be found in a wide range of manuals (Brent, 2006; Mishan and Quah, 2007; Boardman, 2006; Eijgenraam, 2000;

European Commission, 2008; MOW, 2013.) Usually the net present value is calculated for a pre-defined time horizon (depending on the project lifespan) using a specific discounting procedure. Different views exist on what an appropriate discount rate for nature restoration or nature loss should be. We advise a discount rate between 2.5% and 5% with 4% as the central value to take into account future costs and benefits in the analysis. Specific sources in the ecosystem services literature argue that it should be lower (see e.g. TEEB 2010).

4.3.8 Step 10: Reporting

The methodologies described in this guidance document allow you to perform a rough estimation of the benefits of estuaries. Quantifying the different effects in detail depends on site-specific circumstances and requires tailor-made research and calculations. It is therefore important to report the constraints of the valuation exercise. Attention should be paid to: (i) uncertainty concerning estimates of environmental effects (e.g. timing, magnitude and significance); (ii) assumptions embodied in estimates of the relevant number of households, visitors...; (iii) assumptions entailed in the transfer of economic values or functions; (iv) the potential significance of any incomplete information or non-monetised impacts, and (v) caveats associated with the resulting value estimates.

5 Valuation methodologies for provisioning services

5.1 Food: agricultural animals and crops

5.1.1 Information needed

For quantification and valuation we need:
- Standard gross margin per crop (added value per ha excl. subsidies)
- Amount of ha per crop

5.1.2 Identification

Food provision includes the production of crops such as grains, vegetables and fruits or agricultural products for animal consumption.

At first sight food provisioning is not evident to consider as ecosystem service. However, including provisioning services derived from agriculture or agro-ecosystems is essential in a tradeoff analysis e.g. in restoration projects a trade-off needs to be made between keeping the existing agricultural land use or restore estuarine nature. Furthermore, agricultural systems comply in a strict sense with the definition of an ecosystem (Maes et al. 2011).

The production depends on management practices (only production goals, environmental or nature goals), soil characteristics and erosion sensitivity. There is no freely available framework to improve the scoring towards these influencing parameters.

Therefore, we use a very simple qualitative score system where agricultural land is 5 (very important) and non-agricultural land is 1 (not important).

5.1.3 Quantification and monetary valuation

For the valuation of this ecosystem service we suggest to estimate the market prices for animals and crops grown in estuarine ecosystems. It is to be noted that, in general, the current benefits (monetary benefits in particular) obtained from biodiversity resources do not often reflect sustainable extraction or production patterns. The external costs related to this issue are not taken into account.

The estimated value of the biodiversity resource based on market price is equal to the quantity of resources sold multiplied by the standard gross margin (market price – variable costs related to production). This is corrected for taxes and subsidies.

This can be found per country at Eurostat: http://epp.eurostat.ec.europa.eu/portal/page/portal/agriculture/data/database. Unfortunately, this database was not updated since 2004.

For fodder production it is usually difficult to find information on the standard gross margin as the major amount is not sold on markets. An alternative method is to link the production of fodder to the meat and dairy standard gross margins. Calculate the average number of cows and the amount of dairy produced per ha fodder and multiply this with the standard gross margins of cattle and dairy.

5.1.4 Illustration

The table below gives an illustration of standard gross margins estimated for Flanders, Belgium.

Table 7: Average standard gross margins 2008-2010 for Flanders

	Average standard gross margins 2008-2010 excl. subsidies (€/ha.jaar)		
Crops (major classes)	**P25**	**P50**	**P75**
Maize	1.003	1.300	1.526
Cereals, seeds and pulse	718	963	1.233
Grassland	1.245	1.580	1.818
Fodder	1.245	1.580	1.818
Flax and hemp	788	1.159	1.414
Vegetables, spices and ornamental plants	1.714	2.733	4.048
Potatoes	1.727	2.767	4.259
Sugar beet	1.263	1.588	1.905
Fruits and Nuts	5.257	7.601	10.718
Other	1.901	2.507	2.916
Infrastructure	Included in built-up area		
Wood	Included in wood production		

An area of 150 hectares is used to grow 100 ha grassland and 50 ha maize. It is the intention to restore a natural flooding regime in this area which causes the loss of the agricultural production in the current situation.

Identification

The present score is 5 (100% agriculture). The future score is 1 (no agriculture).

Quantative and monetary valuation

The loss of the agricultural production is between 174.650 €/year (100ha * 1.245€/ha + 50ha * 1.003€/ha) and 258.100 €/year (100ha * 1.818€/ha + 50ha * 1.526€/ha).

5.2 Food: other (fish, non-cultivated plants...)

5.2.1 Information needed

- Quantity produced: kg of fish, other products extracted from the estuary itself and from the sea.
- For fish and shelfish extracted from the sea: information or model that attributes the production of juveniles to the adult stock (probably not available).
- Market prices for food products.

5.2.2 Identification

Direct fishing and shellfish breeding inside estuaries occurred more commonly in history. Also other products such as statice are only extracted on a small scale. However, estuaries are still regarded highly important as foraging, breeding or spawning ground for commercial fish species which spend part of their live cycle in fresh or brackish water. This is reflected in the higher scores of shallow and moderately deep subtidal areas.

Habitat	Qualitative importance per estuarine zone*			
	Freshwater	Oligohaline	Mesohaline	Polyhaline
Marsh habitat	3	3	2	2
Intertidal flat habitat	2	2	2	2
Intertidal steep habitat	2	2	2	2
Subtidal shallow habitat	2	3	3	3
Subtidal moderately deep habitat	2	3	3	3
Subtidal deep habitat	2	2	2	2
Agricultural land	1	1	1	1
Built-up area	1	1	1	1
** Score from 1 (not important) to 5 (very important)*				

5.2.3 Quantification

The amount of catch per species, produced salt plants etc. is published on a regular basis. The difficulty however lies in atributing the amount of fish caught to specific estuaries. Assumptions need to be made about the number of juveniles originating from an estuarine habitat being caught as adults.

5.2.4 Valuation

Market prices can be used. The market prices that e.g. the fishermen get for their catch should be subtracted with the variable costs they have in order to be able to retreive the fish.

5.2.5 Illustration

At the estuary of the Ijzer, Belgium, a management plan created a natural flooding area (marsh habitat) on 27 ha agricultural land. This created a positive effect on the nursery function for the shrimp catch in the North-Sea. To link the benefits of the shrimp catch to the particular project several assumptions were made about the production of juveniles and the recruitment to the adult stock. A population model was developed dividing the shrimps in 20 classes of different lengths. Growth and mortality were incorporated. Based on the assumptions made it was estimated that between 0.5 and 0.9 tonnes of adult shrimps are recruited to the shrimp fishery (source: Liekens et al. 2006, in Dutch).

This recruitment is valued with a market price method. The total value for this habitat service was estimated between 0.14 mio€/year and 0.46 mio€/year for the different scenarios.

5.3 Water for industrial use

5.3.1 Information needed

- Water use industrial sector abstracted from the estuary
- Average abstraction cost per m³
- Average cost tap water per m³ or estimated production losses because of water scarcity

5.3.2 Identification

Industry in the estuary can abstract water from the river to use as cooling water or processing water.

Habitat	Qualitative importance per estuarine zone*			
	Freshwater	Oligohaline	Mesohaline	Polyhaline
Marsh habitat	2	2	2	2
Intertidal flat habitat	1	2	2	2
Intertidal steep habitat	2	2	2	2
Subtidal shallow habitat	2	2	2	2
Subtidal moderately deep habitat	3	3	3	2
Subtidal deep habitat	3	4	4	4
Adjacent land				
** Score from 1 (not important) to 5 (very important)*				

5.3.3 Quantification

m³ water directly taken from the estuary for processing and cooling by industrial sector in the estuary.

5.3.4 Valuation

The value of water provisioning for industry can be estimated by the cost differences between abstracting the water directly from surface water or replacing the natural water supply by an alternative, mostly tap or groundwater.

An alternative methodology is to calculate the contribution of water to the added value of the products or the damage costs when companies are faced with water scarcity.

Water provisioning is interlinked with other services, more specifically regulation of waterflows.

5.3.5 Illustration

A Belgian study (IMDC 2006) estimated the economic consequences of water scarcity based on different methods (value of production losses, replacement costs, willingness to pay) in the Albertkanaal. The values are:

- Production losses agriculture: 0.5-18 €/m³ water needed
- Losses drinking water: 1-150 €/m³ water depending on % and number of days of scarcity
- Production losses industry: 5-200 € loss in revenues when no water availability per m³ water needed
- Production losses energy: 0.073 €/m³ water needed

The wide range in numbers reflect the very company specific damage costs when industry is faced with water scarcity.

5.4 Water for navigation

5.4.1 Information needed

- Tonne-km transported over water in the estuary
- Changes in the channel that changes the potential in navigation
- Additional costs/benefits for the transportation sector and external costs for different modes.

5.4.2 Identification

Rivers are used for transportation by ships which is a relative cheap and clean transportation mode. The value of the navigation service can be estimated by looking at the additional costs (or gains) for the transportation sector and for society (environmental costs) if the navigation possibilities decline (improve). The potential for navigation depends on the characteristics of the fairway. If greater vessels with a larger draught can enter the port or navigate the river the transportation costs per tonne-km transported may be lower. This is likely to lead to a higher amount of goods transported by ship. If the transportation by ships replaces transportation by other modes, this modal shift will result in less transportation and environmental costs (external costs), e.g. related to health impacts from air pollution or global warming.

The characteristics of the fairway may be affected by a wide range of ecosystem processes including erosion and sedimentation, energy dispersion, speed of the tides and water levels in the inland rivers. In addition, navigation possibilities will be affected by man-made actions including dredging and navigation help.

Habitat	Qualitative importance per estuarine zone*			
	Freshwater	Oligohaline	Mesohaline	Polyhaline
Marsh habitat	1	1	1	1
Intertidal flat habitat	1	2	1	1
Intertidal steep habitat	1	2	1	1
Subtidal shallow habitat	1	2	1	1
Subtidal moderately deep habitat	3	3	3	2
Subtidal deep habitat	4	5	5	5
Adjacent land				
** Score from 1 (not important) to 5 (very important)*				

5.4.3 Quantification

The indicator to quantify navigation is typically the amount of tonne-km per year transported in the estuary.

