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Authors: Hérivaux C., Le Coent P., Gnonlonfin A.

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Introduction

Nature Based Solutions (NBS) are solutions that are inspired and supported by nature (ecosystems) to address societal challenges such as climate change, food security, water security, disaster risk, human health and economic and social development (Cohen-Shacham et al., 2016). For example, NBS can involve conserving or rehabilitating natural ecosystems and/or the enhancement or creation of natural processes in modified or artificial ecosystems, and can be applied at micro- (e.g. a vegetated roof) or macro- (e.g. restoration of large flood plains) scales (WWAP, 2018).

One important characteristic of NBS is their multi-functionality, i.e. their capacity to deliver a multiplicity of benefits. Indeed, NBS developed for decreasing water-related risks (considered as the “**primary benefits**” in NAIAD) generate additional environmental, economic, and social benefits. Within NAIAD, we use the term “**co-benefits**” to refer to these secondary benefits. Assessing these co-benefits is particularly important because NBS may not be cost-effective solutions as compared to grey solutions if only primary benefits are considered, but may become cost-effective and socially acceptable if co-benefits are considered. They can therefore represent strong arguments for stakeholders to adopt, accept or finance NBS.

Assessing the co-benefits of NBS raises two main questions (Figure 1): (1) what are the co-benefits associated to the NBS (compared to grey solutions) and (2) how can these co-benefits be valued?

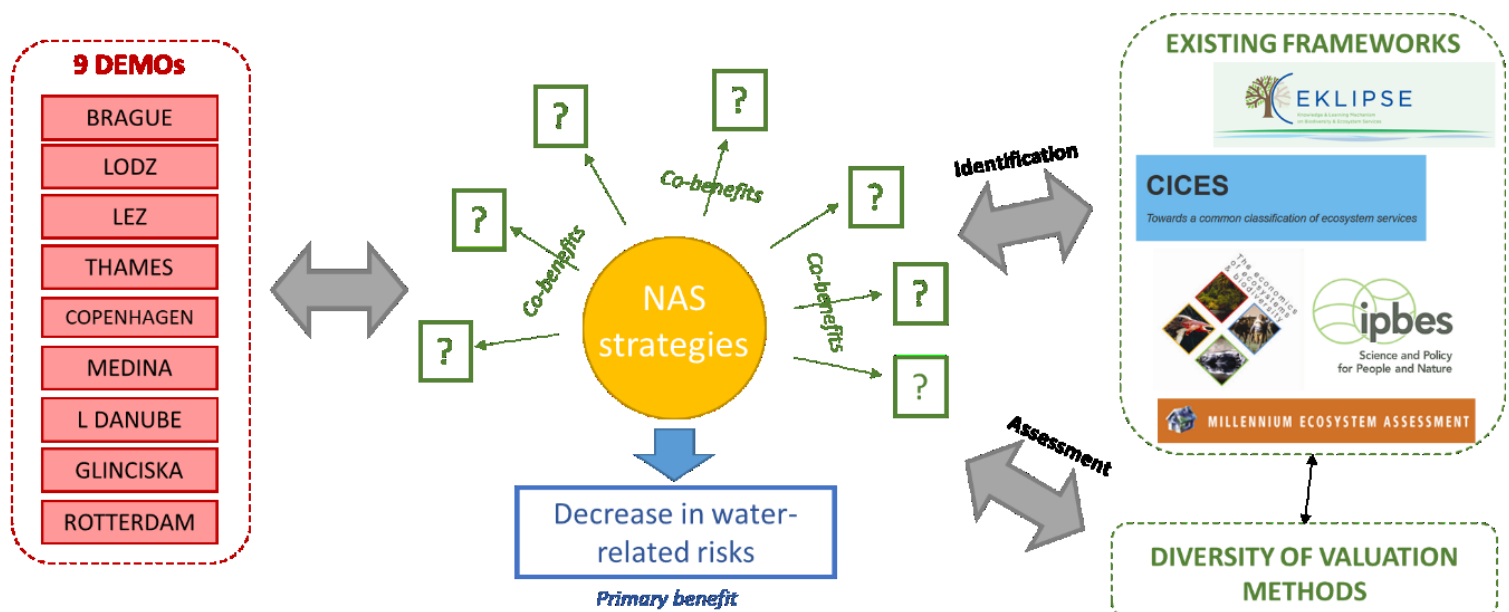


Figure 1. Assessing the co-benefits of NBS in NAIAD



There is no standardized method for identifying and valuing the co-benefits of NBS. There are many ways for assessing them, by using biophysical models, participatory methods, monetary methods... The valuation of nature or ecosystems is a highly divisive question among scholars. For example, Gómez-Baggethun and Martín-López (2015) report that there are environmental economists who accept and defend the monetary valuation of ecosystems as a pragmatic choice, and those who reject it on methodological, ethical or political grounds. The rise in ecosystem services research has resulted in further criticism against monetary valuation, highlighting incommensurability and problems of commodification (Primmer et al., 2018).

However, these last years, there is a growing number of scientists and practitioners that explore how combinations of ecological, socio-cultural and economic valuation tools can support real-life resource and land-use decision-making. In fact there is a growing body of literature which argues that valuation approaches of ecosystem services that target single value-types, be it economic, ecological or socio-cultural values fail to capture the importance of nature and to fully represent the society and its worldviews, interests and preferences. As a response, integrated valuation approaches are increasingly put forward (Gomez-Baggethun and Barton 2013; Elmqvist et al. 2015; Costanza et al. 2017; Jacobs et al., 2018). This emerging field of integrated valuation has percolated in various global science-policy interface initiatives such as IPBES¹. The recent EU OpenNESS project² also supports this plural valuation culture and provides guidance for selecting a complementary set of valuation methods in order to develop integrated valuation in practice that includes values of all stakeholders into environmental decision-making.

The NAIAD project also endorses a value pluralism perspective for co-benefits assessment. Co-benefits are assessed with the use of three main types of indicators: biophysical, socio-cultural and monetary (Figure 2).

¹ The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, see §2.1.3 for a short description

² <http://www.openness-project.eu/>



Co-benefits valuation in NAIAD

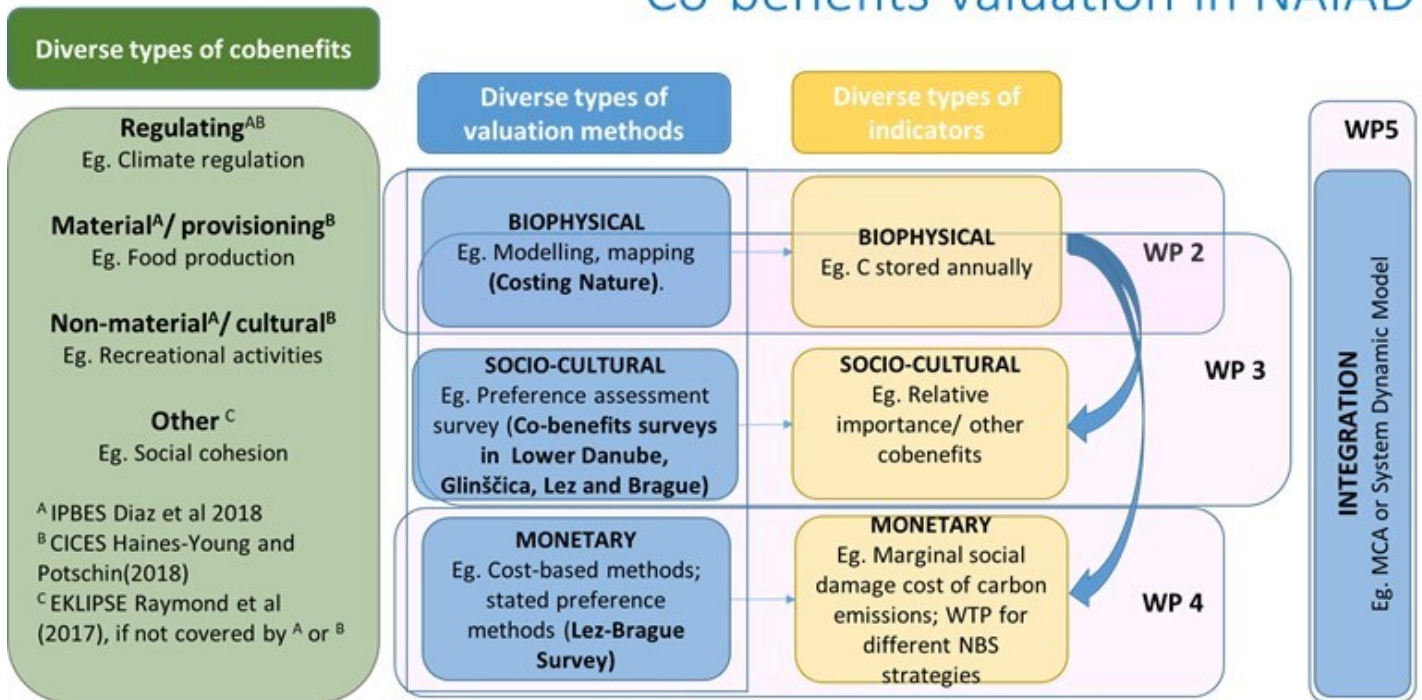


Figure 2. Co-benefits valuation in NAIAD

WP2, through Co\$ting Nature mainly provides information on biophysical values. WP3 through the risk perception assessment identifies the importance granted by stakeholders to NBS co-benefits and therefore tackles the socio-cultural values. The aim of WP4 is to provide guidance on the economic valuation of co-benefits. Finally, WP5 will provide guidance for the integration of the diversity of co-benefits values for the multi-criteria analysis of NBS.

The purpose of this deliverable is to provide guidance for the identification and the economic valuation at the DEMO scale of the main co-benefits associated with NBS developed for the objective of reducing water-related risks. **Thus, this report contributes to steps 3 and 4 of the methodological framework** for the NBS economic assessment (a). Chapter 2 presents key existing frameworks that can be used to classify the co-benefits of NBS and the participatory approaches used to identify them at the DEMO scale (contribution to step 3 "Impact assessment"). Chapter 3 provides guidelines with concrete examples for the monetary valuation of the main co-benefits associated with NBS



developed for the objective of reducing water-related risks (contribution to step 4 “economic assessment”; Figure 3b).

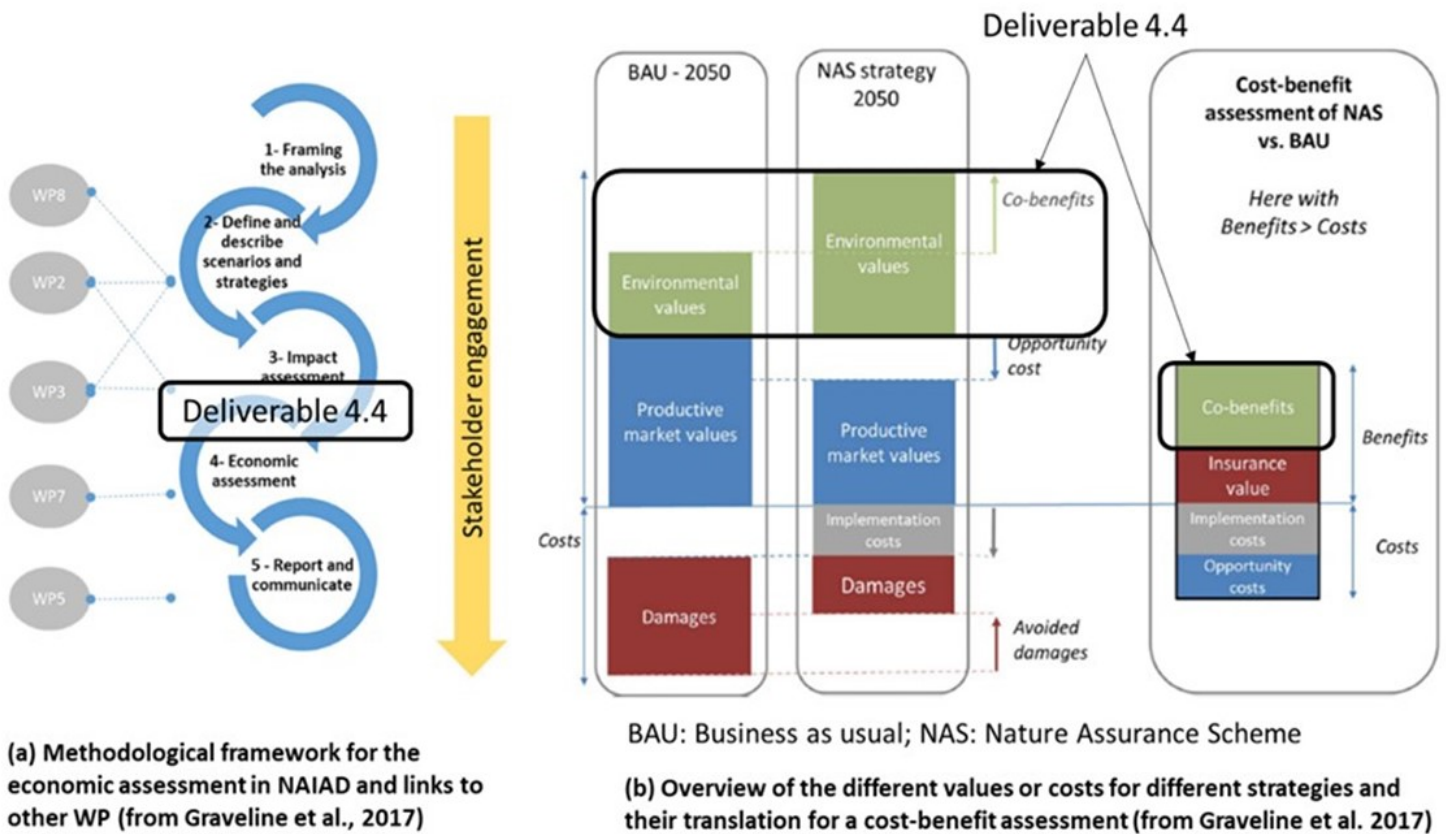


Figure 3. Contribution of the deliverable 4.4 to the general economic assessment in NAIAD

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Identification of co-benefits

This chapter first presents different frameworks that may be used as a basis for the identification of co-benefits (top-down approach, §2.1). It then presents the approaches used at the DEMO scale to identify co-benefits, based on the knowledge and the extensive fieldwork (bottom-up approach, §2.2). The last section (§2.3) then provides an overview of the main types of co-benefits identified at the local scale.

2.1. Existing frameworks for classifying co-benefits

Several classifications and frameworks can be used to classify NBS co-benefits. When focusing on ecosystem services, three main classifications are commonly used: The Millennium Ecosystem Assessment (MA, 2005), the common international classification of ecosystem services (CICES) (Haines-Young and Potschin, 2018), the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) framework (Diaz et al., 2018). Through the EKLIPSE framework, Raymond et al. (2017) recommend to move from the assessment and valuation of ES to a wider assessment of the co-benefits (and costs) of NBS by considering a range of other social, economic and environmental impacts.

2.1.1 The Millennium Ecosystem Assessment and The Economics of Ecosystems and Biodiversity (TEEB)

The Millennium Ecosystem Assessment (MA, 2005) defines the ecosystem services as the benefits people obtain from ecosystems. These include provisioning, regulating and cultural services that directly affect people and supporting services needed to maintain the other services (Figure 3).

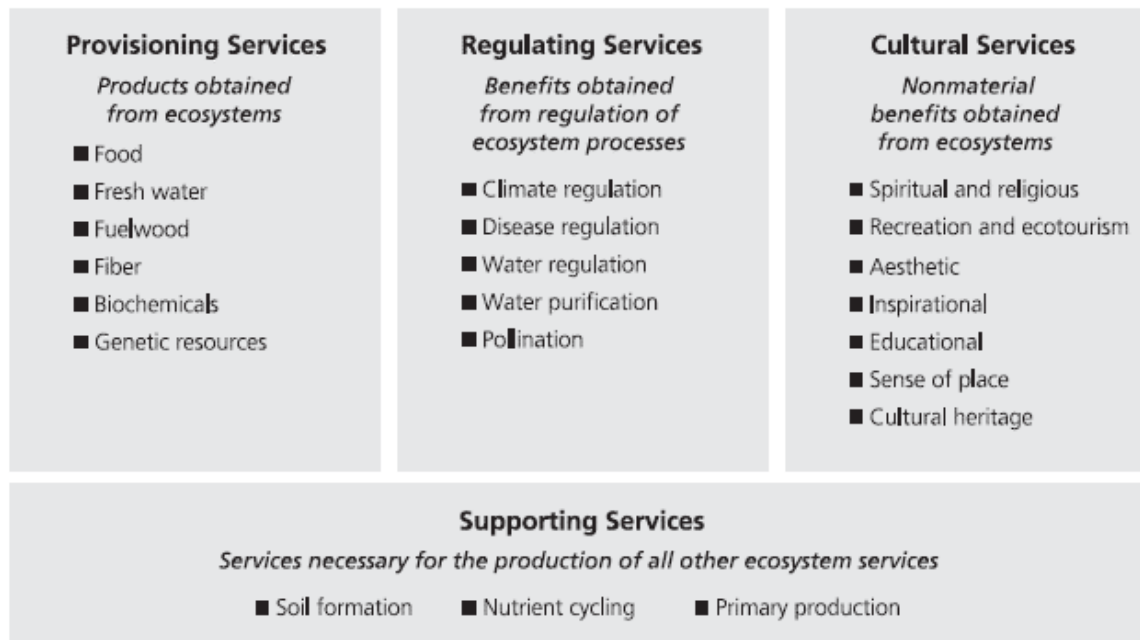


Figure 3. MA categories of ecosystem services

The Economics of Ecosystems and Biodiversity (TEEB) was set up in 2007 and led by the United Nations Environment Programme to provide a comprehensive global assessment of economic aspects of ecosystems and biodiversity. TEEB (De Groot et al., 2010) defines ecosystem services as “the direct and indirect contributions of ecosystems to human well-being.” This basically follows the MA-definition except that it makes a finer distinction between services and benefits and explicitly acknowledges that services can benefit people in multiple and indirect ways. TEEB proposes a typology of 22 ecosystem services divided in 4 main categories: provisioning, regulating, habitat and cultural & amenity services, mainly following the MA classification (Annex).