5.4.4 Valuation

The valuation of the transportation service can only be determined in comparison with an alternative situation for which it is specified how goods will be transported. Transportation costs for shipping may be affected due to changes in the estuary and the fairway. As shipping is on average a cheaper and cleaner mode, this will lead to additional transportation costs for the sector and society. This will probably require a more detailed analysis. Five different types of costs can be estimated (Kind 2004):

a) Efficiency gains (losses), e.g. due to more (less) tons/ships
b) Time gains (losses) due to faster (slower) trajectories for shipping and or time required to enter the port.
c) Additional costs (benefits) due to longer (shorter) trajectories.
d) Modal shift benefits/costs, if goods are transported by other modes, that are less/more expensive
e) Environmental benefits (costs) linked to shorter (longer) trajectories and modal shifts.
f) Costs of additional measures (e.g. dredging) to prevent the cost categories a to d.

The full analysis may require detailed studies, using local data and or specialised models. For some types of costs, more generic data are available.

a) Efficiency gains (losses)

   A temporarily lower draught will cause additional costs for transportation because (temporarily) the full loading capacity of the ships cannot be used. Less tonnes per trip per vessel will increase the costs per ton transported.

   For a detailed analysis, the use of specific models may be required, e.g. the PAWN model (PAWN= Policy Analysis for the Watermanagement of the Netherlands; Kind 2004)
   For a rough estimate, it may be assumed that the increase in costs per tonne-km equals the reduction in capacity. If only 90 % of the full capacity can be used, the costs will increase with 10 %.

b) Time gains (losses)

   Changes in the fairway may affect the speed of ships or the time required to enter the port. This will require detailed analysis. As an example, in the CBA concerning the deepening of the

Westerschelde this was estimated at 46€ per hour draft restricted[1] per TEU (Twenty feet Equivalent Unit) (Nistal 2004).

c) Benefits (costs) from shorter (longer) trajectories

This impact can be valued based on data for costs per tonne-km for shipping. If no local data are available, the data from the European COMPETE project can be used as a proxy (see below, d).

d) Benefits (costs) from modal shift

If there are no case or country specific data on transportation costs available, the data of the European COMPETE project (Maibach M. et al. 2006) can be used. It estimates the average operational costs per transport mode for the EU member states and Europe. Costs differ between member states because of different factors:

- Transport volumes
- Fleet structure and age
- Market prices and financing conditions of equipment (vehicle market, garage, maintenance, equipment, interest rates, insurance etc.): These prices are in addition dependent of the level of liberalisation of the equipment market.
- Energy consumption (depending on average energy use of the fleet)
- Structure of charges and taxes (infrastructure use, road taxes, environmental taxation)
- Taxation structure (transport taxes, others)
- Wage level (usually depending on general economic conditions according to GDP per capita
- Level of competition/liberalisation of the transport sector.

The numbers below illustrate that the costs advantage for shipping is very important (data for 2005).

Transport mode	€/tonne-km
Road	0.14
Rail	0.09
Water	0.009
Air	0.75

[1] “aanloopweerstand (Dutch): weighted average of the waiting time in hours because of ‘depth limits’ in the channel with as weights the total capacity of the ships of a certain TEU-class.

e) Environmental costs
In general, transportation over water will impose less external costs to society related to environmental impacts (greenhouse gasses, air quality, noise), congestion and road safety (esp. compared to road transport).
Specific data for external costs per tonne-km may be available. If not, one can use the data from the "handbook on the estimation of external costs in the transport sector (IMPACT project) (Maibach 2008).

f) Costs of additional measures
It is assumed that the impact of changes in the ecosystem on the fairway will be born by the transportation sector. In practise, the impact may be compensated by actions from other administrations or port authorities that may lead to higher (or lower) costs. A typical example is dredging to maintain the depth of the fairway.

In this case, the addional (or savings on) expenditure per year for dredging is a measure of the willingness to pay to preserve navigation on a specific waterway.

5.4.5 Illustration

In the cost-benefit analysis of the third deepening of the river Schelde for the harbour of Antwerp the transport benefits (mainly shorter waiting time) were estimated around 2 billion € (Nistal 2004).

5.5 Materials: sand

This service includes the extraction of sand from sand banks and dunes. This can however lead to environmental degradation. Therefore, we only include the amount of sand extracted sustainably.

5.5.1 Information needed

- Amount (m^3) of sand extracted sustainably per year
- Added value per m^3 sand

5.5.2 Identification

This service was not included in the TIDE survey. We have no information available to link estuarine habitats with sand production.

5.5.3 Quantification

The quantification if performed by estimating the amount (m^3) of sand extracted sustainably from the estuary.

5.5.4 Monetary valuation

The monetary valuation can be performed by multiplying the amount of abstracted sand with the market price for unprocessed sand (excluding processing costs). In correspondence to other provisioning services, we use the net added value, excluding operational costs.

5.5.5 Illustration

In the Westerschelde 2 million m^3 sand per year is extracted on different locations. In the Zeeschelde this is 1 million m^3, only at one location. Market prices being paid for this sand are 2€/m^3 for the Westerschelde and 0.30€/m^3 for the Zeeschelde. The total economic value of this ecosystem service amounts to 4,3 million €/year.

6 Valuation methodologies for regulating services

6.1 Carbon sequestration and burial

The ecosystem service “climate regulation” encompasses the “influence of ecosystems on local and global climate through land-cover and biologically mediated processes” (De Groot 2011). For estuaries this service covers the balance and maintenance of the chemical composition of the atmosphere and ocean, on different scales; they regulate global and regional climates by sequestering or releasing carbon dioxide and other greenhouse gases (GHGs).

6.1.1 Information needed

- Measured data on soil properties (carbon concentration, bulk density, vertical accretion rates) or total C-burial rates and greenhouse gas fluxes at the different habitats in your estuary.
- Or IF you don’t have these data: use the table with indicator data.
- Area of habitat (m^2) and salinity zone. You need this for the present and the future situation.

6.1.2 Identification

Estuarine ecosystems are biologically extremely productive (Bianchi, 2007), with net primary production rates among the highest of the world. Consequently, these systems play globally an important role as carbon sinks in terms of carbon burial (Chmura et al. 2003). Most of the studies on carbon sequestration only account for carbon burial, and not for GHG emissions such as carbon dioxide, methane and nitrous oxide emissions. However, these emissions may decrease the potential benefits of CO_2-sequestration through gross organic burial by at least 50%.

It was attempted to synthesize available knowledge on biogeochemical cycling resulting in carbon sequestration in temperate estuarine environments. The focus lays on long-term carbon burial in estuarine sediments and the emissions of greenhouse gases (GHGs) to the atmosphere, the sum of which gives total carbon sequestration.

For this study it was not possible to use the same habitat classification, since the exact details of the studied habitats are not known. Instead a more general distinction was made; tidal marsh, intertidal flats and the water column (pelagic). It was also impossible to distinguish freshwater and oligohaline habitats and polyhaline and marine; the salinity zonation used is freshwater (0-5 PSU), brackish (5-18 PSU) and salt (18 – 50 PSU).

Habitat	Qualitative importance per estuarine zone*			
	Freshwater	Oligohaline	Mesohaline	Polyhaline
Marsh habitat	4	5	4	4
Intertidal flat habitat	3	4	4	4
Intertidal steep habitat	2	2	2	2
Subtidal shallow habitat	3	3	3	3
Subtidal moderately deep habitat	2	2	2	2
Subtidal deep habitat	1	1	1	1
Adjacent land				
** Score from 1 (not important) to 5 (very important)*				

6.1.3 Quantification

Long-term carbon storage should be based on carbon removed over approximately 100 years (Crooks et al. 2010) and therefore only sequestration in sediments is taken into account. Carbon sequestration capacity is determined by long-term CO_2eq-fluxes (carbon burial and GHG emissions); when the sum of these fluxes is negative, carbon is stored on the long run.

We used a bottom-up approach to calculate carbon sequestration in estuarine habitats; the used calculation for C-sequestration is as follows: C sequestration = (C-burial) - (CO_2+CH_4+N_2Ofluxes), where the used unit for all fluxes is CO_2eq. When the value is positive, carbon is sequestered in the system. These processes do not form a trade-off: the higher input of organic matter, the higher may be the burial, but also decomposition of this organic matter and with that CO_2 and CH_4 emissions!

It is not recommended to use the average values given in tables 8 to 11 for the quantification of the different processes that determine carbon sequestration, since these processes are dependent on various factors; values found in literature for all of these parameters give high variability, cross-systematically, spatially and temporally. A large database should consequently be constructed to cover this variability and a basic requirement of such a database should be the internal consistency of the data. A literature study was performed to create such a database, however, differences in site-specific factors and in methods between the studies may have lead to inconsistencies of the data. Data from literature was pooled for different habitats for this study. However, it is debatable to simply aggregate numbers from several studies from several locations into one mean value. This means that mean values given in the table are questionable and correct determination of carbon sequestration should be site-specific and requires tailor-made research and calculations.

The data on carbon sequestration and underlying processes were collected from studies which focus on these processes in existing situations. One should keep in mind that for the calculation of carbon sequestration for a new situation a comparison should be made between the existing situation and new situation after estuarine action. Also the project might affect the estuarine and/or system functioning on other scales (local/regional/..).

C-burial can be calculated from carbon densities, which are calculated by multiplying the carbon concentration and bulk density, and vertical accretion rates which then gives soil C accumulation rates (Chmura et al. 2003). Measurements on carbon concentration and vertical accretion rates differ among studies; differences in methods may give an overestimation of carbon burial up to factor 3. The fluxes of the three gases in mol or gram/area/time unit, in mass unit/area unit/time unit were then transformed to CO_2 equivalents/m^2/yr using the ratio 1:25:298 for CO_2:CH_4:N_2O. fluxes were calculated using different parameterizations; over- and underestimations may have occurred.

Table 8: Numbers for carbon sequestration and underlying determining processes in the freshwater part of the estuary.

salinity	habitat	proces	units	mean	min	max	Refs
Fresh	Flat	C-burial	-	-	-	-	-
		CO_2-flux	-	-	-	-	-
		CH4-flux	$g/m^2/yr$	1134.9	-0.3	3241	(23, 24, 28)
			CO_2-eq	77825.768	-20.5725	222251.58	
		N_2O-flux	g/m2/yr	0.26	0.16	0.44	(27)
			CO2-eq	77.48	47.68	131.12	
		C- sequestration	**CO2-eq**	**- - -**	**C-burial <**	**GHG emissions**	
	Marsh	C-burial	g/m2/yr	174	9	930	(13, 18, 26, 31)
			CO2-eq	636.84	32.94	3403.8	
		CO_2-flux	g/m2/yr	1732	12.8	6029	(27, 30)
			CO2-eq	1732	12.8	6029	
		CH4-flux	g/m2/yr	386	16	1543	(1,11)
			CO2-eq	26469.95	1097.2	105811.23	
		N2O-flux	-	-	-	-	-
		C- sequestration	**CO2-eq**	**- - -**	**C-burial <**	**GHG emissions**	
	Pelagic	C-burial	-	-	-	-	-
		CO_2-flux	g/m2/yr	213.5	57.1	517	(1, 12, 27)
			CO2-eq	213.5	57.1	517	
		CH4-flux	g/m2/yr	36.3	1.2	46.7	(14, 23)
			CO2-eq	2489.2725	82.29	3202.4525	
		N_2O-flux	g/m2/yr	x	0	694	(7, 19, 21, 34, 35, 36, 38)
			CO2-eq	x	0	206812	
		C- sequestration	**CO2-eq**	**-**	**C-burial <**	**GHG emissions**	

The freshwater part of the estuary comprises both the freshwater and the oligohaline zone according to the Venice classification.

References: 1) Abril & Borges 1999; 7) Borges & Frankignoulle 1999; 11) Bridgham et al. 2006; 12) Cai et al. 1999; 13) Callaway et al. 2012; 14) Chanton et al. 1989; 18) Craft 2007; 19) Ferron et al. 2007; 21) Garnier et al. 2006; 23) Kelley et al. 1995; 24) Lipschultz 1981; 26) Megonigal & Neubauer 2009; 27) Middelburg 1995; 28) Middelburg 1996; 30) Neubauer & Anderson 2003; 31) Neubauer 2008; 34) Seitzinger 1988; 38) Veeck 2007.

Table 9: Numbers for carbon sequestration and underlying determining processes in the brackish part of the estuary.

salinity	habitat	proces	units	mean	min	max	refs
Brackish	Flat	C-burial		-	-		-
		CO2-flux		-	-		-
		CH4-flux	g/m2/yr	35.7	0.64	133	(28)
			CO2-eq	2448.1275	43.888	9120.475	
		N2O-flux	g/m2/yr	0.65	-0.02	3.5	(27, 33)
			CO2-eq	193.7	-5.96	1043	
		C- sequestration	**CO2-eq**	**-**	**C-burial <**	**GHG emissions**	
	Marsh	C-burial	g/m2/yr	203	70	640	(13, 18)
			CO2-eq	742.98	256.2	2342.4	
		CO2-flux	g/m2/yr	503	70.4	849.4	(27)
			CO2-eq	503	70.4	849.4	
		CH4-flux	g/m2/yr	164.4	4.5	359	(5, 37)
			CO2-eq	11273.73	308.5875	24618.425	
		N2O-flux		-	-		-
		C- sequestration	**CO2-eq**	**- -**	**C-burial <**	**GHG emissions**	

	Pelagic	C-burial		-	-		-
		CO2-flux	g/m2/yr	98.9	0	278	(1, 12, 27)
			CO2-eq	98.9	0	278	
		CH4-flux		-	-		-
		N2O-flux	g/m2/yr	x	0	694	(7, 19, 21, 34, 35, 36, 38)
			CO2-eq	x	0	206812	
		C- sequestration	**CO2-eq**	**- -**	**C-burial <**	**GHG emissions**	

The brackish part of the estuary corresponds to the mesohaline zone according to the Venice classification.

References: 1) Abril & Borges 2004; 5) Bartlett et al. 1993; 7) Borges & Frankignoulle 1999; 12) Cai et al. 1999; 13) Callaway et al. 2012; ; 18) Craft 2007; 19) Ferron et al. 2007; 21) Garnier et al. 2006; 27) Middelburg 1995; 28) Middelburg 1996; 33) Robinson et al. 1998; 34) Seitzinger 1988; 35) Seitzinger et al. 1984; 36) Texeira et al. 2010; 37) Vandernat & middelburg 2000; 38) Veeck 2007.

Table 10: Numbers for carbon sequestration and underlying determining processes in the salt part of the estuary.

salinity	habitat	proces	units	mean	min	max	refs
Salt	Flat	C-burial	g/m2/yr	93.7	x	x	(2)
			CO2-eq	342.9			
		CO2-flux		-	-		-
		CH4-flux	g/m2/yr	0.6	-1	1.03	(2, 28)
			CO2-eq	41.145	-68.575	70.63225	
		N2O-flux	g/m2/yr	0.61	-0.03	2.8	(2, 27, 33)
			CO2-eq	181.78	-8.94	834.4	
		C- sequestration	**CO2-eq**	**+ or 0**	**C-burial >**	**/= GHG emissions**	
	Marsh	C-burial	g/m2/yr	172	0	928	(2, 13, 16, 18)
			CO2-eq	629.52	0	3396.48	
		CO2-flux	g/m2/yr	930	787	1201	(25, 27, 29)
			CO2-eq	930	787	1201	
		CH4-flux	g/m2/yr	11.3	-4.8	89.8	(2, 3, 4, 5, 11, 17, 25)
			CO2-eq	774.8975	-329.16	6158.035	
		N2O-flux	g/m2/yr	0.03	-0.39	0.3	(2)
			CO2-eq	8.94	-116.22	89.4	
		C- sequestration	**CO2-eq**	**-**	**C-burial <**	**GHG emissions**	
	Pelagic	C-burial		-	-		-
		CO2-flux	g/m2/yr	37.5	0	189	(1, 12)
			CO2-eq	37.5	0	189	
		CH4-flux		-	-		-
		N2O-flux	g/m2/yr	x	0	694	(7, 19, 21, 34, 35, 36, 38)
			CO2-eq	x	0	206812	
		C- sequestration	**CO2-eq**	**??**			

The salt part of the estuary corresponds to the mesohaline zone according to the Venice classification. References: 1) Abril & Borges 2004; 2) Adams et al. 2012; 3) Atkinson & Hall 1976; 4) Bartlett et al. 1987; 5) Bartlett et al. 1993; 7) Borges & Frankignoulle 1999; 11) Bridgham et al. 2006; 12) Cai et al. 1999; 13) Callaway et al. 2012; 16) Chmura et al. 2003; 17) Chmura et al. 2011; 19) Ferron et al. 2007; 21) Garnier et al. 2006; 25) Magenheimer et al. 1996; 27) Middelburg 1995; 28) Middelburg 1996; 29) Morris & Whiting 1986; 33) Robinson et al. 1998; 34) Seitzinger 1988; 35) Seitzinger et al. 1984; 36) Texeira et al. 2010; 38) Veeck 2007.

Table 11: Numbers for greenhouse gas emissions measured in whole estuaries

salinity	habitat	proces	units	mean	min	Max	refs
Estuary	Pelagic	CO2-flux	g/m2/yr	x	-32.4	828.5	(1, 6, 7, 8, 9, 10, 12, 15, 19, 20, 22, 27, 32)
			CO2-eq	x	-32.4	828.5	
		CH4-flux	g/m2/yr	0.24	0.07	0.41	(39, 40, 41)
			CO2-eq	16.458	4.80025	28.11575	
		N2O-flux	g/m2/yr	x	0	694	(7, 19, 21, 34, 35, 36, 38)
			CO2-eq	x	0	206812	

References: 1) Abril & Borges 2004; 6) Borges & Abril 2012; 7) Borges & Frankignoulle 1999; 8) Borges et al. 2004; 9) Borges et al. 2005; 10) Borges et al. 2006; 12) Cai et al. 1999; 15) Chen & Borges 2009; 19) Ferron et al. 2007; 20) Frankignoulle et al. 1998; 21) Garnier et al. 2006; 22) Gazeau et al. 2005; 27) Middelburg 1995; 32) Ortega et al. 2005; 34) Seitzinger 1988; 35) Seitzinger et al. 1984; 36) Texeira et al. 2010; 38) Veeck 2007; 39) Middelburg; 40) De Angelis & Scranton; 41: Abril & Iversen 2002.

6.1.4 Valuation

Sequestering carbon stock in living vegetation and burial of organic matter in soils potentially reduces the amount of GHG in the atmosphere and the climate change effect. The range of available estimates to value carbon is very broad. These values often refer to the so-called Social Cost of Carbon (SCC), the value of climate change impacts over the next 100 years (or longer) of one additional tonne of carbon emitted to the atmosphere today, i.e. the marginal global damage costs of carbon emissions. Results of a recent literature review on avoided costs and avoided damage costs for carbon are shown in Table 12. Numbers are relevant globally and can easily be transferred to other estuaries. The benefits of carbon sequestration will rise in the future because the damage by climate change will increase in the future due to growing populations, infrastructure,...The values for the years in between can be estimated by linear interpolation.

Table 12: Monetary value indicators for external costs of climate change in period 2010-2050.

Ref year (1)	euro/ton CO_2-eq.	euro/ton C (2)
2010	20	73
2020	60	220
2030	100	366
2040	160	586
2050	220	805

Ref year = year of emission or sequestration (2) 1 ton C = 3.66 ton $C0_2$ source: De Nocker et al, 2010

In between years need to be lineary extrapolated.

6.1.5 Illustration

The illustration is based on the study of Adams et al. (2012). This study observed the impact on GHG for a natural saltmarsh (NSM) and a natural intertidal mudflat (ITMF) in the Blackwater estuary:

C burial: NSM: -434; ITMF: -343 g C / m^2/ yr
CH_4 flux: NSM: 4,4; ITMF:8,3 g CO_2Eq / m^2/ yr
N_2O flux: NSM: 10; ITMF: 101 g CO_2Eq / m^2/ yr
CO_2 flux: not measured
Total C sequestration in CO_2eq:
NSM: -420 g CO_2Eq / m^2/ yr
ITMF: - 234 g CO_2Eq / m^2/ yr

In 2012 this sequestration is valued 28€/ton CO_2-eq * 420*10000/1000000 ton/ha.year= €118/ha.year for the natural salt marsh and 65.5€/ha.year (28€/ton CO_2-eq *234*10000/1000000) for the natural intertidal mudflat.

6.2 Disturbance prevention or moderation (services 6-8)

Europe has suffered over 100 major damaging floods in recent years. It has been estimated that since 1998 floods have resulted in about 700 fatalities, the displacement of about half a million people and at least €25 billion in insured economic losses. In addition, floods can also have negative impacts on human health. For example, substantial health implications can occur when floodwaters carry pollutants, or are mixed with contaminated water from drains and agricultural land.

It is also widely acknowledged that the flooding risk in Europe is increasing as a result of climate change - i.e. due to higher intensity of rainfall as well as rising sea levels (IPCC 2001).

Preventing a flood event or reducing the severity of flood events is thus an important service.