2.1.2 The Common International Classification of Ecosystem Services (CICES)

CICES (<https://cices.eu/>) builds on the existing classifications (MA, TEEB) but focuses on the ecosystem service dimension. CICES was developed from the work on environmental accounting undertaken by the European Environment Agency (EEA).

In CICES, ecosystem services are defined as the contributions that ecosystems make to human well-being, and distinct from the goods and benefits that people subsequently derive from them (Haines-Young and Potschin, 2018). CICES aims to classify the contributions that ecosystems make to human well-being that arise from living processes. The last version (v5.1 of January 2018) also addresses abiotic processes. At the highest or most general level there are three of



the four categories used in the Millennium Assessment: ‘provisioning’, ‘regulation and maintenance’, and ‘cultural’; so-called supporting services are not recognised in CICES.

The last version (v5.1) lists 90 classes of ecosystem services (details are provided in Annex).

- Provisioning (42 classes of services): “all nutritional, non-nutritional material and energy outputs from living systems as well as abiotic outputs (including water)”
- Regulation and maintenance (31 classes of services): “all the ways in which living organisms can mediate or moderate the ambient environments that affects human health, safety or comfort, together with abiotic equivalent”
- Cultural (17 classes of services): “all the non-material and normally non-rival and non-consumptive, outputs of ecosystems (biotic and abiotic) that affect physical and mental states of people”

2.1.3 The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) framework

The third framework we consider here was developed by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES), an independent intergovernmental body, established by member States in 2012, that provides policymakers with objective scientific assessments about the state of knowledge regarding the planet’s biodiversity, ecosystems and the benefits they provide to people, as well as the tools and methods to protect and sustainably use these vital natural assets.

A key element of this framework is the notion of nature’s contribution to people (NCP) [which builds on the ES concept]. NCP are all the contributions, both positive and negative, of living nature to people’s quality of life. They define 18 NCP categories (**Figure 4**), organised in three partially overlapping groups: regulating, material and nonmaterial, defined according to the type of contribution they make to people’s quality of life.

- Material contributions: substances, objects or other material elements from nature that directly sustain people’s physical existence and material assets (typically materials transformed into food, energy or ornamental purpose)
- Non material contributions: nature’s effects on subjective or psychological aspects underpinning people’s quality of life, both individually and collectively (typically recreation and inspiration; spiritual or social-cohesion experiences)
- Regulating contributions: functional and structural aspects of organisms and ecosystems that modify environmental conditions experiences by people and/or regulate the generation of material and non-material contributions



NCP differs from earlier work of ES assessment, by recognizing the central role that culture plays in defining the links between people and nature, in comparison with more traditional ES assessments generally dominated by knowledge from the natural sciences and economics (culture permeates through and across all 3 NCP groups rather than being confined to the category “cultural services”). It also emphasizes the role of local knowledge in understanding nature’s contribution to people. The NCP approach explicitly recognizes that a range of views exists.

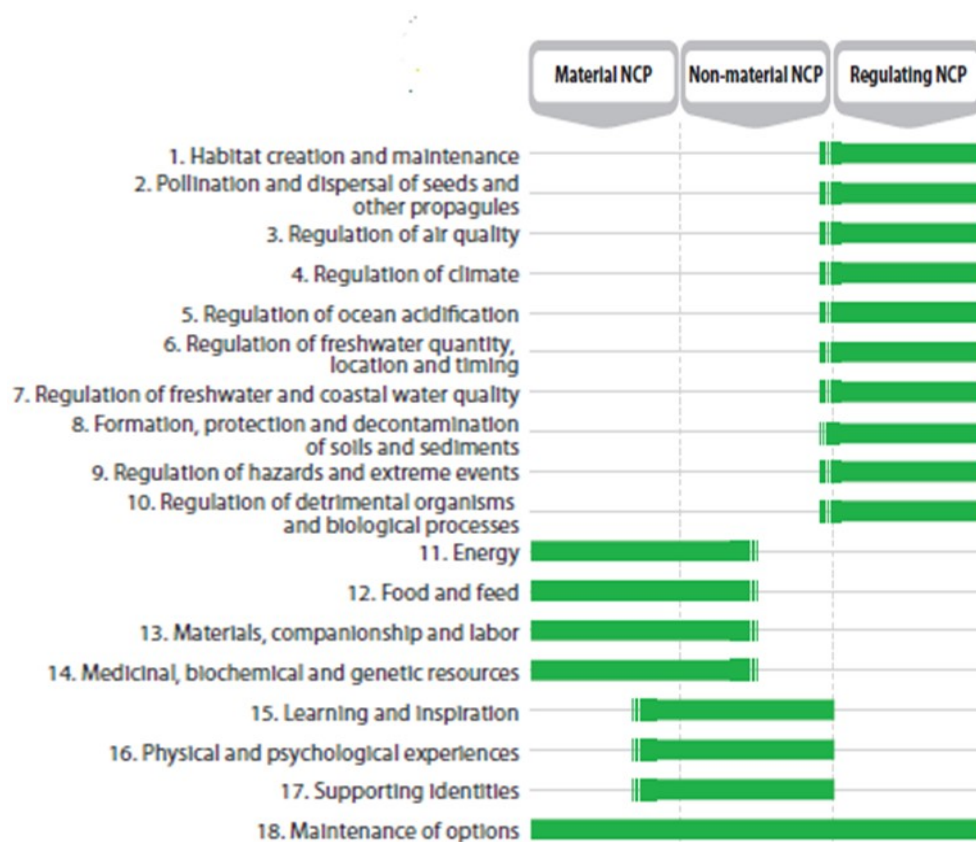


Figure 4. Nature’s contributions to people (NCP) (Diaz et al., 2018)

2.1.4 EKLIPSE framework

The EKLIPSE expert Working Group (EWG) developed a framework that enables the assessment of impacts related to specific NBS actions to promote climate resilience in urban areas (Raymond et al., 2017).



Figure 5. The 10 climate resilience challenges considered in the EKLIPSE impact assessment framework (Raymond et al., 2017)

It organizes the impact assessment within and across 10 challenge areas (Figure 5):

- 1) Climate mitigation and adaptation
- 2) Water management
- 3) Coastal resilience
- 4) Green space management
- 5) Air/ ambient quality
- 6) Urban regeneration
- 7) Participatory planning and governance
- 8) Social justice and social cohesion
- 9) Public health and well-being



10) Potential for new economic opportunities and green jobs

For each challenge area, the EWG presents a small number of representative examples of indicators that are important for assessing key impacts of specific NBS actions (details are provided in Annex), as well as a range of methods for assessing each indicator. The authors strongly encourage researchers and practitioners to move from the assessment and valuation of ES to a wider assessment of the co-benefits (and costs) of NBS by considering a range of other social, economic and environmental impacts.

2.1.5 Comparative analysis

There is not one perfect framework for NAIAD, as the concepts of “co-benefits” and “ecosystem services” are intertwined. An important part of NBS co-benefits are ecosystem services, and pertain to existing list of co-benefits, with the exception of some examples such as social cohesion.

The main advantage of the EKLIPSE framework is the fact that it encompasses both ecosystem services (e.g., air quality) and social benefits (e.g., participatory planning and governance). However, the EKLIPSE framework has two main limitations for the NAIAD project. First, it does not include the full range of ecosystem services that may be provided by NBS (for instance provisioning services such as food and materials, or some regulating services such as pest and disease control, landslide protection, control of erosion rates...) while these services may be of relevance for the DEMOs. Second, it is mostly appropriate in urban contexts, as it focuses on urban challenges. However, some DEMOs encompass also rural areas that may face additional challenges not listed in the EKLIPSE framework (e.g., maintenance of a rural fabric, food security). In the EKLIPSE framework, recreation and cultural services mostly refer to the challenge “green space management”, which seems too restrictive to include other types of non-urban ecosystems (wetlands, riverbanks, woodlands, pastures ...).

While this deliverable will mostly focus on the ecosystem services, other co-benefits may also be relevant at the DEMO scale. **For the co-benefits identification step, a combination between these frameworks was thus highly encouraged at the DEMO scale, in order to embrace a wide range of potential co-benefits and specific local issues.** §2.2 present the approaches used at the DEMO scale to identify co-benefits, based on the knowledge and the extensive fieldwork. The last section (§2.3) then provides an overview of the main types of co-benefits identified at the local scale.

2.2. Local knowledge and participatory approaches at the local scale

Different methods can be used to identify the co-benefits for the assessment of NBS in a given context. In NAIAD, different approaches were used. In some DEMOs, the most important co-benefits were identified through a more or less structured participatory process during



stakeholders' workshops. In WP3, more sophisticated methods were developed to identify the co-benefits associated with NBS. In order to understand flood risk perception, semi-structured interviews were carried out with local stakeholders in order to collect their narratives, that included a discussion on co-benefits. These interviews were confronted with the EKLIPSE framework to translate the narrative in a list of potential co-benefits. This list was then used for supporting the discussion with the stakeholders in the different DEMOs, whose main scope was to narrow down the list, to introduce different locally-based co-benefits suggested by the stakeholders and to score the importance of these co-benefits (primary benefits are also included) (Giordano et al., 2018). Results of this process in the Slovenian DEMO are presented in Table 1. This process was then used for the identification of preferred NBS through the development of a system dynamic model simulating flooding benefits and co-benefits.

Table 1. Set of benefits and co-benefits identified in the Glinščica river case study. Scoring: 3: very important; 2: important; 1: not important (Giordano et al., 2018)

Expected benefits and co-benefits	Average score
Reduce flood peak	2,74
Increase retention capacity of green areas	2,62
Enhance the control of the water speed	2,56
Reduce building and infrastructure damages	2,56
Reduce runoff	2,39
Enhance biodiversity	2,35
Increase the population protected by risk management measures	2,36
Improve the state of the green areas and ecosystem	2,45
Increase the institutional cooperation for risk management	2,33
Reduce soil erosion	2,29
Social value of urban ecosystem	2,21
Positive health effects of urban green areas	2,23
Increase cultural richness and diversity in urban area	2,30
Recreational values	2,11



Improve community's risk awareness	2,18
Groundwater quality	2,08
Increase the accessibility of green areas	1,95
Enhance air quality	2,04
Increase urban attractiveness toward business companies	1,99
Temperature reduction	2,03
Enhance community involvement in risk management	2,00
Abatement of pollutants	1,93

2.3.Synthesis: Overview of co-benefits identified at the local scale by DEMO

The combination of the existing general frameworks (§2.1) with the knowledge and extensive fieldwork carried out at the DEMO scale (§2.2) led to identify a set of co-benefits associated to NBS implementation for each DEMO.

First, WP4 partners prepared a template in March 2018 presenting the existing frameworks (§2.1), aiming at helping the nine DEMOs in the identification of the potential co-benefits. This template also identified five WP4 support persons that could help the other DEMOs to fill the template. The template suggested that each DEMO explores the relevance of using existing frameworks for co-benefits identification. The idea was to explore if these frameworks are relevant, based on the knowledge and the extensive fieldwork already carried out at the DEMO scale (and whether there are other impacts that are not included in these frameworks).

Then, WP4 organised a “cobenefits working group session” during the Montpellier meeting of July 2018. The objective was to share the advancement of co-benefits identification between DEMOs. The session was a mix of plenary and working group sessions. The co-benefits working group session was organised by small DEMO groups (2 or 3 DEMOs per group, with one WP4 support lead per group). Each working group was asked to list the cobenefits on their DEMOs and explain how they were identified (based on a specific framework, based on interviews...). The results of this session are depicted below (Figure 6 and Table 2).



Figure 6. Contribution of DEMOs partners to the co-benefit session

The list of co-benefits result from a mix of the existing frameworks. We classified them into five main categories. The four first refer to the traditional ecosystem services classification, namely regulating, cultural (or non-material), provisioning (or material) services, and supporting services which are then grouped in ecosystem services sub-categories. The fifth category "Other" refers to the co-benefits analysed by Raymond et al. (2017) which do not belong to the ecosystem services classification. These "other" co-benefits are partly grouped following the main challenge areas proposed by Raymond et al. (2017).



Table 2. Co-benefits identified at the local scale by DEMO in June 2018 (Montpellier meeting)

		Co-benefits (as quoted by DEMOs)	Brague	Lodz	Lez	Thames	Copenhagen	Medina	Lower	Głinska	Rotterdam
Regulating and maintenance	C storage and sequestration	Climate regulation (carbon storage)	•	•	•	•					
	Local climate regulation	Local temperature regulation			•						•
	Air quality regulation	Air quality	•	•	•						
	Water cycle regulation	Water quality	•	•	•	•			•		•
		Cost reduction of water treatment									•
		Water storage/ Water availability	•	•	•				•		
		Storm water reuse									•
		Support to groundwater dependent ecosystems						•			
		Increased river flow (locally)						•			
		Improved navigability conditions							•		
Non material/ Cultural	Recreation	Recreational activities/ Recreation	•	•	•	•	•				
		Eco tourism	•								
		Ornitological tourism						•			
		Rural ecological tourism							•		
	Education	Education/ learning about nature		•			•				
	Landscape	Landscape		•	•						
		Aesthetic benefits				•					
		Spatial quality betterment									•
		Spatial environment beneficial for human					•				
	Cultural heritage	Cultural heritage					•				
Material/ Provisioning	Food	Urban agriculture	•								
		Food production			•						
		Agriculture spatial management							•		



		Co-benefits (as quoted by DEMOs)	Brague	Lodz	Lez	Thames	Copenhagen	Medina	Lower	Głinska	Rotterdam
Supporting/ maintenance of habitat	Biodiversity/ habitats	Hydromorphological status	•								
		Biodiversity/ habitats maintenance	•	•	•	•	•			•	
		Improve the state of the green areas and ecosystems								•	
		Increase green									•
		Recovery of riverbank vegetation						•			
Other	Participatory planning and governance	Institutional cooperation for risk management								•	
		Water management knowledge							•		
		Increase water awareness									•
		Community risk awareness							•		
	Social justice and cohesion	Territorial cohesion	•								
		Social cohesion	•		•		•		•		•
	Public Health and well- being	Physical and mental health				•					
		Quality of life	•								
		Community well-being							•		
	Potential for economic opportunities	Economy					•				
		Local development							•		
		Business/ agriculture productivity							•		
		Redevelopment	•								
	Security	Security					•				
	Soft mobility	Soft mobility	•								

Monetary valuation of NBS co-benefits

Following Graveline et al. (2018), the assessment of co-benefits results from a comparison of strategies, namely one (or several) NBS strategy(ies) with a Business as usual (BAU) or reference strategy (Figure 3b). The monetary valuation of NBS co-benefits can start only once strategies have been clearly defined and described in terms of types of NBS implemented (Step 2 of the methodological framework, Figure 3a) and once the main types of co-benefits have been identified (Step 3 of the methodological framework).



This chapter aims at providing guidelines for the economic valuation of co-benefits associated with NBS developed for the objective of reducing water-related risks. **As described in the NAIAD description of work, it will mostly concentrate on the valuation of ecosystem services that can be improved through the implementation of NBS.** General guidelines on the monetary valuation of ecosystem services are widely diffused. For instance, Barton et al. (2017) from the OpenNESS project propose guidelines in the form of factsheets for a wide diversity of methods, including monetary methods, with key associated references, but without concrete example of application. Our aim is here to provide **concrete guidelines**, based on illustrative examples mainly taken from existing scientific literature and in some cases on the work carried out in NAIAD demos. These guidelines will however not cover methodologies associated with the primary benefits of NBS such as damage reduction for water related risk, which are covered in D4.3. In addition, as noted in chapter 2, co-benefits associated to NBS are not limited to ecosystem services (e.g., social justice and social cohesion). Complementary approaches required to integrate these other categories of co-benefits into the NBS assessment are beyond the scope of this deliverable.

3.1. Overview of economic values of ecosystem services and main valuation methods

In Environmental economics, the economic value of ecosystem services, usually referred as the Total Economic Value (TEV), is divided in different categories of values. Pearson et al. (2006) describe this typology. The TEV is first divided in use and non-use values. Use values relate to actual use of the good or service in question (e.g. the recreational use of a multifunctional recreational basin or the reduction of energy consumption linked to the installation of green roofs), planned use (a visit planned in the future) or possible use. Actual and planned uses are fairly obvious concepts, but possible use could also be important since people may be willing to pay to maintain a service in existence in order to preserve the option of using it in the future, i.e an option value. Non-use value refers to the value people grant for maintaining some good or service in existence even though there is no actual, planned or possible use. One typical non-use value is the value people grant for keeping a good or service in existence in a context where the individual expressing the value has no actual or planned use for his/herself or for anyone else (for example people may value the restoration of a wetland because they consider that species present in these ecosystems should be conserved even though there is no avoided cost or direct or indirect benefit they derive from the existence of these species).