6.2.1 Information needed

- Hydrologic/hydrographic and hydraulic data characterizing the potential flood events (hydrologic models)
- Land use data (agriculture, number of houses, buildings etc) in the potentially flooded areas
- Depth/damage functions for different land use categories in the potentially flooded area (e.g. what is damaged if flood water reaches 1 m)

6.2.2 Identification

Estuarine ecosystems can potentially store additional flood water and as such prevent flood events and damages elsewhere. They can also reduce the water current or the wave intensity which also has an impact on flood events.

Regulation extreme events or disturbance: Water current reduction

Habitat	Qualitative importance per estuarine zone*			
	Freshwater	Oligohaline	Mesohaline	Polyhaline
Marsh habitat	3	4	3	3
Intertidal flat habitat	3	3	3	3
Intertidal steep habitat	2	2	2	2
Subtidal shallow habitat	3	3	3	3
Subtidal moderately deep habitat	2	2	2	2
Subtidal deep habitat	1	2	2	2
Adjacent land				
** Score from 1 (not important) to 5 (very important)*				

Regulation extreme events or disturbance: Flood water storage

Habitat	Qualitative importance per estuarine zone*			
	Freshwater	Oligohaline	Mesohaline	Polyhaline
Marsh habitat	4	5	3	3
Intertidal flat habitat	3	4	2	2
Intertidal steep habitat	3	3	2	2
Subtidal shallow habitat	2	2	2	2
Subtidal moderately deep habitat	2	2	2	2
Subtidal deep habitat	1	2	2	2
Adjacent land				
** Score from 1 (not important) to 5 (very important)*				

Regulation extreme events or disturbance: Wave reduction

Habitat	Qualitative importance per estuarine zone*			
	Freshwater	Oligohaline	Mesohaline	Polyhaline
Marsh habitat	3	4	3	3
Intertidal flat habitat	3	3	3	4
Intertidal steep habitat	3	3	3	3
Subtidal shallow habitat	2	2	2	2
Subtidal moderately deep habitat	1	1	1	1
Subtidal deep habitat	1	1	1	1
Adjacent land				
** Score from 1 (not important) to 5 (very important)*				

To obtain the qualitative score for this service take the average score of the above tables.

6.2.3 Quantification and monetary valuation

Services related to disturbance prevention or moderation can reduce flood risk. The benefits of flood alleviation comprise the flood damage averted in the future as a result of schemes to reduce the frequency of flooding or reduce the impact of that flooding on the property and economic activity affected, or a combination of both. This is reflected in less material and immaterial damages.

The methodology for assessing the benefits of flood prevention includes an assessment of risk in terms of the probability or likelihood of future floods to be averted and a vulnerability assessment in terms of the damage that would be caused by the floods and therefore the economic saving to be gained by their reduction (FHRC 2010).

A wide range of methodologies is available to estimate the benefits of flood prevention. We refer to FHRC 2010 the benefits of flood and Coastal Risk management: a handbook of assessment techniques 2010 for a stepwise approach to assess the benefits of flood prevention.

For Flanders, MOW-WL developed the LATIS-method to quantify and value avoided flood risks. Hydrogic models are used to create flood maps. The flood maps give information on the magnitude of the flood and the water depth for a given chance of occurance (e.g. 1/1000 years). This is performed for different chances of occurance. These maps are used as input for economic and human damage (potential casualties) estimations. The LATIS-method starts from a maximum damage calculation of an area depending on the land use and the replacement value (damage as if everything would be destroyed). Next, it estimates how much is actually damaged due to specific flood events. This is reflected in damage functions that indicate the percentage of the replacement value at risk as a function of the inundation depth.

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 benefit is equal to the reduced annual flood risk with and without the estuarine ecosystems.

It is not possible to translate the assessment methods into easily applicable indicators that can be applied in different estuaries.

6.2.4 Illustration

The tides from the Scheldt river 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 was 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.

The results of the cost benefit analysis show flood protection benefits of flood areas (and especially controlled reduced tide areas) could be quit high (Broekx et al. 2011). Measures evaluated include a storm surge barrier, dyke heightening and additional floodplains with or without the development of wetlands.

The total safety benefits of an optimal scenario, combining 24km dyke heightening and the construction of 1325ha additional floodplains were estimated at 737 million euro or approximatelty 30 million a year. This reduces the flood risk with 78%.

6.3 Regulation of waterflows (service 9-12)

6.3.1 Information needed

This service is considered as a supporting service and the impact is included in the valuation of other services (see 6.3.2).

6.3.2 Identification

Estuaries play an important role in the regulation of flow in the lower part of river streams. From an ecosystem functioning point of view, the water flow regulation service is largely determined by the combined effect of bathymetric and surface characteristics (i.e. soil type, vegetation cover).

Water flow in estuaries is forced by several processes. Tidal forcing at the seaward side and river discharge(s) at the upstream side are the most obvious hydrodynamic forcing processes. These hydrodynamic forcing conditions and the estuary's morphology determine the flow pattern in an estuary. In addition to the tidal flow and river flow as such, density stratification can occur as a result of insufficient mixing of the river's freshwater inflow with the saline sea water. Furthermore, water flow in estuaries might also be affected by the presence of marsh vegetation or hydraulic structures such as groins, quay walls and dykes, especially on a smaller scale.

The hydrodynamic characteristics of flow/currents in an estuary are of great importance for its ecosystem services. In general, all ecosystem services are dependent on the flow regulation of an estuary, as the entire ecosystem is related to the occurring hydrodynamic conditions. Alteration and regulation of water flows can have significant consequences on these ecosystem services. Examples of ecosystem services that are regulated by water flow and are directly affected by changes in hydrodynamic conditions are:

- Presence of specific ecosystems themselves;
- Drainage of river catchment and surrounding polders;
- Energy dissipation of storm surges and river peak discharges;
- Safety, i.e. protection level and stability of dykes or other sea defense structures;
- Shipping, i.e. availability and safety for navigation in estuary channels.

- Water for industrial and agricultural use;

Ecosystems and habitats

The presence of specific estuarine ecosystems and habitats does directly depend on the hydrodynamic conditions, such as water level variations and current speeds. For instance, flooding frequencies of intertidal areas determine the type and density of vegetation cover in tidal marshes. Changes in the hydrodynamic conditions, due to sea level rise or changes in basin geometry, might in the worst case lead to siltation or drowning of these areas. In addition, the balance between fresh, brackish and salt water in the hydrological system is of importance for the flora and fauna present in the estuarine ecosystems (see chapter 7)

Drainage of polders and river catchment

Part of the water flowing through estuaries is that of the upstream river input, meaning that drainage of the river catchment is one of the flow regulation services of an estuary. Besides, drainage of surrounding polders is a flow regulation service of rivers and estuaries as well. Whether and to what extent surrounding polders can be drained gravitationally depends on the hydrodynamics in the estuary. Higher water levels in a river or estuary could lead to a decrease in natural gravitational drainage, and hence an increase in demand for pumping capacity.

Energy dissipation

Tidal flats and marshes in estuaries play an important role in the dissipation of energy of for instance tides, storm surges and river floods due to bottom friction. The same holds for energy dissipation of shorter waves and protection for wave impact on shores and sea defenses. This way, estuaries can protect coastal areas from extreme hydrodynamic conditions during river floods and storm surges. Energy dissipation on tidal marshes is generally higher than on tidal flats, as vegetation provides for additional bottom friction. The potential of intertidal areas to reduce storm surges depends on characteristics of individual storms and on local landscape characteristics such as vegetation, elevation and the presence of structures (see for instance: Wamsley et al. 2010) (see chapter 6.2).

Safety

Hydrodynamic conditions in estuaries and rivers form the basis for the design of dykes and other flood protection structures. The protection level of these sea defenses is directly related to the estuary's hydrodynamics. Hence, changes in the flow regulation (water levels, current speeds, wave conditions) of an estuary directly affect the flood protection level of surrounding areas. Allowing for higher water levels in an estuary reduces the protection level of dykes and sea defense structures. Besides, prevailing wave conditions might become more severe when the water depth increases. Too low water levels might on the other hand lead to geotechnical stability problems of some hydraulic structures such as quay walls (see chapter 6.2).

Navigation

Navigation is another service that is dependent on the flow regulation in an estuary. In shallow estuary channels and above natural sills, navigation might only be possible at high water. Similarly, ships with a large draught can only access certain channels during high water, when the water depth is sufficient. High water levels might on their turn limit the possibilities for navigation under bridges. The magnitude and direction of flow in estuarine channels do also influence the possibilities and costs for shipping through an estuary. Flow regulation can be used to improve possibilities for safe navigation in estuaries. An example is providing for longer time frames to safely pass obstacles like sills or bridges, by dredging shipping lanes or altering the estuarine hydrodynamics as such (see chapter 5.4).

Water availability

Flow regulation of an estuary is important for the availability of water and for the quality of the available water. This can for instance be of importance for the possibility to obtain and release cooling water for energy plants and industrial purposes (see chapter 5.3). Other ecosystem services that are directly influenced by the availability and quality of water in estuaries are providing drinking water or making water available for agricultural purposes.

Water quantity regulation: dissipation of tidal and river energy

Habitat	Qualitative importance per estuarine zone*			
	Freshwater	Oligohaline	Mesohaline	Polyhaline
Marsh habitat	2	3	3	3
Intertidal flat habitat	3	4	4	4
Intertidal steep habitat	2	2	2	2
Subtidal shallow habitat	3	4	4	4
Subtidal moderately deep habitat	3	3	3	3
Subtidal deep habitat	1	2	2	2
Adjacent land				
** Score from 1 (not important) to 5 (very important)*				

Water quantity regulation: drainage of river water

Habitat	Qualitative importance per estuarine zone*			
	Freshwater	Oligohaline	Mesohaline	Polyhaline
Marsh habitat	2	2	2	2
Intertidal flat habitat	3	2	2	2
Intertidal steep habitat	2	2	1	1

Subtidal shallow habitat	2	3	3	3
Subtidal moderately deep habitat	2	3	3	3
Subtidal deep habitat	2	3	3	3
Adjacent land				
** Score from 1 (not important) to 5 (very important)*				

Water quantity regulation: landscape maintenance

Habitat	Qualitative importance per estuarine zone*			
	Freshwater	Oligohaline	Mesohaline	Polyhaline
Marsh habitat	4	4	4	4
Intertidal flat habitat	3	4	4	4
Intertidal steep habitat	2	2	2	2
Subtidal shallow habitat	3	4	4	4
Subtidal moderately deep habitat	2	3	3	3
Subtidal deep habitat	1	2	2	2
Adjacent land				
** Score from 1 (not important) to 5 (very important)*				

Water quantity regulation: transportation

Habitat	Qualitative importance per estuarine zone*			
	Freshwater	Oligohaline	Mesohaline	Polyhaline
Marsh habitat	1	2	2	2
Intertidal flat habitat	1	2	2	2
Intertidal steep habitat	1	2	2	2
Subtidal shallow habitat	1	2	2	2
Subtidal moderately deep habitat	3	3	3	3
Subtidal deep habitat	5	5	5	5
Adjacent land				
** Score from 1 (not important) to 5 (very important)*				

6.3.3 Quantification

The hydrodynamic conditions in an estuary can be characterized by several parameters, such as water levels, discharges, current velocities or the tidal prism. Another important aspect for especially the interaction with the basin morphology is the asymmetry of the tide, which is largely dependent on the estuary morphology (Dronkers 1986). A general overview of some important hydrodynamic characteristics is given below. Empirical equilibrium relationships are available in literature for the ratio between hydrodynamic and geometric characteristics. Townend (2005) gives a brief overview of such relationships and an application to some UK estuaries. However, it is highly recommended that numerical modeling is used to determine the detailed effect of measures in an estuary on water flow regulation.

<u>Water levels</u>
Water levels are generally given with respect to mean sea level (MSL) or local ordnance levels such as the Dutch ordnance level (NAP). The water levels imposed by the tide have a spring-neap variation, implying that tidal water levels vary throughout time. Important water level

parameters are mean low water spring (MLWS) and mean high water spring (MHWS), representing the average spring tide conditions. The difference between consecutive low and high waters is referred to as the so-called tidal range. As the tide propagates through the estuary, reduction in water depth and basin width slows the progress of the tidal wave, leading to an increase in amplitude. Conversely, increasing friction reduces the tidal range. The tidal range in the Western Scheldt estuary has its maximum of nearly 5.5 m at approximately 75 km from the estuary mouth, close to the port of Antwerp (Wang et al. 2002; Vandenbruwaene et al. 2012). In the Elbe estuary, the maximum tidal range of 3.6 m occurs near Hamburg. The maximum tidal range in the Weser estuary of nearly 4 m is located near the city of Bremen. In the Humber estuary, the maximum tidal range is about 5 m, close to Hull (Vandenbruwaene et al. 2012).

The above parameters and values do apply to tidal flow only. River discharge can increase water levels in an estuary, which can be of importance for levee heights or periodic flooding of tidal marshes with a higher elevation. Especially peak discharges as a result of heavy rainfall or snowmelt upstream can lead to significantly higher water levels in rivers and estuaries. The same holds for storm surges, which can push large amounts of water into an estuary, leading to significantly higher water levels.

Flow velocities

Flow velocities, and especially their variation in magnitude and direction, are of importance for the estuary's morphological behavior and sediment transport. The variation in tidal current is also referred to as the horizontal tide. The duration of the slack tide, the period in which flow reverses and velocities are close to zero, is also of great importance for the net sediment transport as fine particles settle during this period. Flow velocities vary largely over the different parts of an estuary. For instance, the magnitude of currents in channels is generally much higher than the flow velocities over intertidal flats and in marshes. Hydrodynamic models and/or field measurements are needed to predict these spatially varying flow velocities in estuaries over a tidal cycle and under different hydrodynamic forcing conditions.

Tidal asymmetry

Asymmetry of the tidal wave is of importance for residual sediment transport in estuaries, as differences in maximum currents during ebb and flood cause residual transport of courser material. Systems in which the flood period is shorter and flood velocities are higher than ebb velocities are called flood-dominant, because this situation enhances net sediment transport in flood direction. Similarly, systems in which the ebb duration is shorter and ebb velocities are higher are called ebb-dominant. Tidal asymmetry inside an estuary is controlled by basin geometry and the asymmetry of the tide at the seaward boundary of the estuarine system (Dronkers 1986, 1998; van der Spek 1997). Similarly, the asymmetry of the tide at a certain point in an estuary is a result of the asymmetry of the tide in downstream sections (Wang et al. 2002). Several indicators for the relation between tidal asymmetry and estuarine morphology have been proposed in literature. For instance, Friedrichs and Aubrey (1988) use the ratio of intertidal storage volume over channel volume. Dronkers (1986) uses the wet surface area at high water over the wet surface area at low water. All relationships show a tendency in which systems with relatively shallow channels and small intertidal storage areas are likely to be flood-dominant and systems with deep channels and large intertidal storage area tend to be ebb-dominant. Fortunato and Oliveira (2005) show that ebb-dominance is largest when tidal flats are at MSL or above. Ebb-dominance can also be enhanced by the presence of a river flow that increases the ebb velocities and decreases the flood velocities. Conversely, density stratification due to the presence of fresh water river inflow counteracts net sediment import due to ebb-dominance.

A second type of asymmetry is that of the slack water periods. Differences between slack water periods after flood and ebb influence residual sediment transport of especially finer materials (Dronkers 1986). Estuaries or tidal basins with relatively shallow channels and intertidal storage areas below MSL generally have a longer high water slack than low water slack, enhancing the import of fines. Basins with intertidal storage areas above MSL and relatively deep channels do on the other hand have a shorter high water slack. The second situation would theoretically enhance export of (fine) sediment. However, a counteracting effect may occur in basins with large intertidal storage areas. The small water depth on intertidal flats during high water slack can lead to much higher sedimentation during high water slack than sedimentation during the longer low water slack, as settling of fines is generally stronger in small water depths. The latter effect often dominates the effect of the shorter high water slack period, in particular when the wet surface area at HW is significantly larger than the wet surface area at LW. This is one of the reasons why tidal basins and estuaries are often accumulation areas for fine sediments (silt).

In general, distortion of the tidal wave has a direct influence on sediment transport patterns in estuaries. Changes in basin geometry affect this tidal asymmetry and do consequently alter the hydrodynamic conditions (i.e. water levels and flow patterns) and sediment transport pattern in an estuary. An indication of possible effects is given by Van der Spek

(1997), who has analyzed the effect of historical land reclamations and other morphological changes on the hydrodynamics in the Western Scheldt.

6.3.4 Valuation

The valuation of the services of water flow regulation and water provision is very site specific. This depends on the ecosystem extent, state and functions. Besides, the value also depends on the available alternatives for water provision or transportation.
When valuing flow regulation separately, careful consideration should be given to potential double-counting, especially with the provisioning services (water for industry, water for transportation).

An economic benefit related to water regulation and not accounted for in other ecosystem services is the impact on maintenance costs of infrastructure. More steady hydrodynamic conditions due to the presence of estuarine ecosystems may lead to less maintenance costs of protecting structures such as dykes. Additionally, too low water levels might lead to geotechnical stability problems of some hydraulic structures such as quay walls. This can be valued if information is available on existing maintenance costs.

6.4 Water quality regulation

The filter function of estuaries is considered as one of the most valuable ecosystem services and can be regarded as a natural complementary waste water treatment service (Costanza 1997, Dähnke et al. 2008). In this study, the quantification of the ecosystem service for waste-water treatment is restricted to nitrogen removal due to time constraints. If you have information on phosphor removal, you may include this making use of the shadow prices of phosphor.

6.4.1 Information needed

- Measured denitrification, anammox and burial rates
- Or IF you don't have these data: use the table with indicator data
- Area/habitat (m^2) and volumes (m^3) of estuarine sections (flat-marsh-estuary)
- Shadowprices for nitrogen removal

6.4.2 Identification

It was attempted to synthesize available knowledge on biogeochemical cycling resulting in nitrogen removal in temperate estuarine environments.

Losses within the estuary (before the ocean is reached) are mostly attributed to classical denitrification (Davidson & Seitzinger 2006). Burial, anammox and other newly discovered pathways of nitrogen removal have been shown to be of limited significance in estuaries (Jickells & Weston 2011). Assimilation can be an important temporary sink, however is not considered as a long term sink within this study.

It has to be noted that losses in estuaries are balanced by nitrogen generating processes such as ammonification, nitrification and nitrogen fixation. Furthermore, there are also nitrogen transforming processes, e.g. dissimilatory nitrate reduction to ammonium (DNRA). Within this study, only nitrogen losses are considered as indicator for the regulating ecosystem service of waste water treatment. More background information on nitrogen transformation processes and general estuarine ecological functioning can be found in Geerts et al. (2013).

For this study it was not possible to use the same habitat classification, since the exact details of the studied habitats are not known. Instead a more general distinction was made: tidal marsh, intertidal flats and the water column (pelagic). It was also impossible to distinguish freshwater and oligohaline habitats and polyhaline and marine; the salinity zonation used is freshwater (0-5 PSU), brackish (5-18 PSU) and salt (18 – 50 PSU).

Water quality regulation: reduction of excess loads coming from the catchment

Habitat	Qualitative importance per estuarine zone*			
	Freshwater	Oligohaline	Mesohaline	Polyhaline
Marsh habitat	5	4	4	4
Intertidal flat habitat	3	3	3	3
Intertidal steep habitat	2	2	2	2
Subtidal shallow habitat	3	3	3	3
Subtidal moderately deep habitat	2	2	2	2
Subtidal deep habitat	2	2	2	2
Adjacent land				
** Score from 1 (not important) to 5 (very important)*				

Water quality regulation: transport of pollutants and excess nutrients

Habitat	Qualitative importance per estuarine zone*			
	Freshwater	Oligohaline	Mesohaline	Polyhaline
Marsh habitat	2	2	2	3
Intertidal flat habitat	2	2	2	2
Intertidal steep habitat	2	2	2	2
Subtidal shallow habitat	3	3	3	3
Subtidal moderately deep habitat	3	4	4	4
Subtidal deep habitat	4	5	5	5
Adjacent land				
** Score from 1 (not important) to 5 (very important)*				

6.4.3 Quantification

Numbers of main nitrogen loss processes in different northern temperate estuaries are summarized as much as possible by means and ranges (min, max) in tables below taking into account salinity (fresh-brackish-salt-estuary) and habitat (flat-marsh-pelagic-estuary).

Uncertainties in using the numbers delivered in the tables below for further up scaling and valuation at the ecosystem level are given by the fact that the various processes involved in nitrogen removal vary spatially and temporally in function of various influencing factors. This does not only vary within an estuary, but also between estuaries.

Next, different methods have been used to estimate the rate of these processes. Each method has its own assumptions and implications. Some give an underestimation while others rather give overestimation depending on the method used. Hence, it is debatable to simply aggregate numbers from several studies. Nevertheless, the range of nitrogen removal can be indicated this way.

Furthermore, not all studies use the same units and sometimes numbers simply cannot be converted to one another, because of lack of specifications within the respective studies (e.g. the area upon which the denitrification rate is measured).