There is not one single monetary valuation method. Many techniques have been developed to measure monetary values of ecosystem services (**Figure 7**).

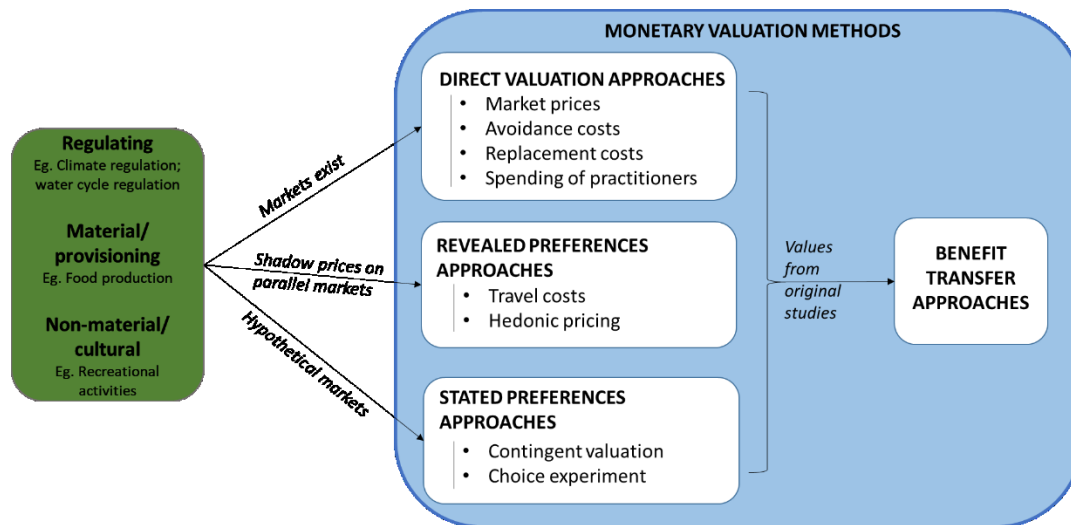


Figure 7. Potential approaches for valuing co-benefits in monetary terms

We distinguish here four main group of approaches:

Direct valuation approaches (market price and cost-based methods). When markets exist, monetary values can be derived directly from market prices and costs. In general, these approaches require biophysical indicators (e.g., annual carbon sequestration expressed in t-eqCO₂/year; water availability expressed in m³/year; nitrate concentration in water expressed in mg/L). They are generally easy to implement and easy to grasp for non-economists. Both French³ and Spanish⁴ National Ecosystem services assessments used direct valuation approaches for the economic valuation of ecosystem services. However, these methods only capture direct use values: they do not reflect the whole economic value. Result should thus be considered as lower bound estimates. These methods and their application to NBS are presented in details in section 3.2.

Revealed preference approaches. Monetary values may be estimated from shadow prices in parallel markets e.g., such as housing prices or travel costs. For example, the value people grant to an urban park can be derived from the statistical estimation of the increase of housing prices in the surrounding of this park. Revealed preference methods are used widely to estimate the

³ <https://www.ecologique-solidaire.gouv.fr/levaluation-francaise-des-ecosystemes-et-des-services-ecosystemiques>

⁴ <http://www.ecomilenio.es/ecosystems-and-biodiversity-for-human-wellbeing-snea-synthesis-of-key-findings-download/3661>



monetary value of cultural ES like recreation and ecotourism. These methods will not be detailed here due to their high level of econometric complexity.

Stated preference approaches. When no parallel markets exist, monetary valuation typically relies on expected consumer behaviour in hypothetical markets. Stated preferences are obtained through representative surveys of the population in which people are asked to provide their willingness to pay (how much they would contribute in terms of fee or tax increment) for a hypothetical modification of the environment. This type of method could easily be implemented to reveal residents' preferences for different NBS scenarios. An example of implementation of this methodology in NAIAD is presented in section 3.3. Such approaches can be used to value one single ecosystem service or a bundle of services. Non-use values can only be evaluated using these approaches since for these values it is difficult or impossible to identify some behavioural change which affects a price or quantity which can be observed.

Benefit transfer. Values from original studies are sometimes applied to other sites through benefit transfer. The NATURVATION project (<https://naturvation.eu/>) database⁵ provides open access to knowledge about where, when, and under which circumstances, integrated multi-functional and systemic impacts of nature-based solutions can be assessed. This database was created from a review of assessment methods (i.e. frameworks, models and tools) currently used to assess the ecological, economic and social benefits of urban NBS. Bockarjova and Botzen (2017) provides a description of a benefit transfer assessment tool, and a meta-analytic methodology for the derivation of economic value transfer functions for urban NBS based on this database.

This deliverable focuses on two main types of methods: direct valuation and stated preference approaches. Sections 3.2 and 3.3 provide a description of these methods, with illustrative examples that may be of interest for the NAIAD project.

3.2. Direct valuation approaches (market price and cost-based methods)

This section details in concrete terms how direct valuation approaches may be used to assess specific ecosystem services delivered by NBS, based on a literature review, with illustrative examples. When direct valuation approaches are used, the economic valuation typically relies on two main steps (Figure 8):

⁵ <https://naturvation.eu/result/value-and-benefit-assessment-methods-database-urban-nature-based-solutions>



(a) the quantification of the level of ecosystem services provided by each strategy in non-monetary terms. In the case of regulating services, it consists for instance in assessing the service level with biophysical indicators (e.g., annual carbon sequestration expressed in t-eqCO₂/year; water availability expressed in m³/year; nitrate concentration in water expressed in mg/L) derived from models, functions or reference values obtained in similar contexts. The level of service generally depends on the area occupied by each type of NBS (which should therefore be clearly described for each strategy). The comparison of the NBS strategy with the reference strategy provides an indicator of the change in the level of service provided by the implementation of NBS.

(b) the monetary valuation of the change in the service level obtained in (a).

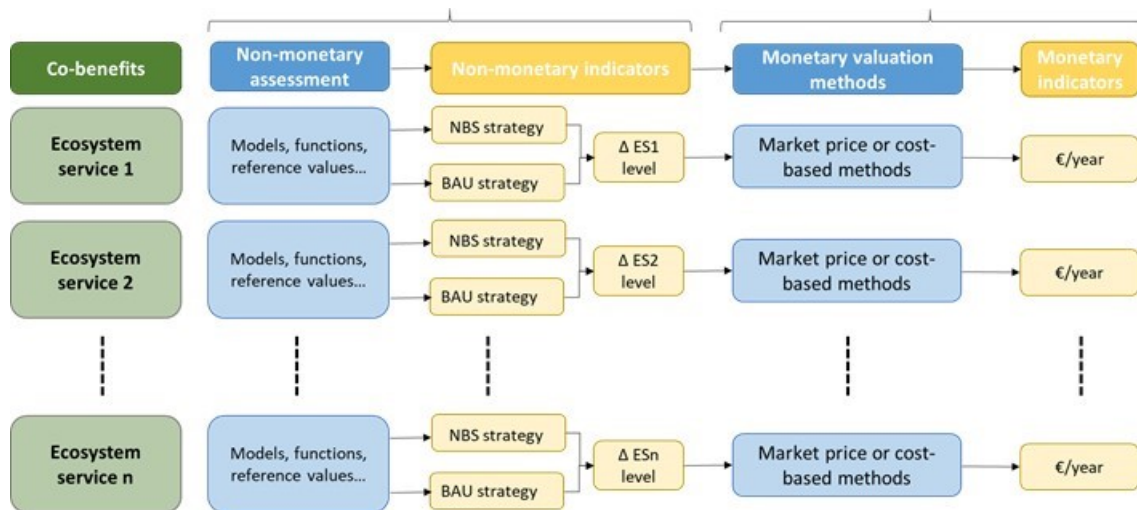


Figure 8. Stepwise approach for ES valuation when direct valuation approaches are used

Each ecosystem service generally requires specific valuation methods both for biophysical and monetary assessment. We selected five ecosystem services (corresponding to the first services listed in **Table 2**): carbon storage and sequestration (3.2.1); local climate regulation (3.2.2); air quality regulation (3.2.3); water cycle regulation (3.2.4); and recreation (3.2.5). Each sub-section is structured in three main parts: after a short description of the service, it describes how the ES can be assessed in biophysical terms and then valued in monetary terms. Some concrete examples of valuation are provided, and potential linkages with Costing Nature developed as part of WP2 are highlighted.



3.2.1 Carbon storage and sequestration

Description

Forests and grasslands that are essential parts of many NBS contribute significantly to climate regulation (IPCC, 2014) through carbon storage and regulation. Urban green spaces such as green roofs and gardens have also a significant potential as carbon reservoir, although their contribution is generally relatively small and undervalued in national assessments (Derkzen et al., 2015; Johnson and Geisendorf, 2019). Ecosystems conserved, restored or created through NBS contribute to global climate regulation through two processes: annual carbon sequestration and long-term carbon storage.

- Carbon sequestration: ecosystems absorb carbon-containing atmospheric compounds during photosynthesis. The absorbed carbon is stored within soil (in organic form) and biomass (both above-ground such as in leaves or branches, and below-ground such as in roots). Ecosystems also release carbon dioxide through respiration, combustion and decomposition. The difference between the amounts of carbon absorbed – which depend upon vegetation growth – and released – which depend upon harvests and tree mortality – is the annual carbon sequestration.
- Carbon storage: the total amount of carbon sequestered over the ecosystem lifetime is the carbon storage. Carbon storage is the mass of carbon already captured by vegetation and soils. Indeed, some of the carbon absorbed by vegetation and soil is integrated into organic matter and is not directly released into the atmosphere. It is stored in the above- and underground compartments of the ecosystem on the long run. Although storage is not permanent, it helps to delay the greenhouse effect and regulate temperatures. In this sense, it should be taken into account in the evaluation of the global climate regulation service.

The economic valuation of the carbon storage and sequestration service typically relies on two main steps (Figure 9): (1) the biophysical quantification of the annual carbon sequestration and the long term carbon storage, and (2) the monetary valuation. Indicators may be expressed either in terms of carbon (kgC, tC) or carbon dioxide (t-eqCO₂, kg-eqCO₂). A tonne of CO₂ contains 0.2727 tonnes of carbon.

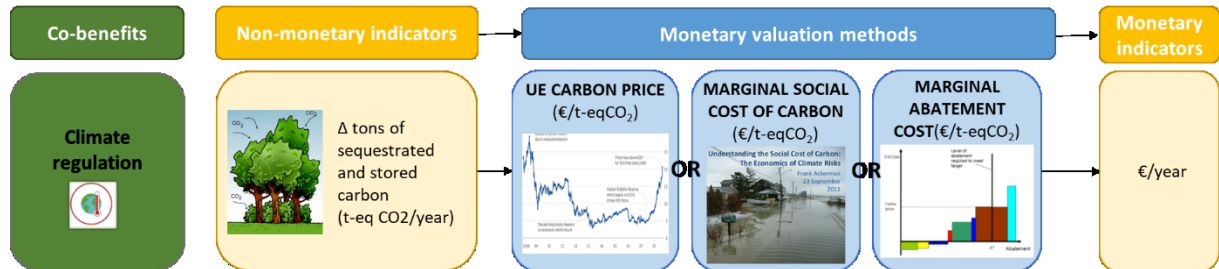


Figure 9. Overview of the economic valuation of the carbon storage and sequestration service

Biophysical assessment

Carbon storage can be estimated as the sum of carbon stored in vegetation and in soil. In quantifying carbon storage in vegetation, two factors are important (Derkzen et al., 2015): (1) biomass volume, which is proportional to the carbon storage capacity of trees; (2) vegetation type: almost all aboveground carbon storage takes place in trees and only a few percent is stored in shrubs and herbaceous vegetation (Nowak *et al.*, 2013). Carbon storage is commonly quantified by using information from field sampling in biomass and growth equations. Some studies perform a Life Cycle Analysis to assess a green area's carbon balance, while others produce storage estimates per unit of land use (e.g. residential, institutional) or vegetation cover (e.g. shrub, woodland, trees). Indicators are generally expressed in kg C/m². There are also some tools available to calculate the quantities of CO₂ fixed and stored by trees, such as i-Tree-Eco⁶, the Invest Carbon storage and sequestration model⁷ or Costing Nature⁸.

Net carbon sequestration rates can be estimated as a function of tree conditions, sizes and estimated tree mortality. Indicators are generally expressed in kg C/m²/year.

Estimates

If field sampling is not possible, carbon storage and sequestration rates can be derived from existing studies in similar contexts. For instance, Derkzen et al. (2015) derived carbon storage rates (kg C/m²) for different types of urban green spaces from studies presenting estimates for cities comparable to Rotterdam (Table 3). Other estimates provide reference rates for trees: e.g., 7.57 kg C/m² in Germany (Strohbach and Haase, 2012), 7.69 kg C/m² in US urban areas

⁶ www.itreetools.org

⁷ <http://data.naturalcapitalproject.org/nightly-build/invest-users-guide/html/carbonstorage.html#summary>

⁸ <http://www.policysupport.org/costingnature>



(Nowak et al., 2013). There is not much literature about carbon stored in grass or shrubs (Aevermann and Schmude, 2015). Davies et al. (2011) calculate 0.15 kg C/m² for public herbaceous vegetation with large and infrequently mowed grass areas in an urban environment.

Table 3. Overview of Carbon storage indicators specified per urban green space (UGS) type for cities comparable to Rotterdam (Derkzen et al., 2015)

UGS type	Description	Carbon storage (kg C/m ²)
Tree	Individual tree, mostly street trees	10.64
Woodland	Clustered treed, urban forest	15.62
Tall shrub	Shrub of hedge sized 2-5m	7.79
Short shrub	Shrub of hedge sized <2m	5.61
Herbaceous	Low vegetation consisting of non-woody plants, mostly grass and herbs	0.17
Garden	Domestic garden consisting of a mix of vegetation, water and sealed surface	1.07

Soil carbon storage can be derived from Pouyat et al. (2002) who estimate that urban soils can store 8.2 kg C/m², although variations in land use and land cover distributions can lead to variation in carbon density of soil (Pouyat et al., 2006). In comparison, Arrouays et al (2002) estimate carbon storage in French forest soils at 6.9 kg C/m².

The average net annual sequestration rate for urban trees is estimated at 0.21 kg C/m²/year in the United States (Nowak et al., 2013). Lower sequestration rates have been cited in the literature for green roofs with 0.17 kg C/m²/year (Getter et al., 2009). Derkzen et al. (2015) do not consider the annual sequestration in their valuation of urban green spaces, considering that typical Urban Green Space such as pruned trees, lawns, and flower beds generally do not sequester much CO₂ and their maintenance can even emit sizeable amounts of CO₂ and N₂O through fertilization practices.

Monetary assessment

Economic valuation here consists in applying a value to the quantities of CO₂ stored and sequestered by ecosystems. Monetary values are generally expressed in €/t-eqCO₂. When expressed in tC, biophysical indicators should thus be converted for the economic valuation, by considering that a tonne of CO₂ contains 0.2727 tonnes of carbon.

$$VC_{sequestration} = C_{sequestration} \times C_{value}$$

$$VC_{storage} = C_{storage} \times C_{value} \times RR$$

To account for carbon value changes in time:



$$VC_{storage} = C_{storage} \times \sum_{t=1}^n \frac{Carbon\ value\ (t) \times RR}{n \times (1+a)^t}$$

With $VC_{sequestration}$ the economic value of carbon sequestration (€/year); $VC_{storage}$ the economic value of long term carbon storage (€/year); $C_{sequestration}$ the annual sequestered carbon (t-eqCO₂/year) ; $C_{storage}$ the total long term carbon storage (t-eqCO₂); $C_{value}(t)$ the carbon value as a function of time (€/t-eqCO₂); RR the rate of return of capital (%); a the discount rate (%)

Several approaches can be used to assess the economic value of carbon sequestration and long-term carbon storage: the market price method, the social cost of carbon method, and the marginal abatement cost method.

The **market price method** values each ton of sequestered and stored carbon at the average spot prices of CO₂ Emission Allowances on the EU Emission Trading System, i.e. 26€/t-eqCO₂ in May 2019 (

Figure 10). This market price is supposed to reflect the price of climate constraint by imposing a scarcity on CO₂ emissions through emission permits. But these prices are incomplete, as the markets currently only cover a fraction of the emissions, produced mainly by the energy sector and some industrial activities (less than 50% of total emissions in Europe) and the prices that are emerging mainly indicate that the allocation of allowances has been very generous (CGDD, 2015).