Finally, there are still several knowledge gaps in finding general trends.

Table 13: Numbers for nitrogen removal in the freshwater part of the estuary.

salinity	habitat	proces	Units	m	min	max	refs
Fresh	Flat	Anammox	-	-	-	-	-
		Denitrification	mmol N/(m².d)	9.79	1.21	31.61	(18, 19, 20)
		Burial	-	-	-	-	-
	Marsh	Anammox	-	-	-	-	-
		Denitrification	mmol N/(m².d)	28.8	9.6	48	(47)
		Burial	-	-	-	-	-
	Pelagic	Anammox	-	-	-	-	-
		Denitrification	mmol N/(l.d)	0.68	<0.01	1.56	(1, 2, 3)
			mmol N/(m².d)	312	-	-	(1)
			kton N/d	1450	650	2400	(3)
		Burial	-	-	-	-	-

The freshwater part of the estuary comprises both the freshwater and the oligohaline zone according to the Venice classification. References 1: Abril et al. 2000, 2: Sebilo et al. 2006, 3: Vanderborght et al. 2007, 18: Ogilvie et al. 1997, 20: Rysgaard et al. 1999

Table 14: Numbers for nitrogen removal in the brackish part of the estuary.

salinity	habitat	proces	Units	m	min	max	refs
Brackish	Flat	Anammox	nmol N/(ml$_{wet\ sed}$.h)	1	-	-	(25)
		Denitrification	mmol N/(m².d)	73.67	0.7	470.78	(19, 20, 22, 23, 24, 26, 27)
		Burial	-	-	-	-	-
	Marsh	Anammox	-	-	-	-	-
		Denitrification	-	-	-	-	-
		Burial	-	-	-	-	-
	Pelagic	Anammox	-	-	-	-	-
		Denitrification	mmol N/(l.d)	11.06	0.05	35	(3, 5, 6)
			% NO_3^-	39	11	67	(7)
		Burial	-	-	-	-	-

The brackish part of the estuary corresponds to the mesohaline zone according to the Venice classification. References 3: Vanderborght et al. 2007, 5: Abril et al. 2010, 6: Billen et al. 1985, 7: Dähnke et al. 2008, 19: Barnes et al. 1998, 20: Rysgaard et al. 1999, 22: Cabrita & Brotas 2000, 23: Trimmer et al. 2000, 24: Dong et al. 2000, 25: Trimmeret al. 2003, 26: Nielsen et al. 1995, 27: Thornton et al. 2007

Table 15: Numbers for nitrogen removal in the salt part of the estuary.

salinity	habitat	proces	Units	m	min	max	refs
Salt	Flat	Anammox	nmol N/(ml$_{wet\ sed}$.h)	6	1,5	10	(25)
			nmol N/(cm³.h)	4	0.8	6	(35)
		Denitrification	mmol N/(m².d)	1.93	0	29.38	(14, 18, 20, 22, 23, 27, 28, 30, 32, 33, 34, 36, 37, 38, 39)
			mmol N/(l.d)	1.11	0.003	3.00	(31, 35)
			nmol N/(g$_{wet\ sed}$.h)	10.46	0.03	38	(33)
			nmol N/(ml$_{wet\ sed}$.h)	7.27	4	10	(25)
		Burial	mmol N/(m².d)	5.48	0.00021	35.18	(28, 38, 42, 43, 44)
	Marsh	Annamox	-	-	-	-	-
		Denitrification	mmol N/(m².d)	4,455	3.77	5.14	(51)
		Sedimentation	mmol N/(m².d)	3.36	0	11.53	(38, 52)
	Pelagic	Annamox	-	-	-	-	-
		Denitrification	mmol N/(l.d)	6.27	0.03	20	(2, 3)
			kton N/d	14.48	2.8	29.42	(3)
		Burial	-	-	-	-	-

The salt part of the estuary corresponds to the mesohaline zone according to the Venice classification.

References 2: Sebilo et al. 2006, 3: Vanderborght et al. 2007, 11: Allen 1997, 14: Hofmann et al. 2008, 18: Ogilvie et al. 1997, 20: Rysgaard et al. 1999, 22: Cabrita & Brotas 2000, 23: Trimmer et al. 2000, 25: Trimmer et al. 2003, 27: Thornton et al. 2007, 28: Middelburg et al. 1995, 30: Rysgaard et al. 1995, 32: Risgaard-Petersen 2003, 33: Texeira et al. 2010, 34: Rocha & Cabral 1998, 35: Risgaard-Petersen 2005, 36: Risgaard-Petersen 2000, 37: Jensen et al. 1996, 38: Jickells et al. 2000, 39: Nielsen et al. 2001, 42: Adams et al. 2012, 43: Andrews et al. 2006, 44: Andrews et al. 2008, 51: Erikson et al. 2003, 52: Caçador et al. 2007

Table 16: Numbers for nitrogen removal for the entire estuary.

salinity	habitat	proces	Units	m	min	max	refs
Estuary	Flat	Annamox	-	-	-	-	-
		Denitrification	mmol N/(m².d)	0.975	0.4	1.55	(26)
		Burial	ton N/yr	3000	-	-	(6)
			mmol N/(m².d)	1.57	0.21	2.93	(12)
	Pelagic	Annamox	-	-	-	-	-
		Denitrification	mmol N/(m².d)	5.53	0.13	15.37	(12, 13, 14, 15)
			% TN	21	-	-	(13)
			% DIN	33	-	-	(13)
		Burial	-	-	-	-	-

References 6: Billen et al. 2007, 12: Dettmann 2001, 13: Middelburg & Nieuwenhuize 2001, 14: Hofmann et al. 2008, 15: van Beusekom & de Jonge 1998, 26: Nielsen et al. 1995

6.4.4 Valuation

The cost for lowering one unit of pollution can be used as a proxy for the value for the service. This is called the shadow price. A recent literature review on shadow prices shows the following ranges:

Reference	Shadowprice (€/kg N)	Shadowprice(€/kg P)
Broekx et al. 2008	74	800
Wisconsin department of natural resources 2012		$23,56/pound
Byström 2000	13.2-17.2	
Hernandez 2011	35,2	82.5
Molnos-Senante 2010		42.7
Corcoran et al. 2010	4.6-65.2	7.5-103.4
De Bruyn et al. 2010	7-12.5	1.8-10.9
DEFRA 2003		52.2
Windolf et al. 2012	40-189	

Best is to find out shadowprices for N, P in your own country based on river basin management plans and cost efficiency tools. If these are not available the following indicator ranges can be used based on the literature review:

Nitrogen: 5-65€/kg N
Phosphor: 8-103€/kg P

For Flanders we make use of the environmental cost model water (MKM) to estimate the avoided costs of taking technical measures for water treatment by maintaining the estuarine ecosystem (Cools et al. 2011; Broekx et al. 2008). High shadow prices are an indication of the difficulty for reaching the water quality standards in urbanised areas and the necessity of costly measures to implement.

6.4.5 Illustration

In the Scheldt estuary the OMES-model calculated the extra nitrogen retention and denitrification rates for the different alternitives of the SIGMA-plan (a flood safety plan for the river Scheldt). The plan exists of controlled inundation areas flooding sometimes and controlled reduced tidal areas flooding with every tide. Average values show improved denitrification of 176 kg N/ha.year for fresh water marshes and 107 kg N/ha.year for salt water marshes and an average N-burial of 252 kg/ha.year. This means a yearly benefit between 2140 and 27800€/ha for freshwater marshes and between 1800 and 23300€/ha for salt marshes.

6.5 Erosion prevention and sediment retention

Erosion is commonly defined as the displacement of solids (e.g. sediment and soil) and other particles by wind or water. Erosion is a natural process, but is heavily increased by specific types of land use, in particular by intensive and inappropriate land management practices such as deforestation, overgrazing, unmanaged construction activity and road-building.

Managed areas, e.g. areas used for the production of agricultural crops, generally experience a significant greater rate of erosion than areas under natural vegetation. This capacity of natural ecosystems to control soil erosion is based on the ability of vegetation (i.e. the root systems) to bind soil particles, thus preventing the fertile topsoil from being blown or washed away by water or wind.

Sedimentation is defined as the net retention of sediments carried in suspension by waters inundating the estuarine nature area.

6.5.1 Information needed

- Deposition and erosion rates in the estuary
- Prices for dredging

6.5.2 Identification

Erosion and sedimentation due to water flow occur continuously within the subtidal and intertidal areas of an estuary. Controlling these erosion and sedimentation processes is important to guarantee water levels throughout the estuary. This can for instance be beneficial for protection of surrounding land against floods, but also to limit the need for maintenance dredging. These two examples already indicate that sedimentation might be beneficial in case of some purposes or areas, while erosion is beneficial for other services that the estuary provides. Hence, the value of erosion and sedimentation in an estuary can be spatially varying. Interaction between the prevailing hydrodynamic conditions and the estuary’s geomorphology determines the erosion- and sedimentation pattern on estuary scale (Dronkers 1998).

Important factors are:

- Vegetation: On a small scale, the capacity of natural ecosystems to control soil erosion and sedimentation processes is largely based on the ability of vegetation (i.e. its root systems) to bind soil particles and reduce wave energy and current velocities. Through this, vegetation helps preventing the fertile topsoil from being washed away and it can enhance sediment accumulation. Different habitat types with varying types and extents of vegetation cover play different roles in sediment retention and erosion prevention.

- *Management:* Benefits from sediment deposition could be influenced by projects that alter flooding frequencies and sedimentation rates, such as embankments, dike reinforcements and port constructions. Such projects generally affect the intertidal flat and marsh surface area and its morphological development. Direct anthropogenic alterations on the morphology, such as dredging activities, do also influence sedimentation and erosion patterns in a river or estuary. Projects that transform agricultural land into natural land (e.g. wetlands) have a decreasing effect on soil erosion, as wetlands have a better soil binding capacity. This is also the case for the application of natural field edges and ditch edges (Ruijgrok 2006). The broader the vegetated strip, the more energy will be absorbed by the vegetation.

- *Hydrodynamics:* The sediment retention service and the erosion prevention service of estuarine ecosystems are directly related to natural morphodynamic processes (i.e. erosion and sedimentation). Large-scale sedimentation and erosion patterns in estuaries are determined by the tidal hydrodynamics. These tidal flow conditions in an estuary are the result of an interaction between the tide at the seaward boundary and the geomorphology of the estuary (see also section flow regulation; Dronkers 1986; Wang et al. 2002).

- Other important aspects for the rates of erosion or sedimentation are the fluvial input (i.e. upstream sediment influx) and the availability of sediment offshore in ebb-tidal deltas or adjacent coasts. It should be noted that whenever an estuary imports sediment from offshore, this could possibly lead to erosion and coastline retreat in the coastal cells.

Erosion and sedimentation regulation by biological mediation

Habitat	Qualitative importance per estuarine zone*			
	Freshwater	Oligohaline	Mesohaline	Polyhaline
Marsh habitat	4	4	4	4
Intertidal flat habitat	4	3	3	3
Intertidal steep habitat	2	2	2	2
Subtidal shallow habitat	3	3	3	3
Subtidal moderately deep habitat	1	2	2	2
Subtidal deep habitat	1	2	2	2
Adjacent land				
** Score from 1 (not important) to 5 (very important)*				

Erosion and sedimentation regulation by water bodies

Habitat	Qualitative importance per estuarine zone*			
	Freshwater	Oligohaline	Mesohaline	Polyhaline
Marsh habitat	4	5	4	4
Intertidal flat habitat	5	4	4	4
Intertidal steep habitat	2	3	2	2
Subtidal shallow habitat	5	5	4	4
Subtidal moderately deep habitat	4	4	4	4
Subtidal deep habitat	4	4	4	4
Adjacent land				
** Score from 1 (not important) to 5 (very important)*				

6.5.3 Quantification

For management purposes, one is interested in the net sedimentation and/or erosion over a longer period of time (years, decades). It is common practice to express the net erosion or deposition of sediment as volumes of solid grains (m^3). Similarly, sediment transport is defined in transport rates (m^3/s or m^3/yr). Sedimentation and erosion can also be expressed in bed level changes (m).

Deposition and erosion rates can vary enormously throughout an estuary and especially between different types of estuarine environments. Here, a division is made between (vegetated) marshes and mudflats, channels and the river system. Ruijgrok (2006) calculates sediment deposition by multiplying the habitat surface (ha) with the average sediment deposition in cubic meter per hectare per year ($m^3/ha/yr$).

Sediment deposition (m^3/yr) = habitat surface (ha) × average sediment deposition ($m^3/ha/yr$)

Once long-term surface level differences are known, one can easily calculate the sediment accumulation in a marsh or estuary, using the above formula.

Sediment deposition on estuary mudflats and marshes

The vertical accretion rate of marshes and mudflats depends on the establishment and presence of vegetation, as vegetation has the ability to trap sediment (Allen 2000). It is important to account for sea level rise when estimating sediment accumulation on mudflats and salt marshes. Normally, the elevation of salt marshes and tidal flats will follow this sea level rise if the availability of sediment is sufficient. Examples are British salt marshes and the Wadden Sea marshes, which indeed appear to be in balance with sea level rise (Allen 2000; Dijkema et al. 1990; French 1993). For young marshes, vertical accretion may well be higher than just sea level rise (Allen 2000 Allen (1990) states that accretion rates develop from order 10 cm/yr for young marshes to about 0.01 cm/yr after hundreds to thousands of years.The lowest accretion rates are thus found for mature marshes that just have to keep up with sea level rise.

In order to calculate sediment accumulation on marshes and mudflats, Ruijgrok (2006) uses the sediment deposition rates that are given in table 17. These deposition rates are based on marsh growth rates of ca. 2 cm/yr for young marshes and ca. 1 cm/yr for old marshes. In reality however, sediment accumulation on marshes and mudflats varies strongly, both locally and between estuaries. It could therefore be difficult to estimate for a project the impact on marshlands and especially on intertidal mudflats. To improve the accuracy of this kind of estimations, it is recommended to use morphological model calculations executed by experts (see below).

Table 17: Average sediment deposition, from Ruijgrok (2006).

Habitat type:	Deposition Rate (m^3/ha/y)
Soil related:	
Mudflat and young marshes	200
Old marshes	100
Water related:	
River (brackish)	30
River (fresh water)	30

Sediment deposition and erosion in estuary channels

Erosion and sedimentation in estuarine channels can be quantified with the help of equilibrium relationships between channel cross-section and tidal prism (Townend 2005). If channels are relatively deep compared to the tidal prism, sediment accumulation is likely to occur. Conversely, if the tidal prism is large compared to the channel cross-section, channel deepening and widening will likely occur. Large-scale changes to the estuary's geomorphology will alter the hydrodynamic conditions, amongst which the tidal prism, and hence affect the cross-sectional properties of estuarine channels. Numerical modeling and field measurements are needed to determine the effects for sedimentation and erosion in the main estuary channels.

Sediment deposition and erosion in rivers

Similar as for deposition on marshes and mudflats, Ruijgrok (2006) also gives average deposition rates for rivers in both brackish as well as fresh water environments (see table 17). The given average amount of 30 m^3/ha/yr sediment deposition is based on a sedimentation rate in river floodplains of 36.6 ton/ha/yr, which is the result of a large number of model calculations and measurements for Maas, Rijn and Waal by different experts (from Ruijgrok 2006). Accumulation rates in river environments may be higher as the fluvial sediment input is high compared to estuaries. Again, sediment deposition varies per location and per management action. Therefore, it is recommended to use hydrodynamic and morphological modeling (see below).

Erosion control
The benefit of erosion control measures can be quantified by multiplying the habitat surface (ha) with the average avoided sediment transport to the river in cubic meter per hectare per year (m^3/ha/yr). This can be prevented erosion on river beds, estuarine channel and shoals, but also erosion prevention on banks.

Erosion control (m^3/y) = habitat surface (ha) × average erosion control (m^3/ha/yr)

Furthermore, erosion control measures could lead to a reduction of design water levels and wave heights for the conventional bank protection. These reductions are site-specific and depend on the prevailing hydrodynamic conditions and characteristics of the vegetation strips or reed belts. Field measurements are needed to quantify the local reduction in wave height and water level.

Numerical modeling

Hydrodynamic and morphological models can be used to obtain a better insight in the erosion and sedimentation processes in estuarine environments. Given the large spatial variability and complexity of these processes, it is highly recommended to use such models for the quantification of sediment accumulation and erosion. Examples of hydrodynamic models that can resolve water flow and sediment transport equations are DELFT3D, TELEMAC-MASCARET and MIKE21. All these models are validated for estuarine environments. Another option for determining sediment accumulation on marshlands is to use the zero-dimensional physically based MARSED model, which computes vertical accretion of particular marshes based on the environmental conditions (Temmerman et al. 2004).

6.5.4 Valuation

Sediment retention and ersosion prevention can be valued with the avoided cost method. Due to sediment retention in marshlands and mudflats, dredging costs can be avoided elsewhere in the estuary. This benefit is closely linked with navigation, as dredging mostly happens for maintaining navigation channels. The average costs for maintenance dredging in Flanders are between €5-€10/m³ (Broekx et al. 2008). The costs for treatment of the sludge range from 20€ to a few hundred €/m³ depending on the contamination of the dredged material. It should be noted that sediment accumulation in (artificial) tidal marshes is not directly linked to the decrease in dredged volume. To calculate benefits with respect to saved dredging costs more precisely, one should best use morphological models and consult experts.

6.5.5 Illustration

Soil erosion is a common problem in Flanders, with soil losses between ca. 500 kg to 5.000 kg sediment per hectare per year on agricultural land (VMM 2003, from Ruijgrok 2006). Assuming an average mass of sediment of 1.600 kg per cubic meter, the reduction of sediment transport to the navigation channels amounts ca. 0.31 to 3.1 cubic meter per hectare per year (m³/ha/yr) if soil erosion can be prevented. If we assume that this amount would otherwise have been dredged to maintain the navigation channel, the change of one hectare of agricultural land into estuarine nature (preventing erosion) will have a benefit between 1.55 and 31 €/ha per year. This amount may be several times higher if we also take the treatment of the sludge into account.

7 Valuation methodologies for biodiversity

The group of habitat and supporting services, or simply **"biodiversity"**, is the collection of all biophysical processes and structures (eg. photosynthesis, respiration, nutrient cycling and evapotranspiration) responsible for the internal functions of an ecosystem such as the possibility for evolution, the resilience, stability and carrying capacity of ecosystems. Biodiversity, or the diversity of life, exist in different forms from genetic diversity to the variety of all living species in a certain area, such as plants, animals, bacteria, etc. That is the reason why biodiversity is often considered as a unit for the health of an ecosystem. The larger biodiversity in an ecosystem, the healthier the system and the more ecosystem services could be delivered. In general it is assumed that the delivery of ecosystem services could be guaranteed with biodiversity. The supply of many ecosystem services indeed depends on biodiversity. Regarding **provisioning services**, biodiversity are all the eatable plants and animals, but also renewable resources (such as wood) and medicinal character of plants. Regarding **regulating services**, biodiversity plays an important role in many processes such as the purification of water or the pollination of plants. Regarding **habitat and supporting services**, biodiversity itself is an ecosystem service but also (the stability of) primary production and soil fertility. Regarding **cultural services**, biodiversity plays a direct or indirect role for many leisure and touristic activities such as visiting the zoo or fishing. Biodiversity acts also as an inspiration for art, culture, science and industrial innovation. Biodiversity also contributes to human health by the medicinal power of plants, but also on the psychological health of people due to the ability to visit or to see green landscapes.

However, it is not clear to which extent biodiversity determines the ecosystem functioning and the delivery of ecosystem services. There is more and more evidence for a clearly positive relationship between diversity and function, certainly when considering multiple functions or ecosystem services. Indeed, an ecosystem fulfils different function for which mostly also different species are needed. In other words, a habitat with many species will, on average, supply more ecosystem functions compared to the same habitat with few species.

7.1.1 Information needed

For the qualitative valuation land use class types are needed.
Biodiversity could be quantified, but different units are possible.
Monetary valuation is not possible.

7.1.2 Identification

In general, estuaries are considered to be very important regions for biodiversity. Also the different estuarine habitats, in the different estuarine zones, are all important to very important for biodiversity.

Habitat	Qualitative importance for biodiversity (per estuarine zone)*			
	Freshwater	Oligohaline	Mesohaline	Polyhaline
Marsh habitat	5	5	5	5
Intertidal flat habitat	5	5	5	5
Intertidal steep habitat	3	4	4	4
Subtidal shallow habitat	4	4	4	4
Subtidal moderately deep habitat	3	4	4	4
Subtidal deep habitat	3	4	4	4
** Score from 1 (not important) to 5 (very important)*				

7.1.3 Quantification and valuation

Biodiversity is typically classified as a supporting service. This service supports other services and accounting for it means double-counting with other ecosystem services. In economic valuation, biodiversity is often not taken into account as such. Nevertheless one can also assume that biodiversity on its own has a specific value. This is often linked with the non-use value of an ecosystem. A common valuation methodology for the non-use value is the Willingness-to-pay methodology. However for valuing biodiversity this methodology is highly discussed and contested. Therefore, particularly with regard to maintaining biodiversity and the linkages within ecosystems, issues such as ensuring sustainability of a given service, preserving critical components of 'natural capital' and maintaining safe minimum standards of species populations and habitat requirement are important (Turner, 2008). Many units and indices are used to quantify biodiversity, among which number of species, number of targeted species (e.g. red list species from the Water Framework Directive), and relative species abundance.