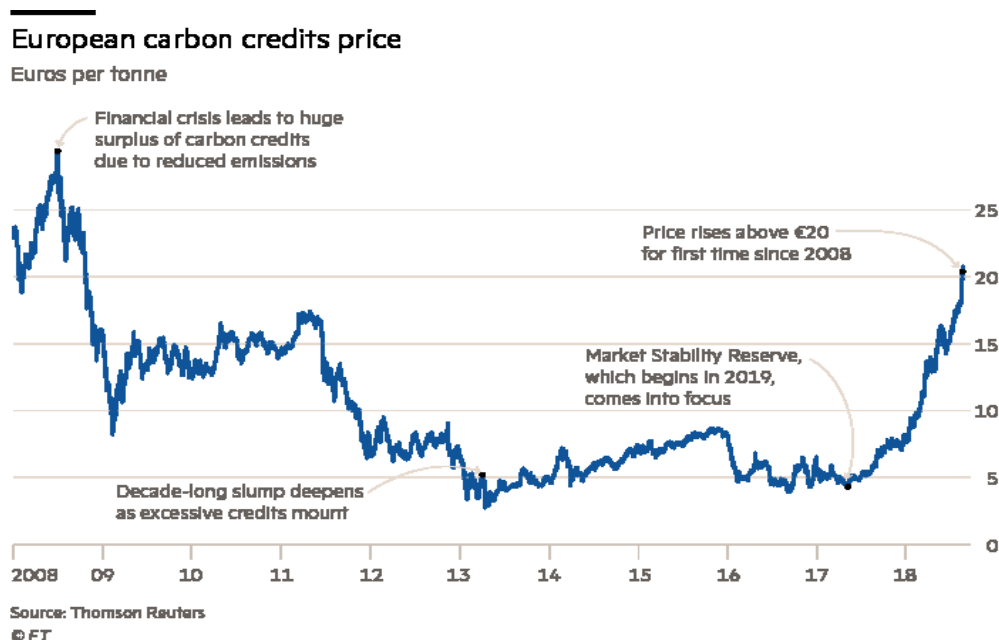




Figure 10. Evolution of the spot prices of CO₂ Emission Allowances on the EU Emission Trading System

The **social cost of carbon (SCC) method** values carbon sequestration and storage at the marginal global damage costs of carbon emissions, that is the net present value of climate change long-term impacts of one additional ton of carbon emitted today (Watkiss and Hope 2011). Unfortunately, there is no globally agreed value for SCC (Binner, 2017) as it is itself an active area of research (see Tol, 2013; Tol, 2011; Greenstone et al., 2013; Nordhaus, 2014). In Germany, Johnson and Geisendorf (2019) use for example the value of 80 € per ton of CO₂ equivalent (€/t-eqCO₂) as recommended by the Federal Environment Agency for the social cost of carbon. In France, the recent Energy transition law proposed an increasing trajectory of the carbon tax to reflect this SCC: from 56 €/t-eqCO₂ in 2020 to 100 €/t-eqCO₂ in 2030 (Meem, 2016).

The **marginal abatement cost (MAC)** method is an alternative approach that reflects the cost to polluters of reducing emissions by an incremental amount in order to reach a specific target on carbon emissions imposed by a regulatory authority. This is illustrated, in simplified form, below in Figure 11, which shows how a target-consistent carbon value would be set. Taking into account emissions projections and abatement options, we can determine the effort level, A*, that is required in order to meet the target. Reading across from the abatement curve produces the carbon valuation level that is required in order to be target-consistent (DECC, 2009). The MAC is generally valued at the national scale, to reflect the national policy target. For instance, the UK Government publishes a range of carbon values based on the abatement costs of meeting target emissions reductions for use in UK policy evaluation (DECC, 2009). The UK MAC for non-traded sectors is expected to increase from £57 to £78/t-eqCO₂ between 2010 and 2030, to peak in 2077 at a value of £341/t-eqCO₂ and fall thereafter⁹. In France, investment choices in infrastructures should include in their socio-economic analysis the shadow value of carbon, estimated by the Quinet commission to 32€/teqCO₂ in 2010, increasing at a 5.8% annual rate so as to reach 100 €/t-eqCO₂ in 2030 (Quinet 2008). It corresponds to the estimated price of carbon that should be put in place to achieve the national objective to reduce greenhouse gases emissions by a factor of 4 by 2050 (Meem, 2016).

⁹ From 73 to 100 €/t-eqCO₂ (269 to 368 €/tC) between 2010 and 2030, to peak in 2077 at a value of 439 €/t-eqCO₂ (1609 €/tC).

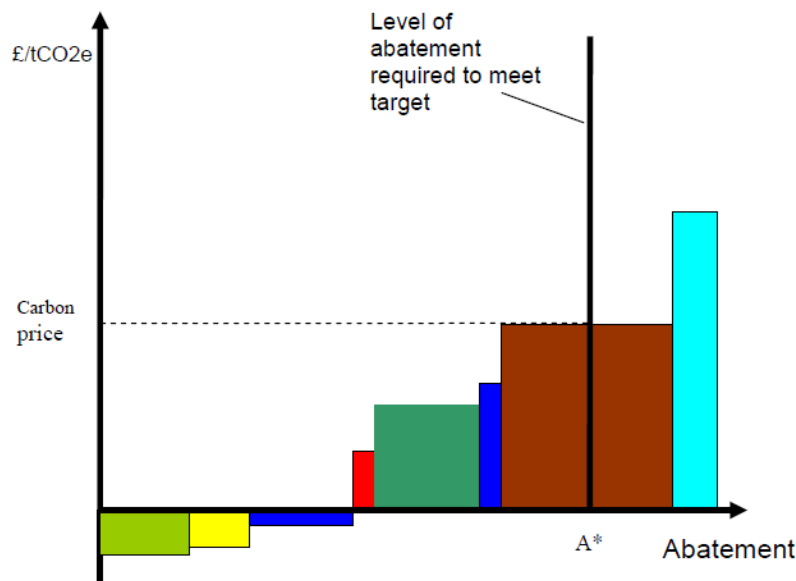


Figure 11. Example of an abatement cost curve (DECC, 2009)

Box 1. Example of the valuation of carbon storage and sequestration by London's urban forests: London i-Tree Eco project

London i-Tree Eco project aimed to improve the understanding of London's urban forests benefits. Rogers et al. (2015) presents the results of this assessment and provides a quantitative baseline of the air pollution, carbon storage and sequestration benefits of trees as well as the amenity and stormwater benefits they provide. The project was run by a partnership of the UK Forestry Commission, Greater London Authority, Treeconomics, Greenspace information for Greater London, London Tree Officers Association, Natural England, Trees for Cities, the Tree Council and Forest Research.

The project uses the i-Tree Eco tool (www.itreetools.org), designed to use standardised field data from randomly located plots across the whole study area. A total of 724 plots (0.04 ha/plot) were randomly selected from both Inner and Outer London.

An estimated 2,367,000 tonnes (approximately 15t/ha) of carbon is stored in London's trees with an estimated value of £147 million (based on the marginal abatement cost method). Of this total 1,868,000 tonnes is stored in Outer London and 499,000 tonnes in Inner London.

The gross sequestration of London's trees is about 77,198 tonnes of carbon per year. Allowing for dead and dying trees the net sequestration is estimated at 65,534 tonnes of carbon per year (approximately 2.4 t/year/ha). The value of this sequestered carbon is estimated at 3.9 million pounds per year, by using the marginal abatement cost method.



Table 4. Assessment of carbon stored and sequestered by London's urban forests

Area	Carbon Stored (tonnes)	Value	Carbon Sequestration 2014(tonnes)	Value
Inner London	499,000	£29.9 million	15900	£955,000
Outer London	1,868,000	£112 million	61300	£3.68 million
Greater London	2,367,000	£142 million	77200	£4.63 million

Box 2. Example of valuation of the carbon storage and sequestration service provided by the protection of strategic groundwater resources for the future.

Hérivaux and Grémont (2019) investigate if ecosystem services valuation improves the understanding of local stakeholders regarding the benefits of groundwater preservation. They develop and test a systemic approach based on local data collection, cost-based monetary valuation methods and stakeholder involvement. The study site is a strategic groundwater area (7400 ha) located in Southern France that could be of interest in the future to supply the growing population of coastal urban areas. They characterize, quantify and value in monetary terms nine ecosystem services, including the carbon storage and sequestration service.

On the study area, most of the carbon is sequestered and stored in soils and biomass by forests (83 % of the area). The annual carbon sequestration of forests was estimated using two indicators of the net annual growth rate of trees provided by the National Forest Office (local average) and the National Institute of Geographic and Forest Information (regional average). Net growth of trees was then combined with carbon sequestration coefficients per tree species in France (Loustau, 2004). For grasslands, a similar approach was adopted based on national carbon sequestration coefficients (Puydarrieux and Devaux, 2013).

The amount of carbon that is stored over the long run varies according to the relative share of carbon that is stored above and below ground. For forests, the volume of standing timber was estimated based on a regional average. Two hypotheses were then formulated in line with recent literature. First, we considered that 2/3 of the carbon stored was stored below ground (in biomass and soils) against only 1/3 stored above ground (Dupouey et al. 2002). Second, we considered that respectively 3/4 and 1/4 of below and above ground carbon were stored over the long term (Chevassus-au-Louis, 2009). For grasslands, carbon stored in above-ground biomass was considered negligible (Puydarrieux and Devaux, 2013). Long-term carbon storage by grasslands was therefore estimated using coefficients representing the average amount of



carbon stored in soils by south eastern grasslands of France over the long-term (Arrouays et al., 2002).

Finally, two valuation methods were used to assess the economic value of carbon sequestration and long-term carbon storage: the market price method, considering the 2010 average spot prices of CO₂ Emission Allowances on the EU Emission Trading System (13€/t-eqCO₂); and the marginal abatement cost method, considering the shadow value of carbon estimated in France by the Quinet commission to 32€/t-eqCO₂ in 2010, increasing at a 5.8% annual rate so as to reach 100 €/t-eqCO₂ in 2030 (Quinet, 2008). For long-term carbon storage, a discount rate of 2.5% was used over a 30 year time period (Quinet, 2013).

The economic benefits resulting from carbon storage and sequestration on the groundwater strategic area are estimated to range from 0.27 to 1.04 million €/year, that is between 37 and 140 €/ha/year (Table 5).

Table 5. Economic valuation of the carbon storage and sequestration service

	Market price		Marginal abatement cost	
Carbon value (€/t-eqCO ₂)	13		32-100	
Annual carbon sequestration (t-eqCO ₂ /year)	6 400	8 955	6 400	8 955
Value carbon sequestration (€/year)	83 198	116 419	204 795	286 570
Long term carbon storage (t-eqCO ₂)	588 588	608 136	588 588	608 136
Rate of return of capital (%)	2,50%			
Value carbon storage (€/year)	191 291	197 644	727 106	751 255
Total (€/year)	274 489	314 064	931 901	1 037 825
Total (€/ha/year)	37	42	126	140



Box 3. Costing Nature and the carbon storage and sequestration service

Costing Nature (Mulligan, 2018) proposes a biophysical assessment of the carbon storage and sequestration service, with the relative potential and realised carbon services index.

Relative potential and realised carbon services index (RRCS)

Carbon storage and sequestration are distinct regulation services that benefit all humanity and may also provide opportunities for carbon finance that benefit national or local beneficiaries. Carbon stored in vegetation and soil is locked out of the atmospheric system for a period and this contributes to their being less carbon in the atmosphere. Carbon storage is not an active service of the ecosystem but rather a consequence of carbon sequestration and of the ecosystem remaining intact so that carbon accumulates (so a passive service). Carbon sequestration is determined by photosynthesis and accrues continuously so the magnitude of the service increases over time, contributing to carbon stocks in the process. The total carbon service is thus the combination of carbon stocks and carbon sequestration and the balance between the two clearly depends on the time period over which sequestration is calculated. Carbon sequestration is calculated here from the dry matter productivity (DMP) analysis of Mulligan (2009b) in which SPOT-VGT DMP1 calculated every 10 days at 1km resolution on the basis of change in NDVI, was averaged over the period 1998-2008, globally. DMP (t biomass/ha/yr) is multiplied by 0.42 (Ho, 1976) to convert to units of tC/ha/yr. Above-ground carbon stock is calculated from Saatchi et al (2011) for the areas in which data are available and Ruesch and Gibbs (2004) elsewhere. This is combined with soil carbon calculated from the map of Scharlemann et al. (2009) to produce total above- and below-ground carbon stocks. The potential (C_p) and also realised (C_r) carbon service is thus the mean of normalised carbon sequestration (C_s), normalised above-ground carbon stocks (C_a) and normalised below-ground carbon stocks (C_b) (eq 1 and 2). The potential and realised services are the same because all Carbon sequestered and stored benefits the global community irrespective of their location relative to the site of service provision (i.e. beneficiaries do not need to be nearby or downstream)

$$C_p = ((C_s + C_b + C_a) / 3)$$

$$C_r = C_p$$

Costing Nature also offers the possibility to users to integrate carbon market price to value carbon sequestration provided by forests.

Carbon: tonnes per year above ground storage+sequestration for forests only

This metric maps directly on to the economic value: *per-tonne carbon market price* for sequestered carbon only, recognising that stored carbon also produces benefits but is a stock of previously accrued service, not a service in itself. The economic value is calculated simply from the biophysical value as the product of the per pixel carbon sequestration (tonnes) and the per-tonne price. Carbon in protected areas may be excluded and a maximum attainable value may be set, in which case the value is ratioed to this value and thus still distributed according to sequestration rates but with a reduced per tonne value. This provides a better representation of true economic value than concentrating the total study area value in a small area of carbon to be valued and assigning no value to the same carbon elsewhere.



3.2.2 Local climate regulation

Description

Climate change is expected to have a strong impact in cities. Even under a 1.5°C increase of temperature, twice as many megacities could become heat stressed, especially through the phenomenon of Heat Islands (HI), exposing more than 350 million more people to deadly heat by 2050 (Matthews *et al.*, 2017; IPCC, 2018). As a reference, in France the heat wave in 2003 caused additional 14800 deaths with 60% increase of mortality rate for the French territory that reached a 134% mortality increase for the city of Paris (CGDD, 2018). Besides these dramatic effects, high temperature have a strong impact on the well-being of urban dwellers.

On the other hand, cities are large source of carbon emissions especially related to energy consumption for heating and air conditioning. These emissions are largely related to local climate and their variation. Indeed, the increased occurrence of heat waves may lead to an increase of the use of air-conditioning which will in turn increase temperature in cities (De Munck *et al.*, 2013), carbon emissions and worsen climate change. On the contrary, the presence of urban parks, street trees, included in urban NBS improves local climate and may reduce energy use.

NBS may therefore play a significant role on the regulation of local climate in urban ecosystems and in turn on energy consumption. Vegetation indeed intervenes through different mechanisms. Firstly, vegetation absorbs and reflects a part of solar radiation. Secondly, trees create shadow and prevent the population from being directly exposed to solar radiation and prevent pavement and building from accumulating heat and reemitting afterwards. Thirdly, evapotranspiration through the transformation of water in vapour absorbs energy and refreshes the atmosphere. Finally, the presence of trees can also increase air rugosity and limit the entrance of hot air in summer and cold air in winter. Nevertheless, trees can have a negative impact in winter by limiting solar radiation and can sometimes increase wind speed for tree alleys. The increase of humidity generated by evapotranspiration may also limit temperature reduction during the night (CGDD, 2018)

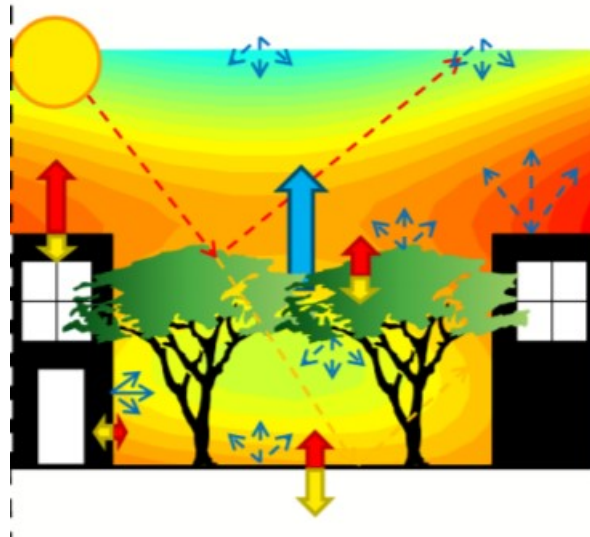


Figure 12. Effects of trees for climate regulation (Chancibault *et al.*, 2017)

Vegetation introduced with the development of NBS in cities may therefore have a significant impact on climate regulation. Some specific NBS such as green roofs, may have additional local effects in buildings with a significant impact on the reduction of heat in buildings in summer and an increase of temperature in winter (especially for upper floors) but limited effect at street level (Alcazar *et al.*, 2016).

The overall approach to estimate the production of air quality co-benefit is the following:

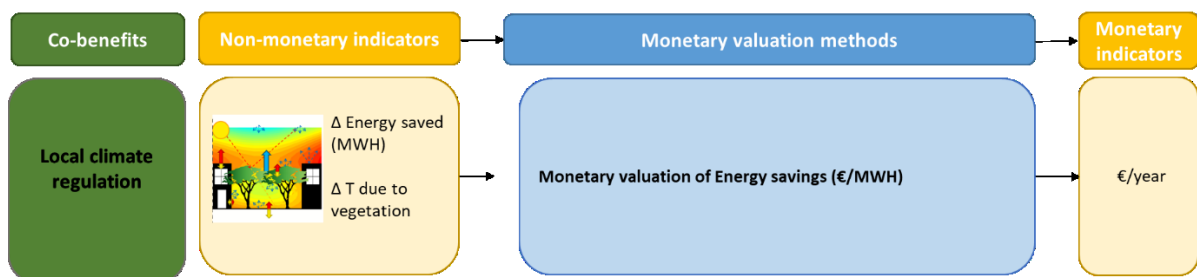


Figure 13. Overview of the economic valuation of the local climate regulation service

Biophysical assessment

At the city level, identifying a simple indicator to estimate the climate regulation effects generated by NBS is complex. These effects are local and strongly depend on the spatial distribution of vegetation throughout the city, on the scale considered and on the type of NBS



and vegetation associated. Some effects may be at the building level (green roofs), at the street level (street trees) or at the city level (urban park).

As in the case of carbon sequestration and air quality improvement, the i-Tree Eco model can provide estimation of the effects of urban trees on energy consumption. I tree-Eco models tree position, orientation and distance relative to buildings to determine the impact of the urban forest on the energy use by buildings, namely on heating and air-conditioning. This model is designed for US climate types, building types and efficiency characteristics, heating fuel types and mixes, energy production methods and emission factors. Although the applicability to other contexts has its limitations, I-tree Eco has been applied to the city of London in the London I-Tree project (Rogers *et al.*, 2015). The analysis thus provides an indication of the likely impact of urban trees across London on energy use by buildings. Results in terms of energy impact are presented in **Table 6**.

Table 6. Annual energy effects of trees near building (MBTU: Million British Thermal Units, MWH: Mega Watt Hour) (Rogers *et al.*, 2015)

		Heating	Cooling	Total
Inner London	MBTU	-47,221	-	-47,221
	MWH	-2,613	8,560	5,947
	Carbon avoided (mt)	-806	1,186	380
Outer London	MBTU	-113,561		-113,561
	MWH	-6,715	17,671	10,956
	Carbon avoided (mt)	-1,949	2,451	502
Greater London	MBTU	-160,782		-160,782
	MWH	-9,328	26,231	16,903
	Carbon avoided (mt)	-2,755	3,637	882

With respect to buildings' use of energy for heating, trees that shelter buildings from the prevailing wind offer energy savings. However, trees planted to the south can shade a building, resulting in more energy being required for heating, especially where the canopy is dense and the height to canopy base is low, restricting wintertime sun from warming the building. This explains the negative results for heating in London. With respect to buildings' use of energy for cooling, trees planted to the west and east can partially block incoming solar radiation, thus reducing air-conditioning usage in the warmer months. Similar types of analysis have been obtained in the city of New York (Nowak *et al.*, 2007). Mc Pherson *et al.* (1997) provide a more general estimate of the effect of tree cover. They estimate that in the city of Chicago, increasing tree cover by 10% (corresponding to about three trees per building) could reduce total heating and cooling energy use by 5 to 10%. On a per-tree basis, annual heating energy can be reduced by about 1.3% (2.1 GJ), cooling energy by about 7% (0.48 GJ), and peak cooling demand by about



6% (90.3 kW). These figures may be more easily transferable but are nevertheless strongly linked to the characteristics of the city of Chicago. For example, the impact on heating energy is estimated to be positive in Chicago and negative in London. Directly transferring values estimated in these studies is therefore hazardous.

Other studies have focused on the quantification of the temperature reduction obtained in the presence of urban vegetation. A meta-analysis by Bowler et al. (2010) shows a 0.94°C reduction of temperature in urban parks. Similarly, effects on temperature are identified related to single street trees and clustered trees. Green roofs are shown to have a relatively limited effect at street level (De Munck, 2013; Alcazar *et al.*, 2016) but a significant impact in terms of energy use reduction in buildings (Zinzi and Agnoli, 2012). For example, Zinzi and Agnoli estimate a 10-14% energy saving for buildings with green roofs in Barcelona due to reduced heating in winter and cooling effect in summer.

Monetary assessment

The assessments of the impact of vegetation on energy consumption based on models such as I-tree Eco are generally quite straightforward to estimate in economic terms. For example, **Table 7** provides an estimation of annual cost and saving due to trees based on the value of £149.20 per MWH and £14.06 per MBTU. Carbon avoided values based on the reduction of the use of fossil fuels are calculated based on £62 per metric ton (See 3.2.1 for the estimation of Carbon values).

Table 7. Annual costs and savings due to trees near buildings (in pounds) (Rogers *et al.*, 2015)

		Heating	Cooling	Total
Inner London	MBTU	-£664086.00		-£664085.00
	MWH	-£389859.00	£1277152.00	£887292.00
	Carbon avoided	-£49980.00	£73543.00	£23564.00
	Total	-£1,103,925.00	£1350695.00	£246770.00
Outer London	MBTU	-15,959,130.11	-	-15,959,130.11
	MWH	-1001878	2636513	1634635
	Carbon avoided	-120857	151987	31129
	Total	-2719784	2788450	68715
Greater London	MBTU	-2261134		-2261134
	MWH	-1391737	3913665	2521927
	Carbon avoided	170810	225494	54684
	Total	-3823682	4139159	315477

Other studies are based on the transfer of values of energy savings from other studies. For example, Tardieu (2014) estimates the benefits of trees by using reference values from Mc Pherson (1997) for the reduction of air conditioning (AC). In Chicago, the impact of a large street tree could reduce energy consumption for AC by 2 to 6%. Considering that in Europe, the cost



of air conditioning is estimated to be of 14€/person/yr and that the average size of household is 2.3, the estimated benefit for household living in buildings near parks is of 1.3 €/HH/yr which needs to be multiplied by the number of household living near parks. This approach is based on very strong assumptions on the possibility to transfer values from other contexts and are therefore rather imprecise. The monetary valuation of avoided carbon emissions, due to reduces use of heating or cooling, can be derived from carbon prices derived from the methods described in the precedent section.

In addition to energy savings, the reductions of temperature quantified in the studies mentioned above are likely to have a significant effect on human health. Methodologies have been developed to estimate a monetary value of the health impact of vegetation due to the reduction of air pollution (see section 3.2.3). It is however complicated to disentangle these two effects and health impact is generally not estimated in the assessment of economic effects of local climate regulation.

3.2.3 Air quality regulation

Description

The last report on Air Quality in Europe (EEA, 2018) reveals that air pollution exceeds EU limit values in large parts of Europe. For example, Particulate Matter (PM) with a diameter of 10µm or less (PM₁₀) concentrations above the EU daily limit value were registered at 19% of the stations in 19 of the 28 EU Member States (EU-28). The annual limit value for nitrogen dioxide (NO₂) continues to be widely exceeded across Europe, even if concentration and exposure are decreasing. In 2016, around 12 % of all the reporting stations recorded concentrations above this standard. Europe's most serious pollutants in terms of harm to human health are PM, NO₂ and ground-level O₃. Estimates of the health impacts attributable to air pollution indicate that in 2015, exposure to PM_{2.5}, NO₂ and O₃ concentrations were respectively responsible for 391,000, 76,000 and 16,400 premature deaths in the EU-28 (EEA, 2018). Air pollutants also affect crop production and ecosystems. For example, estimates of vegetation exposure to O₃ indicate that the EU target value for protection of vegetation was exceeded in 2015 in about 30 % of the agricultural land area of the EU-28. Stationary and mobile combustion processes occurring in industry, energy production, heating and transport are the main source of many primary pollutants.

NBS generate co-benefits in terms of air quality thanks to the action of trees, grass and shrubs, which are either preserved or planted. Trees and vegetation remove gaseous air pollution by uptake via leaf stomata, surface deposition and gas-phase reactions. They also remove pollution by intercepting airborne particles, which are retained on the plant surface (Nowak, 1994). Vegetation may however also have counterproductive effects on pollution. Indeed, the emission



of Biogenic volatile organic compounds (BVOCs) from trees and other vegetation can contribute to the formation of ground-level O₃ and CO air pollutants

The overall approach to estimate the production of air quality co-benefit is the following:

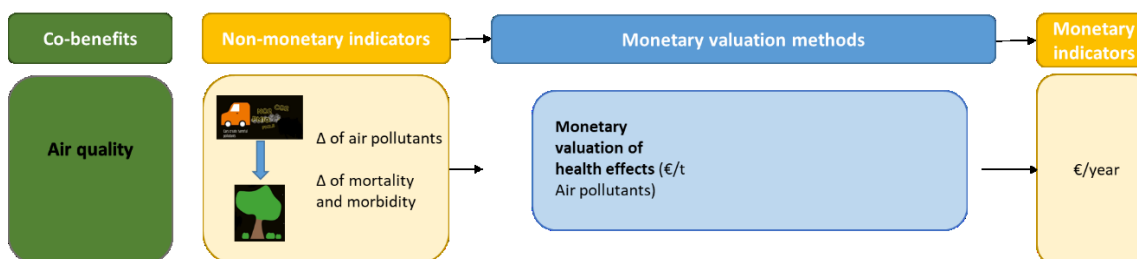


Figure 14. Overview of the economic valuation of the air quality service

Biophysical assessment

A large number of studies have been carried out to quantify the impact of vegetation on the removal of air pollutants (e.g. Baró et al., 2014; Bottalico et al., 2016; Nowak et al., 2006). The Itree-eco model, based on the UFORE model (Nowak *et al.*, 2008) has been widely used throughout the world to quantify this impact. This model uses data collected in the field from single trees, complete inventories, or randomly located plots throughout a study area along with local hourly air pollution and meteorological data to quantify the various impact of trees, including the effect on the removal of air pollutants. Similar approaches have been used to quantify the impact of grassland and shrubs on air quality (Gopalakrishnan *et al.*, 2018) as well as on specific NBS such as green roofs or vegetated walls (Yang *et al.*, 2008).

If field sampling is not possible, air pollution removal rates can be derived from existing studies in similar contexts. Aeverman and Schmude provide a review of existing results in Table 8.

Table 8. Case studies with amounts of air pollution removal (Aeverman and Schmude, 2015)

Study Area and Type	Author	Amount (g/m ² /yr)					
		CO	NO ₂	O ₃	SO ₂	PM ₁₀	PM _{2.5}
Boston (USA), Urban Trees	NOWAK et al. (2006)	0.30	2.10	3.80	1.10	2.80	-
Santiago (Chile), Urban Trees	ESCOBEDO et al. (2008)	-	-	-	-	7.40	-
Santiago (Chile), Grass	ESCOBEDO et al. (2008)	-	-	-	-	1.70	-
Guangzhou (China), Urban Trees in Residential Area	JIM/CHEN (2008)	-	2.53	-	3.06	-	-
Santiago (Chile), Urban Trees	ESCOBEDO/NOWAK (2009)	0.40	1.20	2.80	0.60	7.40	-
Lanzhou (China), Urban Trees	JIM/CHEN (2009)	-	-	-	17.10	-	-
London (UK), Urban Trees, Flux Model	TALLIS et al. (2011)	-	-	-	-	2.73	-
London (UK), Urban Trees, Tiwary Approach	TALLIS et al. (2011)	-	-	-	-	5.85	-
National Park near Mexico City (Mexico), Fir Forest	BAUMGARDNER et al. (2012)	0.02	0.07	1.97	0.23	7.13	-



Other values for other type of covers, in Europe or for NBS are presented in **Table 9**.

Table 9. Case studies of amounts of air pollutant removal thanks to green infrastructure (own compilation)

Study area and type	Author	Annual removal rate of pollutants in g/m ²					
		CO	NO ₂	O ₃	SO ₂	PM ₁₀	PM _{2.5}
Munich, Urban Park	Aevermann and Schmude, 2015	0.2	3.1	4.1	0.4	4.0	0.6
Barcelona, Spain, Trees and shrubs	Baró <i>et al.</i> , 2014					5.1	
Florence, Italy, Coniferous	Bottalico <i>et al.</i> , 2016			0.23		0.20	
Florence, Italy, Deciduous broadleaved	Bottalico <i>et al.</i> , 2016			0.09		0.15	
Florence, Italy, Evergreen broadleaved	Bottalico <i>et al.</i> , 2016			0.31		0.18	
Florence, Italy, Mixed broadleaved and coniferous	Bottalico <i>et al.</i> , 2016			0.21		0.24	
Chicago, USA, Short grass	Yang <i>et al.</i> , 2008		2.33	4.49	0.65	1.12	
Chicago, USA, Tall herbaceous plants	Yang <i>et al.</i> , 2008		2.94	5.81	0.83	1.52	
Chicago, USA, Deciduous trees	Yang <i>et al.</i> , 2008		3.57	7.17	1.01	2.16	
Chicago, USA, green roofs	Yang <i>et al.</i> , 2008		2.23	4.42	0.60	1.19	
US, Urban areas grassland cover	Gopalakrishnan <i>et al.</i> , 2018		0.25	2.93	0.14		0.03
US, Rural areas, grassland cover	Gopalakrishnan <i>et al.</i> , 2018		0.25	2.43	0.14		0.03
US, Urban areas, shrubland cover	Gopalakrishnan <i>et al.</i> , 2018		0.22	1.47	0.08		0.01
US, Rural areas, shrubland cover	Gopalakrishnan <i>et al.</i> , 2018		0.22	1.47	0.08		0.01

Monetary assessment of air quality improvement

Assessing the monetary value of the improvement of air quality requires assessing a monetary value of the health impact related to the reduction of air pollutant associated with the deployment of NBS.

Many studies on the impact of vegetation on the improvement of air quality, use the monetary assessment module included in the I-tree eco software (eg. Aevermann and Schmude, 2015), which is based on BenMAP (Environmental Benefits Mapping and Analysis Program) (Sacks *et al.*, 2018). BenMAP determines health impacts in the USA due to specific pollution concentrations in the United States. These estimates include the cost associated with mortality and morbidity (i.e illness related to air pollution). The limit of the application of these studies



is that they are based on estimates carried out in the USA however it has the advantage to provide direct estimation of the monetary value to users of I Tree Eco.

In Europe, guidelines for the Cost-Benefit Analysis (CBA) of Air Quality Related Issues have been developed in the Clean Air for Europe (CAFE) Programme (EEA, 2005a). The quantification of health impacts addresses the impacts related to both long-term (chronic) and short-term (acute) exposures. The quantification deals with both mortality (i.e. deaths) and morbidity. The mortality effects quantified in the CAFE cost-benefit analysis include impacts on infants as well as adults. The morbidity effects that can be quantified include major effects, such as hospital admissions and the development of chronic respiratory disease. They also include less serious effects, which are likely, however, to affect a greater number of people. Examples of these are changes in the frequency of use of medicine to control asthma, and days of restricted activity. When the impact and the values are combined in the analysis, the most important health related issues relate to mortality, restricted activity days and chronic bronchitis.

Economic valuation of mortality linked to air pollution has been the subject of debate for a number of years. The CAFE methodology is essentially based on a method developed in a study for the European Commission, DG Research (the NewExt study). Besides the ethical debate on whether the cost of deaths should be evaluated economically, one of the debate is on whether the Value of Statistical Life (VSL) or the Value of Life Year (VOLY) should be used to quantify the value of mortality. This debate is quite technical. The Value of Statistical Life is essentially derived from surveys asking people to express their willingness to pay, or the amount of money they are willing to spend (WTP), to secure a reduction in the probability of premature death of one year. The economic benefit of a mitigating action becomes the VSL value multiplied by the number of prevented deaths. The VOLY refers to the willingness to pay to increase the life of one year. It is therefore associated with studies that estimate the number of life years that are lost due to air pollution rather than the number of premature deaths. The values used in the CAFE are presented in **Table 10**.

Table 10. Values for use in CAFE CBA: Effects of chronic exposure on mortality (EEA, 2005)

	VSL	VOLY
Median (NewExt)	€980,000	€52,000
Mean (NewExt)	€2,000,000	€120,000

Although this section normally reports on direct valuation approaches, we include these methods based on stated preference methods as they go side by side with direct valuation approaches used to evaluate the cost of morbidity such as the ones presented below.



In terms of morbidity, the CAFE guidelines take into account chronic effects of PM on Bronchitis as well as acute effect on respiratory hospital admissions, cardiac hospital admissions, consultations with primary care physicians, restricted activity days, use of respiratory medication, symptom days. the guidelines also provide figures for acute morbidity due to O3: respiratory hospital admissions, minor restricted activity days, use of respiratory medication symptom days.

Based on the estimation of the impact of pollutants obtained thanks health data from UN health statistics and data, the EEA has issued externality value in €/T for the major pollutants of the EU-25 (EEA, 2005b). This report also provides per country externality values that can be retrieved in the report.

Table 11. Average damages per tonne of emission of NH₃, NO_x, PM_{2.5}, SO₂ and VOCs for the EU25 (excluding Cyprus) and surrounding sea areas (EEA, 2005b)

Average damages per tonne of emission of NH₃, NO_x, PM_{2.5}, SO₂ and VOCs for the EU25 (excluding Cyprus) and surrounding sea areas under different sets of assumptions. Detailed results specific to each country and sea area are given in the main text.

PM mortality	VOLY median	VSL median	VOLY mean	VSL mean
O ₃ mortality	VOLY median	VOLY median	VOLY mean	VOLY mean
Health core?	Included	Included	Included	Included
Health sensitivity?	Not included	Not included	Included	Included
Crops	Included	Included	Included	Included
O ₃ /health metric	SOMO 35	SOMO 35	SOMO 0	SOMO 0
EU25 (excluding Cyprus) averages				
NH ₃	€11,000	€16,000	€21,000	€31,000
NO _x	€4,400	€6,600	€8,200	€12,000
PM _{2.5}	€26,000	€40,000	€51,000	€75,000
SO ₂	€5,600	€8,700	€11,000	€16,000
VOCs	€950	€1,400	€2,100	€2,800

The sensitivity of these values is based on a different set of methodological uncertainties. The different values presented in the different columns should be used for sensitivity analysis

- Valuation of mortality using the value of statistical life (VSL) and value of a life year (VOLY) approaches.
- Quantification of ozone effects on health with and without a 'cut-point' (effectively, the assumption of a threshold at 35 ppb).
- Separation of health impacts into a 'core' set of functions which we conclude to be most robust, and a 'sensitivity' set of functions that are less robust.

These estimates also include estimation of the impact of air pollution on crop yields which is not detailed here.



An alternative to the use of ITree eco, is therefore to use reference values for the estimation of pollutants removal for the main pollutants due to NBS and then to use the economic values associated with pollutant removal estimated by the EEA.