8 Valuation methodologies for cultural services

The Millennium Ecosystem Assessment (MA) described cultural services as "the non-material benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation and aesthetic experiences" (MA 2005a p.29).

8.1 Recreational value

8.1.1 Information needed

- Number of visits in the studied area
- For valuation of visits to estuarine nature we recommend the value from a meta-analysis (i.e. 4.6€ per visit. (Sen 2011)

8.1.2 Identification

Relevant recreation and tourism related activities include, for example, hiking, biking, fishing, swimming, camping, horse riding, hunting, bird- and nature-watching. Alternatively, nature related tourism can also include visits to sites of cultural heritage.

Recreational value: Opportunities for recreation & tourism

Habitat	Qualitative importance per estuarine zone*			
	Freshwater	Oligohaline	Mesohaline	Polyhaline
Marsh habitat	3	4	4	4
Intertidal flat habitat	3	3	4	4
Intertidal steep habitat	2	3	3	3
Subtidal shallow habitat	3	3	4	4
Subtidal moderately deep habitat	4	4	4	4
Subtidal deep habitat	4	4	4	4
Adjacent land				
** Score from 1 (not important) to 5 (very important)*				

Aesthetic information

Habitat	Qualitative importance per estuarine zone*			
	Freshwater	Oligohaline	Mesohaline	Polyhaline
Marsh habitat	4	4	4	4
Intertidal flat habitat	4	4	4	4
Intertidal steep habitat	3	3	3	3
Subtidal shallow habitat	3	3	3	3
Subtidal moderately deep habitat	3	3	3	3
Subtidal deep habitat	3	3	3	3
Adjacent land				
** Score from 1 (not important) to 5 (very important)*				

This service is closely interlinked with cultural and inspirational services and aesthetic value

8.1.3 Quantification

Data may be available on the number of visits to the site, based on on-site measurements. If not, the number of visits has to be estimated. The number of visits depend on several factors such as population density, characteristics of the area (e.g. accessibility, uniqueness…), distance to the area (how further away, how less likely people are going to recreate at the site), the availability of substitutes (other natural area's that are closer to the population).

8.1.4 Valuation

The value is calculated by multiplying the number of visits with the value per visit.

There is a broad set of studies on the welfare value of a visit to green spaces. These studies are based on both revealed and stated preferences. Revealed preferences methods are based on the assumption that the value is revealed in the costs and efforts made by the recreational users, in particular the "investment" of free time and travel expenses (travel cost method). In a second approach people are asked to state how much they would be willing to pay for e.g. the creation of a forest in their neighbourhood. The exact value for each area depends on a number of factors, including methodology, type of nature, recreation type, duration of visit, income level etc.

For this study, we follow the approach of the UK NEA study (Bateman et al. 2011). Here the valuation of visits is based on a recent meta-analysis of 250 studies worldwide to value a visit to a nature area (Sen 2011).
Sen (2011) derives recreational values per visit, based on a meta-analysis of 193 studies (SEN 2010) or 243 studies (SEN 2011). The meta analysis results in a value function (expressed in $£_{2010}$ per visit), that accounts for characteristics of the site (e.g. type of habitat) , designated or not, and characteristics of the study (e.g. in or outside UK). We take the values for UK, and converted £ to € based on currency rate 2011 (1 £ = 1.226 €) and adapted to price level 2013 (based on data for BE).

Table 18: values per visit per nature type (Sen, 2011).

habitat type	€ 2013/visit	£ 2010/visit
Mountains & heathlands	5,3	4,0
Farmlands & woods*	4,8	3,6
Marine and coastal	4,6	3,4
Freshwater and wetlands	4,1	3,1
Semi-natural grasslands	3,6	2,7

* farmlands = urban fringed farmland

We value a visit with 4,6 €/visit on average based on Sen, 2011 with a range of 3 € to 9 €/visit based on different meta-analyses for different types of nature areas (Bateman and Jones, 2003; Scarpa, 2003; Zandersen and Tol, 2009).

8.1.5 Illustration

The salt marsh area “Zwin” at the Belgian coast welcomes yearly 90000 visitors (range 80000 to 100000) (data for 2009) (Westtoer 2010). If we multiply this with a value 4.6 €/visit (range 3 to 9 €/visit) the recreational value of ‘Zwin’ is 414000 € (with a range of 240000 € to 900000 €).

8.2 Cultural heritage, identity and amenity values

8.2.1 Information needed

- Number of houses with a view on the estuary (nature areas, river... not urban (houses, buildings, concrete...)
- Houseprices in the estuary

8.2.2 Identification

Natural environments have been responsible for shaping cultural identity and values throughout human history. Ecosystems and landscapes also inspire cultural and artistic expression.

People all over the world derive aesthetic pleasure from natural environments. The perception of aesthetic qualities is, however, very subjective and does not necessarily fully match with the ecological quality and integrity of an area.

Inspiration for culture, art and design

Habitat	Qualitative importance per estuarine zone*			
	Freshwater	Oligohaline	Mesohaline	Polyhaline
Marsh habitat	4	4	4	4
Intertidal flat habitat	4	4	4	4
Intertidal steep habitat	3	3	3	3
Subtidal shallow habitat	4	4	4	4
Subtidal moderately deep habitat	4	4	4	4
Subtidal deep habitat	4	4	4	4
Adjacent land				
** Score from 1 (not important) to 5 (very important)*				

We give guidelines for quantification and valuation for the welfare gains for living close to and having a view from the home on the natural area. This is valued based on the increase of real estate prices for houses close to the natural area. We only include the impact for houses with a direct view on the area. In addition, there is an amenity value for houses on a larger distance, up to 1 km from the area. As this amenity values partly overlaps with the recreation value estimated above, we recommend not valuing this but using the (larger) recreational value instead. The recreational value is expected to be larger as more people will benefit, i.e. those living further then 1 km from the site.

8.2.3 Quantification

The amenity value is to be applied to all houses with a direct view on the natural area of the estuary. If these data are not available, the number of houses within a distance of 100 meter can be used as a proxy.

8.2.4 Valuation

The valuation is based on the impact of the natural area on the value of real estate, as estimated in hedonic studies. For houses with a direct view on a natural area, a literature review for the Netherlands indicated a range of 4 % - 15 %, with a central estimate of 9 % (Ruijgrok 2006.). For all houses within a range of 100 meter from the site, we use a value of 3.25 % (2.5 % - 6 %), based on a meta analysis of hedonic studies worldwide (Brander et al. 2011). This value is lower, as it accounts for the fact that not all these houses will have a direct view. In addition, this lower band is applicable in rural areas with lower population density, whereas the first value (9 %) is more applicable in an urban context.

These percentages can be applied to the average sales value of houses in the area where the site is located. As a reference, the average sales value for a house in Flanders, Belgium is 192000 € (Vrind 2011). To calculate a yearly value for housing, we assume a discount rate of 4 % and a time horizon of 50 years. This results in a yearly value of 9000 €/household.year (= 750 €/month) (Broekx 2013). Alternatively, one can use the average rent per year for a house in that area as a proxy.

If we use the data from Flanders, we can calculate the total value:

Value per house with a view on the site = 9 % x 9000 €/yr = 810 €/yr.
Value per house within a 100 meter distance from the site
= 3.25 % x 9000 €/yr = 293 €/yr.

For rural areas with low population density and a lot of green space available, the lower values in the range will apply. For urban areas with less green space, the higher ranges will be more relevant.

8.2.5 Illustration

For a site in Flanders with 100 houses that have a direct view of the estuary, the total amenity value can be estimated as follows:

Total amenity value = 100 houses x value per house
= 100 x (9 % x 9000 €/year)
= 81000 € year

8.3 Cognitive development (education)

8.3.1 Information needed

No information needed as no method is available in this manual for the valuation.

8.3.2 Identification

Ecosystems and landscapes are an invaluable resource for science, scientific research and education**.**

8.3.3 Quantification

Total amount of / trends in the number of visits to the sites, specifically related to educational or cultural activities. No data available

8.3.4 Valuation

The monetary value of ecological knowledge acquired through outdoor learning is estimated by examining the 'cost of investment' associated with these activities. This has to do with entry fees, travel costs and time spent on educational trips. Insufficient methodologies and data are available to value this service.

9 Integration of different ecosystem services

Although in the former chapter the ecosystem services are tackled one by one, it is important to value the ecosystem services as a bundle. Focusing on only one or a few could influence decisions in an unbalanced way. After calculation of the benefits(costs) of each relevant change in ecosystem services, you need to sum them up to have an overall picture of the total benefits (losses). There are some challenges though:

9.1 Aggregation

Most ecosystems produce multiple services and these interact in complex ways. How exactly multiple ecosystems services are interconnected is a research gap, but we will discuss potential trade-offs between ecosystem services A typical example is de-poldering, which reduces provisioning services (crops) but improves regulating services (water quality regulation, cultural services…).

In aggregating benefits it is important to avoid double counting, which is a risk where one benefit estimate potentially overlaps with another.
For example, care is needed in summing estimates of the values of different ecosystem services if some estimates potentially cover more than one service. While stated preference studies may be necessary to fully capture the non-use values of sites, a respondent’s willingness to pay may also be influenced by knowledge or perceptions of other benefits and services that the site may deliver, including for example regulating services. Careful understanding of the scope of different estimates, and the potential overlaps between them, is therefore needed before summing them. – for example pollination and value of agricultural output should not be added given that the value of the agricultural output may already integrate the pollination value. This can be done by doing a trade-off risk assessment. Herefore we refer to chapter 7 in Sanders et al. (2013)

9.2 Scaling up

In addition there are also synergetic and competing interactions possible between sites. Synergetic interactions exist for example for the service flood prevention, which could be delivered by multiple sites. Competing interactions for example exist for the recreation service. If more sites are available visits are being spread over the different sites. Keeping this in mind when scaling up to the entire estuary is very important. The numbers in this guidance only give a first estimation of the benefits of natural measures in estuaries. Quantifying the effects of measures more in detail depends on site-specific circumstances and requires tailor-made research and calculations.

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