Box 4 Example of Quantification and monetary valuation of air quality regulation service in Munich, Germany (Aevermann and Schmude, 2015)

This study quantifies and monetizes four ecosystem services (carbon sequestration and storage, air pollution removal, runoff reduction, groundwater recharge) provided by the urban green space of Schlosspark Nymphenburg in Munich, Germany.

As a start, the area size of different land cover types are estimated through orthophotos. The park includes 1 198 645m² of trees and 234 337 m² of grassland.

In order to estimate pollutant removal, the i-Tree Eco model is used. The following input data are used to run the software: hourly pollution concentrations for six pollutants CO, NO₂, O₃, SO₂, PM₁₀ and PM_{2.5} as well as local hourly weather data for 2011. In this example, no sampling of vegetation is used which is an interesting simplification of the use of this software. A leaf area index (LAI) of 6.0 m²/m² is used as an approximation for managed mixed forests in this region.

To estimate monetary values, i-Tree uses BenMAP to estimate generated costs to the health system in relation to local population density. The output of the ITree –eco software is presented in **Table 12**. The discussion mentions that one of the weakness of the approach is to use a monetary value based on mortality and morbidity values estimated in the United States.

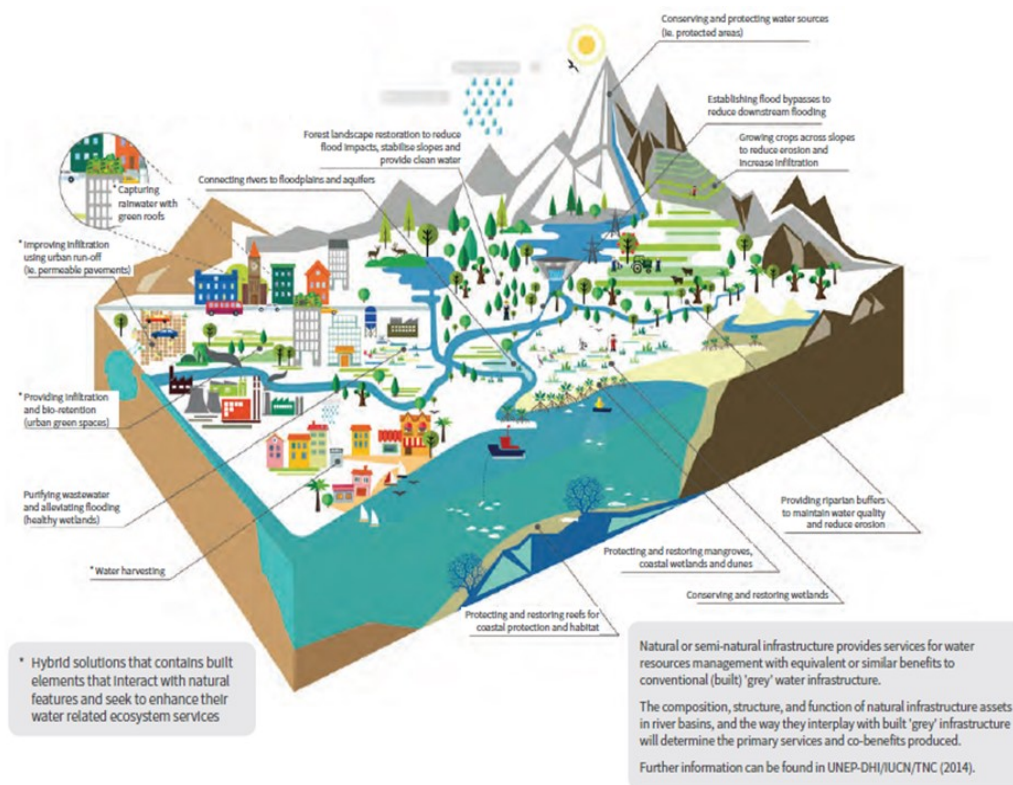
Table 12. Annual amounts and monetized values of air pollution removal for Schlosspark Nymphenburg

<i>Air Pollutant</i>	<i>Amount per Area (g/m²)</i>	<i>Amount (t)</i>			<i>Value per Ton (€/t)</i>	<i>Value (€)</i>		
		Total	Min	Max		Total	Min	Max
CO	0.2	0.28	0.28	0.28	1 116	310	310	310
NO ₂	3.1	3.67	1.38	4.83	998	3 660	1 380	4 820
O ₃	4.1	4.92	1.17	6.75	8 313	40 920	9 700	56 120
SO ₂	0.4	0.44	0.22	0.81	412	180	90	340
PM ₁₀ *	4.0	4.82	2.05	7.14	5 252	25 290	10 790	37 490
PM _{2.5}	0.6	0.68	0.09	1.44	475 429	321 810	43 320	686 230
Total	-	14.81	5.19	21.25	-	392 170	65 590	785 310



3.2.4 Water cycle regulation

NBS can provide a variety of water-regulating services. The physical, chemical and biological properties of NBS affect all the hydrological pathways in the water cycle. Biological processes in a landscape, and especially in soils, influence the quality of water as it moves through a system, as well as soil formation, erosion and sediment transport and deposition – all of which can exert major influences on hydrology (WWAP, 2018). NBS are becoming increasingly recognised as an important opportunity for addressing the challenges of water management (UNEP, 2014; Abell et al., 2017; WWAP, 2018). A wide range of options is available for consideration (**Figure 15**), from those implemented at the catchment scale to improve water supply into urban areas (e.g., reforestation, restoration of wetlands) to those deployed within urban boundaries (e.g., permeable pavements, green spaces).



Source: Infographic 'Natural Infrastructure for Water Management,' © IUCN Water 2015.

Figure 15. NBS for water management across a landscape (WWAP, 2018)

Water regulating services can be broadly divided into services affecting water availability (supply or quantity), those affecting water quality and those moderating risk and extremes (including



flood control¹⁰). These services provide benefits to a variety of beneficiaries and valuing them requires an understanding of the biophysical processes at work and the specific beneficiaries. There is a high degree of variation in the impacts of ecosystems on hydrology both within and between ecosystem types or subtypes, their location and condition, climate and management. This cautions to avoid generalized assumptions about NBS, as site-specific knowledge on their field deployment is required (WWAP, 2018).

This section will mostly focus on the benefits for the drinking water sector, as ecosystems and NBS play a fundamental role in providing cities with fresh water for drinking and by securing storage (Gomez-Bagghetun, 2013). The overall approach consists in assessing the effects of NBS on the water regulation service level in biophysical terms, and to value these changes in economic terms (**Figure 16**).

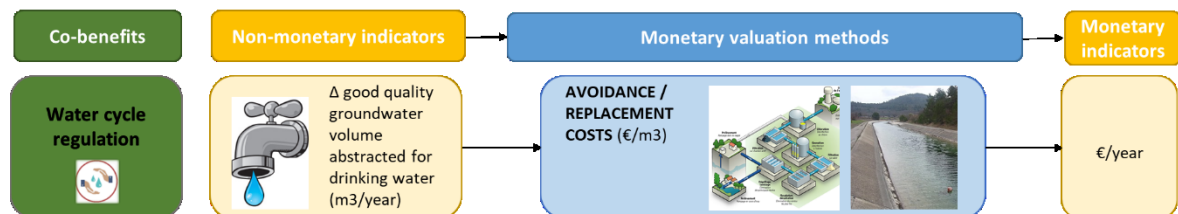


Figure 16. Overview of the economic valuation of the water regulation service for the drinking water sector

Biophysical assessment of water availability

Ecosystems exert a major influence on the quantity of water available in time and space. Most notably the soil–vegetation interface is the key determinant of the fate of precipitation by influencing infiltration from the land surface, and hence groundwater recharge, surface runoff and soil moisture retention in the plant root zone (of particular importance to agriculture), and finally recycling water back to the atmosphere through evaporative fluxes. NBS essentially involve managing these pathways, either through ecosystem conservation or rehabilitation, and through various land use and management approaches, whether at small or landscape scales or in urban or rural settings (WWAP, 2018). Managing water flows through urban landscapes can increase water storage and therefore resilience to variations in water availability.

The impact of NBS on water availability depends not only on vegetation types, species and location, but also on hydrological and/or hydrogeological properties. This service is generally valued in biophysical terms as the water balance (Mulligan, 2018) or the groundwater recharge (Aevermann and Schmude, 2015; Johnson and Geisendorf, 2019). Biophysical indicators are generally obtained from modeling of experimental data on NBS measures (Johnson and

¹⁰ Considered as a primary benefit in NAIAD i.e., out of the scope of this deliverable.



Geisendorf, 2019). The overall role of ecosystems on the supply of water quantity is however still an open empirical question. For woodlands for instance, Binner et al. (2017) highlight that the absence of robust biophysical evidence to quantify the relationship between woodland and changes in the quantity of water available constitutes a significant barrier to reliable valuation and decision-making. Methodological advances are still required to enable the biophysical modelling to make robust predictions of changes in water quantity attributed to woodlands (Binner et al. 2017). Trees can increase or decrease groundwater recharge according to tree type, density and location (Borg et al., 1988; Ilstedt et al., 2016). The tree–soil and moisture–groundwater relationships are also dependent on the size and age of the trees in question (Dawson, 1996). Forests typically have much greater evaporation rates than grasslands where rainfall exceeds 2,000 mm per year, but comparable rates where rainfall is less than 500 mm per year (Zhang et al., 2001).

Moreover, local hydrological and/or hydrogeological properties play a key role. Booth et al. (2016) explains that groundwater recharge is only the entry-point to a complex groundwater flow system. Groundwater recharge leads to an increase in storage in the recharge location that can be quite removed in distance and travel time from pumping locations. Booth et al. (2016) explain that while perceiving groundwater recharge as a beneficial service that provides adequate water supply to human and ecosystem needs in regions where water supplies have been or are threatened to become diminished, this viewpoint does not necessarily hold in other regions where groundwater supplies are abundant and/or have been increasing due to increased precipitation or land use change. A manifestation of excess groundwater supply is groundwater flooding, which occurs when the water table rises above the ground surface. Booth et al. (2016) suggest moving beyond groundwater recharge as a stand-alone ecosystem service, and instead propose that observations and biophysical models should be used to quantify the final service humans receive from groundwater (e.g. reliability of water supply from a municipal well).

Biophysical assessment of water quality

The potential benefits of watershed protection for enhancing the quality of water available for human settlements, and cities in particular, are massive (WWAP, 2018). For example, a recent modelling exercise by Abell et al. (2017) estimated that land conservation and/or restoration activities (such as forest protection, reforestation and the use of cover crops in agriculture) could lead to a 10% (or more) reduction in sediments or nutrients (phosphorus) in watersheds that currently cover 37% of the world's ice-free terrestrial surface (4.8 million km²). More than 1.7 billion people (over half of the world's urban population) living in the 4,000 cities in the area covered by this study could therefore potentially benefit from improved water quality as a result of NBS applied to their source watersheds (Abell et al., 2017).



NBS, when managed properly, provide high-value 'green infrastructure' for enhancing source water protection. They play important roles in regulating water flows and maintaining water quality by reducing sediment loadings, through the prevention of soil erosion and by capturing and retaining pollutants (UNEP, 2014). There is a rapidly growing interest in incorporating green infrastructure in urban planning and design to manage and reduce pollution from urban runoff (UNEP, 2014). Examples include using green walls, roof gardens, trees in streets and vegetated infiltration or drainage basins. Wetlands and other sustainable drainage features are also widely used within urban environments to mitigate the impact of polluted stormwater runoff and wastewater (WWAP, 2018).

Biophysical indicators can be expressed with a range of metrics, depending on how the change in water quality affect potential beneficiaries: e.g., concentration of sediments; nutrients, including nitrates and phosphates; pesticides; the proportion of water that requires treatment. Biophysical models are generally required to assess the effect of NBS implementation on water quality.

InVEST proposes a nutrient delivery model¹¹ to map nutrient sources from watersheds and their transport to the stream. The model uses a simple mass balance approach, describing the movement of a mass of nutrient through space. Unlike more sophisticated nutrient models, the model does not represent the details of the nutrient cycle but rather represents the long-term, steady-state flow of nutrients through empirical relationships. Sources of nutrient across the landscape, also called nutrient loads, are determined based on a land use/land cover (LULC) map and associated loading rates. Input data include a digital terrain model (DTM), average precipitation, bibliographic data on evapotranspiration, water retention and nutrient storage capacities of soils and plants and land use. Nutrient retention is reported (phosphorus and nitrogen) in kg / ha / year.

i-Tree Hydro¹² is an other tool for users interested in comparative analyses of different land cover scenarios and their hydrological impacts at various scales. i-Tree Hydro simulates the effects of changes in urban tree cover and impervious surfaces on the hydrological cycle, including streamflow and water quality. I-Tree Hydro is a combination of two modules. A base module designed to simulate hourly changes in stream flow due to changes in urban tree and impervious cover characteristics and a water quality module that uses outputs from the base program to simulate changes in water quality.

¹¹ <http://releases.naturalcapitalproject.org/invest-userguide/latest/ndr.html>

¹² <https://www.itreetools.org/hydro/>



Monetary valuation of water regulating services

Water regulating services may impact a wide range of human activities (e.g., agriculture, drinking water supply, hydroelectric production, recreational activities) through changes in four attributes: quantity, quality, location and timing of flow (Brauman et al., 2007). This section illustrates how direct valuation approaches can be used to assess the benefits provided by NBS through an increase in the water regulation service. We will illustrate these approaches by focusing on the benefits for the drinking water sector, i.e. in cases where NBS can contribute to improve the quality and/or the availability of water resources used for drinking water production.

A popular approach among practitioners to assess the benefits of an increase in water quality and/or availability is the avoidance-cost method (for examples see Abdalla, 1994; Rinaudo et al., 2005). It consists of assessing the cost of actions undertaken by economic agents to cope with water contamination and/or scarcity. Typical avoidance costs are those related to the closure and displacement of abstraction points (if the existing ones are polluted or not sufficient for water supply), the installation of sophisticated water treatment units (in case of pollution), increased energy costs required for water abstraction (due to lower water levels), or the purchase of bottled water (when the water resource can no longer be used as a safe source of drinking water).

Two types of damage costs may typically result from changes in water quality and/or availability at each withdrawal point: the avoidance actions taken by drinking water companies and the averting behaviors adopted by tap water consumers. **Figure 17** illustrates how these damage costs may evolve as a function of water quality indicators (here nitrate concentration) provided by biophysical models. A similar approach could be used with water availability indicators.

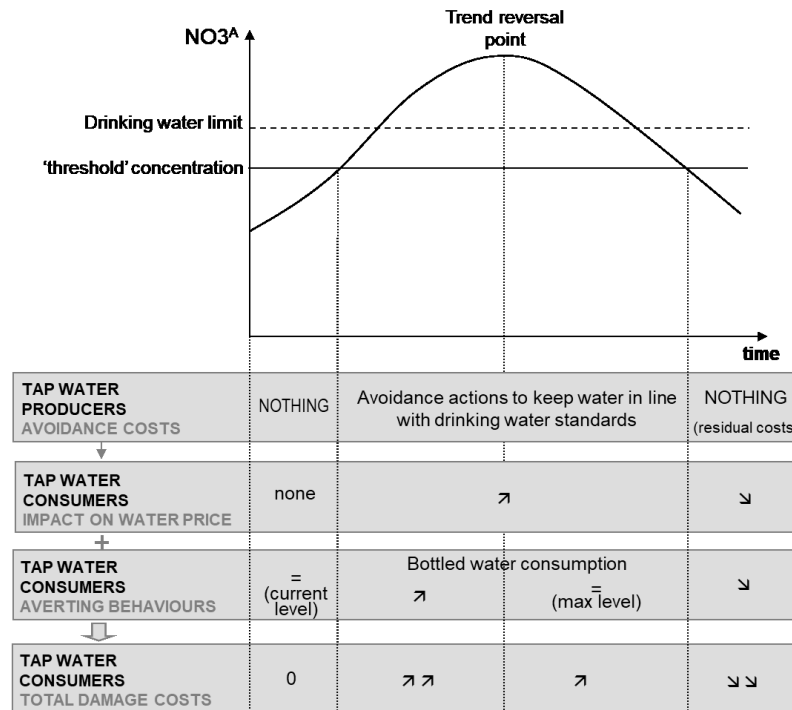


Figure 17. Economic impacts for tap water consumers resulting from a change in groundwater quality (Hérivaux et al., 2013)

Tap-water producers should take avoidance actions if nitrate concentrations at groundwater withdrawal points exceed 50 mg/l (Drinking Water Directive), in order to keep public-water quality in line with drinking-water standards. These may consist of well closure, well displacement, water treatment, or a switch to other water resources. Damage costs (D_P) resulting from these avoidance actions can be gathered through the consultation of experts in the drinking water sector and from a review of local technical literature. In France, Bommelaer and Devaux (2011) estimates that water treatment costs range from 0.40 to 0.60 €/m³ for the treatment of nitrates and from 0.06 to 0.20 €/m³ for the treatment of pesticides.

At the household level, the avoidance-cost method infers damage by measuring the consumption of goods or services that substitute for the environmental quality change (Abdalla, 1994). The underlying assumption is that people make choices in order to maximize their level of well-being when faced with the increased health risks associated with exposure to unsafe drinking water (Whitehead et al., 1998). Tap-water consumers may increase their consumption of bottled water to replace tap water because of a deterioration in groundwater quality. Although nitrate concentrations in distributed water may not exceed the 50 mg/l limit (according to the requirements of the Drinking Water Directive) and bottled water costs 160



times more than tap water, this behavior may be explained by the fact that consumers generally prefer naturally clean water to water that has been polluted and treated (Hasler et al., 2005). In France, these damage costs (D_c) resulting from these averting behaviors were estimated between 151 and 366 €/household/year.

Figure 17 assumes that these avoided actions and averting behaviors are adopted shortly before the drinking water limit is reached, when the nitrate concentration exceeds the 48 mg/l defined here as the ‘threshold concentration’, and then gradually abandoned as soon as the nitrate concentration drops below that threshold.

The benefits B (€/year) resulting from an improvement in groundwater quality are assessed as damage avoided from groundwater contamination. They can be expressed as the difference between damage costs in a baseline situation without NBS (D_{baseline}) and in a situation with NBS (D_{NBS}).

$$B = D_{\text{baseline}} - D_{\text{NBS}} \text{ with } D = D_p + D_c$$

One of the main advantages of this method is that it measures tangible costs that correspond to real expenditures made by concerned economic agents (investment, operation and maintenance costs). Results obtained are thus easy to grasp by policy makers and stakeholders. However, since it considers only direct-use values, this approach provides lower-bound estimates of the benefits.

Econometric approaches can also be used to assess the benefits provided by forests in terms of improved water quality. Such approaches are based on water bills data. They are not coupled with biophysical assessments. Figuepron et al. (2013) developed a cost-based econometric model to assess the benefits provided by forests in France in terms of improved water quality, estimating that on average 1 ha of afforestation would generate a saving for French domestic users of around 22€/user/year (in 2004 euros). Abildtrup et al. (2013) developed a spatial econometric analysis of the effect of forest land use on the cost of drinking water supply in Vosges, France, and found a reduction in household water bills ranging between 99 and 138 €/ha/year (in 2008 euros) of new forest.

Box 5. Example of valuation of the water cycle regulation service provided by the protection of strategic groundwater resources for the future.

Hérivaux and Grémont (2019) investigate if ecosystem services valuation improves the understanding of local stakeholders regarding the benefits that strategic groundwater preservation for future generations can bring to their territories, and may enhance their support towards groundwater conservation. They develop and test a systemic approach based on local data collection, cost-based monetary valuation methods and stakeholder involvement. The



study site in a strategic groundwater area (7400 ha) located in Southern France that could be of interest in the future to supply the growing population of coastal urban areas. They characterize, quantify and value in monetary terms nine ecosystem services, including the water cycle regulation service.

The approach adopted to assess the water cycle regulation service consisted of assessing the benefits associated with maintaining in its current state a land cover capable of recharging the aquifer (water availability) as well as filtering and purifying the water during its transfer to the aquifer (water quality). The beneficiaries of this service are residential users whose drinking water is currently extracted from the aquifer. Benefits were estimated by comparing the current situation with a situation that would require resorting either to another resource for drinking water supply, or to a water treatment facility. Two valuation methods were used: the avoided costs of water treatment and the avoided costs of a network interconnection with an alternative resource. The avoided costs of water treatment were estimated based on the mean cost of water treatment in France (0.46–0.8 €/m³ depending on treatment types) (Bommelaer and Devaux, 2011). According to local stakeholders, since the DWSZ is already crossed by a large water pipe that transports surface water from the Verdon River to the coast (i.e. the Canal de Provence), a straightforward alternative if the aquifer was no longer able to provide enough good-quality water would be to resort to the water provided by the Canal de Provence. Considering that the costs of connecting to this pipe are negligible, the avoided costs of interconnecting with an alternative resource were estimated by directly comparing water prices for consumers that are currently supplied by the aquifer with those that are currently supplied by the Canal de Provence. The assessment was undertaken taking into account differences in water utility management (direct public management or public service concession). All assessments accounted for technical efficiencies of local drinking water networks.

Benefits provided by water cycle regulation were estimated between 208 and 551 k€/year, i.e., between 28 and 75 €/ha/year.



Box 6. Costing Nature and the water regulation service

Costing Nature (Mulligan, 2018) proposes a biophysical assessment of the water regulation service, with the relative realised water provisioning services indices.

Relative realised water provisioning services indices (RRWPS)

Clean water provision is an important provisioning service. The RRWPS is an index of the volume of clean (not human impacted, see [WaterWorld Human Footprint on Water Quality HFWQ index](#)) water available to downstream beneficiaries.

Water quantity is each pixel is calculated as the water balance (rainfall minus actual evapotranspiration) cumulated downstream. See [Mulligan et al. \(2011\)](#) for a description of the global water balance dataset. Potential water provisioning services for each cell are first calculated as the volume of clean (i.e. no 100-HFWQ) water available from upstream. The volume of water is calculated as downstream cumulated water balance based on (rainfall+fog+snowmelt)-actual evapotranspiration (WaterWorld) or rainfall (Hijmans et al., 2005) minus actual evapotranspiration (Mulligan et al. 2011) (Co\$tingNature). This is the potential water service. The realised water services depends on the intensity of downstream use measured as the normalised area of irrigation, number of people and number of dams. The RRWPS is thus the product of the normalised potential service and the normalised downstream beneficiaries. RRWPS is high where the prevailing climate and land use generate high volumes of clean water which can be used (and reused) by large numbers of

downstream users. The greater the downstream population, number of dams and actual water available, the greater the service provided. If there is plenty of water, but no people or dams, then there is no realised service. In this way not all water provides a direct service, only that water that is accessed and used. *Untapped water services* are considered to be the difference between potential and realised water services.



Costing Nature also offers the possibility to users to integrate two types of economic values to assess the water regulation service: the total dam and urban water supply revenue in the study area, and the amount willing to pay per capita per year (rural communities).

Water (intakes): fractional natural capital footprint upstream of intakes

This metric maps directly on to the economic value *total dam and urban water supply revenue in study area (USD)*. By mapping nature's contribution to realised water services at intakes the total revenue in the study area can be spatially distributed to reflect the value provided in each pixel. Rivers flowing into urban areas are considered to provide water resources to those areas so for each pixel the number of these urban intakes downstream is summed as an indicator of the demand side. Similarly the number of downstream dams are summed and added to the urban intakes. Nature's potential contribution to this service at intakes is calculated as the the natural capital footprint in a pixel. The natural capital footprint is 100 minus the HFWQ and reflects the potential dilution service of natural areas on improved water quality provided to these areas. The realised service is calculated as the product of the potential service and the normalised downstream number of urban 'intakes' and dam intakes. The total revenue gained through HEP generation, irrigation and urban water supply is distributed across the study area according to this demand weighted natural capital footprint. Thus 100% intervened (HFWQ) areas upstream of demand are assumed to produce none of this value: all the value is produced on non intervened or partially intervened areas upstream of demand and in accordance with the volume of water flowing through them (which is part of the HFWQ metric).

Water (rural): per-capita fractional natural capital footprint to rural populated areas

This metric maps directly on to the economic value: *amount willing to pay per capita per year*). Water quality services to rural communities without access to water treatment are an important service. We calculate the potential service as the [NFWQ \(Natural Footprint on Water Quality\)](#) (i.e. no 100-HFWQ) or the dilution service provided by water emanating from natural lands. However, rainwater harvesting by roofs is considered a key water source away from rivers (Strahler stream order =1) and effectively disconnects these communities from hydrological ecosystem services since their water does not interact with the land. Thus rural downstream populations for each pixel are calculated net of these hillslope populations that are far from a river or stream. Non urban communities at stream order >1 are considered to use runoff as their main source of water (whether accessed through groundwater, springs or streams). The realised service is calculated as the product of the rural population weighted NFWQ and the per-capita willingness to pay for this service in order to distribute the service use value spatially.



3.2.5 Recreational and ecotourism activities

Foreword on cultural services

Cultural services represent several challenges for economic valuation. They are non-material outputs of ecosystems and their definition encompasses a range of experiential and intellectual benefits which have been described with different concepts of value (Kenter et al., 2015). Cultural services seem to be used as a “catch-all term” to name the remainder of the benefits people obtain from ecosystems; i.e. the benefits which cannot classify in the other identified categories. In the Millennium Ecosystem Assessment framework, the category of cultural services is defined as “non-material benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences, including Cultural diversity, spiritual and religious values, knowledge systems, educational values, inspiration, aesthetic values, social relations, sense of place, cultural heritage values, recreation and ecotourism (Alcamo et al., 2005, pp.58-59) “. The CICES reformulates this definition by describing the category of cultural ES according to the characteristics of ecosystems that enable people to “do” physical or intellectual activities (e.g. sport, recreation, nature watching, education or research) and to “feel” more deep emotions related to culture (e.g. “The things in nature that help people identify with the history or culture of where they live or come from”, “The things in nature that have spiritual importance for people”) (Haines-Young and Potschin, 2017). Individuals can associate these benefits with different domains of value including instrumental domain interpreting values as “ecosystems and biodiversity seen as merely a means to achieve utility”; intrinsic domain interpreting values as “ecosystems and biodiversity have value in themselves”; fundamental domain interpreting values as “conditions to protect the life supporting system, conditions to allow people to define themselves and condition to provide sense to their existence” and eudaimonistic domain interpreting values as “entities and processes which represent condition for having a good human life” (Arias-Arévalo et al., 2018). The fundamental and eudaimonistic domains value correspond to the relational domain value defined by (Pascual et al., 2017). Arias-Arévalo et al. (2018) underline that values within intrinsic, fundamental and eudaimonistic domains, represent shared values and are related to “higher ends” of the meanings of the life. These domains of value challenge the economic valuation mainly for two reasons. First, ES within intrinsic, fundamental and eudaimonistic domains are incommensurable (Arias-Arévalo et al., 2018; Kenter et al., 2015). Second, people hold lexicographic preferences that lead them to classify things/goods in distinct domains that cannot be trade-offs (Choy, 2018; Spash, 2015).

Given the different meanings of cultural benefits for people and the diversity of concepts encompassed in the definition of the category of cultural services; we consider that the definition of the cultural ES at stake, as well as its domain of value need to be specific and explicit. **Only cultural ES within the instrumental domain can be valued with economic**



methods as they are related to people's preferences to gain utility (Arias-Arévalo et al., 2018). Here, we make the general assumption that cultural ES defined from “what people do”, that is physical or intellectual activities belong to the instrumental domain while ES from “what people feel” belong to the other domains. We acknowledge that there is a need for examining the diversity of meanings of cultural ES. However, we omit valuation methods for the other domains of value for the sake of focusing on the central aims of this deliverable.

Description

Recreational and ecotourism activities are by far the most valued sub-category of cultural services in monetary terms. About 85% of estimated values of cultural services reviewed by De Groot et al. (2012) are related to recreational and ecotourism activities which include activities such as swimming, recreational fishing, sightseeing, hiking, climbing, boating, bird watching, etc. (Hermes et al., 2018; Grizzetti et al., 2016). These activities benefit people through improved physical health, psychological and emotional well-being (Hermes et al., 2018). Their economic value ranges from \$7/hectare/year (Int.\$, 2007) for woodlands to \$ 96,302 /hectare/year (Int.\$, 2007) for Coral reefs (De-Groot et al., 2012). The availability of tourist visit data support the development of the economic valuation of recreational and ecotourism activities (Hermes et al., 2018). It is worth noting that although other cultural services such as aesthetic appreciation, inspiration, spiritual experience and cognitive development have been separately evaluated (De Groot et al., 2012), it is increasingly acknowledged in the literature that they are complementary to recreational and ecotourism activities (Hermes et al., 2018). They indeed enhance recreational opportunities.

Monetary assessment

Market-based methods have been primarily adopted for valuing cultural services and encompass direct market pricing, factor income and production function (De Groot et al., 2012).

The **direct market pricing method** derives values from directly or indirectly observed prices in markets. Direct and indirect economic impacts from recreational and ecotourism activities can be estimated by calculating gross revenues associated with a these activities (Harrison et al., 2018; Burke et al., 2008).

The **factor income method** uses observed market prices by focusing on current financial flows in the economy (Burke et al., 2008). It analyses the economic activity generated by recreational and ecotourism activities. This method considers that the remaining surplus, after subtracting the costs of other inputs in production from total revenue, as the value of the environmental input (Brander et al., 2007).



By contrast, the **production function method** focuses on a change in value by relating the output of marketed goods to recreational activities as inputs through the use of econometric techniques. (Jacobs et al., 2018; Burke et al., 2008).

Practically, the direct market method consists in two steps. The first consists in the valuation of direct economic impacts of the recreational and ecotourism activities. Direct economic impact represent the gross revenue received from recreational and ecotourism activities ignoring the cost of all other inputs (Brander et al., 2007). Burke et al. (2008) suggest that non-labour operating costs are not benefits to local economy. Hence, direct economic impact represent the gross revenue net of the non-labour operating costs (Burke et al., 2008). To compute this value, one needs to identify tourists, local visitors and the different monetary flows that they bring to the local economy: what is the profile of tourists and local visitors and how many are they? Where do tourists stay and for how long? How do tourists and visitors move? What do they buy? Here, we refer to local visitor as residents and tourists according to the definition of the United Nations World Tourism Organization (UNWTO). Tourists represent “travellers taking a trip to a main destination outside their usual environment, for less than a year, for any main purpose (business, leisure or other personal purpose) (Yu et al., 2012, p.446).

There are four categories of indicators to evaluate (Table 13). Surveys of visitors and interviews with experts are the most frequently used methodologies for data collection (Hermes et al., 2018). Fee revenues from recreational activities is a more direct monetary flow and can represent a significant part of the total revenue, because both tourists and local visitors pay to enjoy these activities. The fee can be the price paid for individual activities such as diving, snorkelling, viewing, and fishing or for a combination of activities (Brander et al., 2007). Expenditures related to the category of accommodation only concerns tourists. For instance, information can be compiled by individual hotel, by accommodation type (hotel, guest house, etc.), or based on average values for the country, region, or study area (Burke et al., 2008). The category of other expenditures are less visible in comparison to the two former categories, but represent an important part of the total monetary flows. Indeed, the estimates Burke et al. (2008) show that the categories of accommodation and other expenditures represent 57% and 38% of total direct monetary flows, respectively. The total revenue can be estimated by multiplying the number of visitors by per capita expenditures for each category (Horváth and Frechtling, 1999). However, estimating each categories of expenditures can be costly and time-consuming; making the case for transferring an average value per visitor from the literature (Hermes et al., 2018). For example, in the context of coral reefs, Brander et al. (2007) estimate the average value to 184 US\$ per visit while for more general context of Europe, Schägner et al. (2018) find an average value of € 7.17 per visit.



Table 13. Indicators for estimating cultural ES with market price method

Categories	Indicators	Sources
Tourists	– Number of visitors, – Number of staying days.	Official censuses of visitation, tourist visitation data.
Local visitor	– Number of visitors.	Official censuses of visitation
Accommodation	– Room rates, – Catering expenses, – Taxes and service charges.	Hotel, guest houses, camping sites, local or national statistics.
Recreational activities	Fee for the different recreational activities, e.g.: – Visiting, – Fishing, – Mooring, – Diving.	Companies or associations providing the activities, local or national statistics.
Other visitor expenditures	– Expenditures on restaurants, – Expenditures on local transportation, – Expenditures on memories, – Shopping.	General tourism industry data on visitor expenditure, local or national statistics.

The second step consists in assessing indirect impacts of direct monetary flows. Expenditures of tourists generate additional activities for the local economy by encouraging economic growth in other sectors (Rusu, 2011). This is possible because the economics sectors are interlinked (Horváth and Frechtling, 1999). Hence, monetary flows can circulate in the whole local economy. For example, in addition to direct jobs created by the category accommodation expenditures, there may have indirect jobs creation in local agriculture. Burke et al. (2008) put it that food purchased by visitors may be sourced from local farmers; fuel used for transportation is purchased from local fuel distributors, etc. The magnitude of these indirect impacts depend on the amount of initial expenditures and the degree to which they recirculate in the local economy (Horváth and Frechtling, 1999). The “multiplier effect” is used to estimate the value of these indirect impacts. It measures the impact of extra one monetary unit expenditure introduced into an economy attempting to capture the overall impact of direct tourist expenditures (Burke et al., 2008). Different output-input models are mobilized to estimate the multiplier effect of tourism expenditures at national or regional scale; i.e. the macroeconomic impacts of tourism (e.g. Horváth and Frechtling, 1999; Mathouraparsad and Maurin, 2017). Horváth and Frechtling (1999) review number of multipliers for different regions and countries, including output multipliers, earnings multipliers and employment multipliers. The authors suggest, “the output and earnings multipliers provide a good basis for comparison, as the methodologies are virtually the same across different studies” (p.5).



Table 14. Tourism multipliers in selected European countries ranked by earnings multipliers

Country	Year	Earnings multipliers
United Kingdom	1989	1.73
Ireland	1989	1.72
France (Guadeloupe)	2017	1.30
Greece	1990	1.20
Cyprus	1990	1.14
Italy	1984	0.1
Malta	1992	0.56

Source: Horváth and Frechtling (1999) ; Mathouraparsad and Maurin, 2017

Like all valuation methods, the direct market has its own weaknesses and it is important to be aware of them. According to Brander et al. (2007), this method tends to over-estimate producer surplus. It is difficult to estimate non-labor operating costs, because this necessitate private data and businesses may be reluctant to release it (Burke et al., 2008). To get around this limit, one can rely on expert opinion, regional norms, or industry-wide statistical data with a margin of error. In addition, Rusu (2011) underlines that multiplier analysis requires a detailed database, which can require considerable time and money. However, in the particular context of NBS, the main challenge, when using the direct market method, is distinguishing those expenditures that can be attributed to NBS. Indeed NBS are considered embedded in the socio-ecological system (Eggermont et al., 2015; Andersson et al., 2017; Nesshöver et al., 2017). It is important to only consider in the valuation the expenditures and visitors who come at least in part because of the presence of NBS (Burke et al., 2008). Generally, economic and visitors data, which is readily available, does not analyse the economy in sufficient detail (Rusu, 2011).



Box 7. Costing Nature and the recreational service (Mulligan, 2018)

Relative realised nature and culture-based tourism services indices (RRNBTS and RRCBTS)

Nature-based and culture-based services are tightly linked. Both can be classed as cultural services are often considered together as recreational services. We separate them because of their very different spatial distributions and potentially different economic valuations. Potential recreational services are calculated according to the potential natural attraction of an area (defined according to its conservation priority index, slope steepness, lack of human land use and water quality), with its accessibility to populations. Areas that are high in natural attractiveness and accessible to significant populations and urban centres receive higher potential recreational services values (for both local and international tourism) than low natural attractiveness areas or areas that are less easily accessible. Potential recreational value is first as a supply if potential tourists calculated as population (Landscape, 2007) weighted accessibility from each urban centre outwards. Accessibility is defined using the agglomeration index of Uchida and Nelson (2009). A recreational attractiveness index is calculated as the mean of the Co\$tingNature conservation priority metric, the slope gradient, the lack of human land use and the fraction of water bodies with high water quality. These are calculated outside of urban and densely populated areas only as we are only interested to account for culture-based tourism that links with nature based tourism, which may not be the relevant in the densely populated urban context. Where accessibility to large populations and/or natural attractiveness are high the potential recreation index is higher. As always the final index is expressed 0-1.

Realised recreational services will only be a fraction of the potential services because many potentially good recreational sites will not be realised because of infrastructural, market, development and political or security barriers to tourism. We produce an index of realised recreational services using the online georeferenced photographic database of Panoramio which at the time of analysis (2006) contained more than 5 million georeferenced photographs. Photographs uploaded to this database are considered to represent evidence of high value urban-rural or international tourism having taken place. The Panoramio database has been 'scraped' for the number of georeferenced photos by different users (i.e. the number of tourists having taken photos) per 25km. These are interpolated to the standard Co\$tingNature grids at 1km or 1 hectare resolution. To separate nature and culture based services this realised index is split between areas considered as urban or road for which the realised index is attributed to culture-based tourism vs. areas not urban or road in which the index is attributed to nature-based tourism. For the latter the index is scaled to the maximum of conservation priority, slope steepness and high quality water bodies in order to disaggregate the relatively spatially coarse metric to the areas which generate the interest in photography (i.e. the service) even if those areas are distance from where the photographs are geolocated (i.e. taken). As always the final index is expressed



3.3. Stated-Preference approaches

Stated preference approaches are widely used in Environmental Economics. These approaches are particularly useful because many goods and services arising from environmental projects, programmes are of an intangible nature and are not traded in actual markets. Changes in their quality or quantity cannot therefore be measured using market data although it does not mean they do not have an economic value. Pearce et al. (2006) provide a description of stated preference methods: *“By means of an appropriately designed questionnaire, a hypothetical market is described where the environmental good in question can be traded (Mitchell and Carson, 1989). This contingent market defines the good itself, the institutional context in which it would be provided, and the way it would be financed. A random sample of people is then directly asked to express or reveal, in some way, their maximum willingness to pay (or willingness to accept) for a hypothetical change in the level of provision of the good. “*

Two main stated preference methods are used. The Contingent Valuation (CV) method is probably the most widely used. After its criticism following the Exxon Valdez Oilspill, the NOAA Panel Report on Contingent Valuation (Arrow *et al.*, 1993) provided guidelines that standardized the application of the methodology. In this method, after the presentation of the hypothetical scenario to evaluate and the payment mechanism, respondents are asked questions to determine how much they would value the good if confronted with the opportunity to obtain it.

More recently, the Choice Experiment (CE) approach has been increasingly used (Johnston *et al.*, 2017). In CE, the environmental good is described in terms of its attributes and of the levels that these attributes take. Respondents are presented with various alternative descriptions of a good, differentiated by their attributes and levels, and are asked to rank the various alternatives, to rate them or to choose their most preferred. By including price/cost as one of the attributes of the good, willingness to pay (WTP) can be indirectly recovered from people's rankings, ratings or choices.

There are various differences between the two methods. CE is claimed to overcome some of the biases observed in the CV approach. Without entering into this debate, contingent valuation, typically would be used to uncover the value of the total change in a multi-dimensional good. If policy-makers require measures of the change in each of the dimensions or attributes of the good then some variant of CE might be considered (Pearce *et al.*, 2006).

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The application of stated preference approaches to evaluate NBS co-benefits requires the following elements to be determined for the implementation of the survey:

- **Description of the NBS strategy being tested.** In the context of co-benefits assessment, this step requires a description of the co-benefits generated by the NBS strategy. This description can nevertheless be more qualitative than in direct valuation methods. The reference scenario or status quo also needs to be defined as respondents will make choices relative to this status quo.
- **Description of the context in which decision is made.** This should especially include: i) the institution in charge of implementing the NBS strategy and ii) the timing of implementation of the NBS strategy.
- **Description of the method of payment.** This step is fundamental in stated preference approaches. Many considerations enter in the definition of this payment method, but a simple guideline is to use the one which is likely to be employed to finance the NBS strategy (local taxes...) (Pearce, 2006).

Stated preference approaches present specific characteristics as compared to other valuation approaches. First, stated preference approaches provide an evaluation of the bundle of co-benefits associated with NBS strategies, it may therefore be difficult to disentangle the valuation of specific co-benefits with these methods. Second, stated preference approaches provide an assessment of use and non-use values associated with NBS strategy. If several monetary valuation methods are used, values estimated with direct valuation approaches cannot be added to values estimated with stated preference methods, due to the risk of double counting. Finally stated preference surveys can be an opportunity to assess with the same questionnaire socio-cultural values associated with NBS co-benefits.

3.3.1 Example of a CE survey for NBS co-benefits in NAIAD: the Lez survey. Preferences of residents in the South of France for the conservation of agriculture and natural land and for the development of green infrastructure.

The Lez Watershed is challenged by the coexistence of extreme rainfall events (that can reach up to 300-400 mm per 24h) and rapid urbanization (2920 ha artificialized from 1990 to 2012) due to its rapid population growth, increasing flooding risk. During the last major event in 2014, 78% of the damages were due to flooding caused by surface runoff.

Due to these challenges, it was decided to assess two types of NBS in the Lez DEMO:

- The conservation of agricultural and natural land: three strategies of urban planning are considered with more or less ambitious targets of conservation of agricultural and natural areas.
- The development of green infrastructure: three strategies of green infrastructure are considered based on the use of vegetated retention basins, bioswales and deproofing.



In NAIAD, these NBS strategies are assessed with different models. We present here the methodology developed to make a monetary assessment of the co-benefits of these NBS strategies. It is expected that residents of the Lez basin have preferences for these different strategies due to : i) the co-benefits they generate (air quality, aesthetic amenity, recreational benefits...) and ii) the potential negative externalities they may generate such as the limited possibility to live in individual houses and the limitation of car use they may generate.

In order to determine the preference in monetary terms of residents for these scenarios, a CE is carried out. The CE method gives the opportunity to evaluate preferences of the population for different single NBS types within a global NBS strategy. The CE is implemented through a survey carried out with residents of the Lez DEMO. The main feature of this methodology is that respondents choose between hypothetical alternatives of the Lez Basin, characterized by attributes. The aim of the study is to estimate the preferences for these different attributes in monetary terms.

Attributes of the choice experiment

Attribute 1 covers the different strategies of conservation of natural and agricultural land. In CE terminology, these three strategies are called levels. In brief, these three levels are different ambitions of densification of habitat in the territory for the 2040 time horizon. These different levels are described by the **Figure 18**.

Attribute 1: Conservation of natural and agricultural land	LEVEL 0: continuation of urban sprawling	OU	LEVEL 1: densification of new urban areas	OU	LEVEL 2: densification of new and existing urban areas
	 Artificialisation of 3200 ha		 Artificialisation of 1600ha		 Artificialisation of 800 ha

Figure 18. Description of the first attribute of the choice experiment

Level 0 considers that urban sprawl observed in the past will continue, and will result in transformation of 3200 ha of agricultural and natural areas into urban areas.

Level 1 considers that urban sprawl will be limited with the densification of newly constructed urban areas. It will lead to the transformation of 1600 ha of agricultural and natural areas into urban areas.

Level 2 considers that urban sprawl will be strongly limited, with the densification of newly constructed areas and the requalification of existing urban areas. It will lead to the transformation of 800 ha of agricultural and natural areas into urban areas.

Attribute 2 covers different strategies of development of green infrastructures described in **Figure 19**.



Attribute 2: Green infrastructure	LEVEL 0 	OR LEVEL 1   	OR LEVEL 2     Similaire au niveau 1
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Figure 19. Description of the second attribute of the choice experiment

In level 1 the following actions are implemented:

- ① Deproofing and vegetalization of available public space
- ② Transformation of parkings with permeable pavements
- ③ Bioswales (width 50cm) that do not affect car circulation

In level 2 the following actions are implemented:

- ① Deproofing and vegetalization of available public space
- ② Transformation of parkings with permeable pavements
- ③ Bioswales (width 2m) planted with trees that affect car circulation
- ④ Transformation of 25% of parking lots with vegetated multifunctional retention basins

Attribute 3 is the contribution to local taxes (5, 10, 15, 20, 25 or 30 €/household/year) that respondents are willing to accept for the implementation of these different scenarios. This attribute is fundamental as it is the basis to estimate the monetary value that residents grant to the various NBS strategies.

Experimental design

In the survey, the relative preference (in terms of variation of WTP) between attribute levels, will be inferred from the statistical analysis of choices made by respondents in a number of choice sets. A choice set, is a hypothetical situation characterized by different alternatives among which respondents have to choose (see example in **Figure 20**).



	Alternative A	Alternative B	
Conservation of natural and agricultural land	LEVEL 2: densification of new and existing urban areas  Artificialisation of 800 ha	LEVEL 1: densification of new urban areas  Artificialisation of 1600ha	None of the 2 strategies
Green infrastructure	LEVEL 0 	LEVEL 1 	
Contribution in local tax	30€/Household/yr	10€/Household/yr	
Tick your preferred option ☐ ☐ ☐			

Figure 20. Example of choice set in the Lez survey

The number of choice sets, and the alternatives presented in each choice sets is determined thanks to an experimental design. In this survey, respondents have to respond to 6 choice sets. The experimental design is elaborated with the NGENE software.

Questionnaire

The choice experiment is included in a wider questionnaire that serves different purposes. In the first section of the questionnaire, respondents are requested to identify the different co-benefits and disadvantages that they associate with the different NBS strategies proposed in the questionnaire. Besides the importance to use an integrated valuation approach to evaluate co-benefits (Jacobs *et al.*, 2016), this first step is essential for respondents to consider ecosystem services when providing their preferences in the choice experiment (Loomis *et al.*, 1996). In the second section, respondents make their choices among the 6 choice sets mentioned above. In the last section, a number of key socio-demographic variables of interest are collected.

Sample

The objective of this assessment is to estimate the preferences of the Lez Basin population for different NBS strategies. The questionnaire will therefore be sent to all possible medias (local newspapers, municipalities' website...) in order to obtain responses from a representative sample of the population. A minimum of 300 respondents is targeted in this assessment.



Expected outputs

This assessment is expected to bring the following outputs:

- The assessment will provide the average preference of Lez basin residents for the levels of the attribute in terms of marginal WTP. For example, it will be possible to determine that on average, residents of the Lez Basin, are willing to contribute (or to be paid) X€ if the level 2 of conservation of agricultural and natural land is implemented instead of the level 1. It will also be possible to determine the average value granted by the population to all the combinations of attribute 1 and 2. If the sample is sufficiently representative, this value will be multiplied by the number of residents in the Lez Basin and provide a monetary value to the co-benefits associated with all the combinations of NBS strategies.
- It is also possible to analyse the heterogeneity of preferences. This analysis may give insight to i) understand which category of population is more or less in favour of different NBS solutions ii) which ecosystem services is particularly influential in the value granted by the population to the proposed NBS strategies.

This application of the CE methodology is one of the first application of the methodology to NBS however this method has been widely applied to evaluate ecosystems and ecosystem services (Johnston *et al.*, 2017). The Contingent valuation has however been more widely applied for the monetary assessment of NBS.

3.3.2 Examples of contingent valuation of NBS

Derkzen et al (2017) evaluate residents' preferences for Green Infrastructure (GI) in the city of Rotterdam, Netherlands. This study is carried out in two neighbourhoods with various levels of development of Green Infrastructures. Data were collected through face to face surveys with 200 respondents. In the questionnaire they develop, they first enquire about the perception of heat island and flooding issues. In a second section, residents were asked to rate the different co-benefits of isolated GI: green roof, urban park... and then of neighbourhood's combinations of GI, based on photo representations of GI. Finally, a WTP question was carried out at the end of the questionnaire. Residents were asked to indicate what level of contribution (0, 5, 15€ or 40€) they were willing to pay for new GI. In terms of WTP, the analysis reveals that two thirds of the respondents were willing to pay for GI and 15€ was the mostly chosen amount (34%). It also shows that revealing information on ecosystem services delivered by GI had no impact on residents' WTP. This analysis is a rather limited application of the CV that does not provide information on the total monetary value granted by residents to GI but rather on some factors that influence their WTP such as their level of revenue.

A contingent valuation was implemented by Mell et al. (2013) for street tree investments in the city of Manchester. Face to face interviews were carried out with 512 respondents. Two WTP



questions were developed; the first asked how much respondents were WTP in extra taxes for green investments, whilst the second question posed whether people would be WTP more from existing council tax payments to meet these costs. The second question was asked only if respondents had recorded a zero or 'no payment' value to the first question and was used to address the potential issue of protest zero payments. The WTP elicitation question was framed as a regular monthly council tax payment. The constructed market and the WTP questions were used in conjunction with a set of five 2D visualisations of the proposed investments (**Figure 21**).



Figure 21. 2 D representation of street tree project in Manchester (Mell *et al.*, 2013)

The results indicate:

- that 72% of respondents would pay for some form of additional urban greening;
- that larger or more visible GI investments elicit higher payment values than smaller or less visible GI investments (**Figure 22**).



Figure 22. WTP for different scenarios of street tree investment

The analysis then provides elements on the individual factors that influence WTP (Profession, perception of the environment...).

This article is a good example of how contingent valuation could be used to estimate the value granted by city dwellers to individual NBS measures.

Conclusion and next steps

Nature Based Solutions (NBS) implemented for water risk management may deliver a wide range of environmental, economic and social co-benefits (see **Table 2** for those identified at the DEMO scale). Assessing these co-benefits is particularly important because NBS may not be cost-effective solutions as compared to grey solutions if only primary benefits are considered, but may become cost-effective and socially acceptable if co-benefits are considered. They can therefore represent strong arguments for stakeholders to adopt, accept or finance NBS. In NAIAD, NBS strategies aiming at reducing water-related risks have been developed in the 9 DEMOs. This deliverable provides guidance for the identification and the monetary valuation at the DEMO scale of the main co-benefits associated with NBS.

Chapter 2 presents key existing frameworks (MA, TEEB, CICES, IPBES, EKLIPSE) that can be used to classify the co-benefits of NBS as well as a brief description of existing methods for their identification at the DEMO scale. We classified co-benefits into five main categories. The four first refer to the traditional ecosystem services classification, namely regulating, cultural (or non-



material), provisioning (or material) services, and supporting services which are then grouped in ecosystem services sub-categories. The fifth category “Other” refers to the co-benefits that do not belong to the ecosystem services classification.

Chapter 3 provides guidelines with concrete examples for the monetary valuation of the main co-benefits associated with NBS developed for the objective of reducing water-related risks. It focuses on two main types of methods: direct valuation and stated preference approaches:

- Direct valuation approaches (market price or cost-based methods) are generally easier to implement and to grasp for non-economists. In general, these approaches require biophysical indicators. Each ecosystem service requires specific valuation methods both for biophysical and monetary assessment. The report provides a detailed description of these methods based on a literature review for 5 ecosystem services, with illustrative examples that may be of interest for the NAIAD project: carbon storage and sequestration; local climate regulation; air quality regulation; water cycle regulation; and recreation. However, these methods only capture direct use values: they do not reflect the whole economic value. Result should thus be considered as lower bound estimates.
- Stated preference approaches consist in representative surveys of the population affected by the proposed strategies. Such approaches can be used to value one single ecosystem service or a bundle of services. Non-use values can only be evaluated using these approaches. Two stated preference methods are described with illustrative examples: the Contingent Valuation (CV) and the Choice Experiment (CE) methods. The report provides a description of the CE survey under development for the assessment of NBS co-benefits in NAIAD on the Lez DEMO. It also illustrates the implementation of the CV method with two examples found in the literature.

This report directly contributes to the economic assessment of NBS, by providing guidelines for the assessment of the “co-benefits” component of the cost-benefit analysis methodology developed for the assessment of NBS. The aim of this deliverable is not to provide a novel contribution in terms of methodologies but rather to propose guidelines and concrete examples, for the monetary valuation of NBS co-benefits. Finally, this report focuses only on monetary valuation methods. They should be combined with biophysical (WP2) and socio-cultural (WP3) approaches to bring a whole picture of the values associated with co-benefits, to ensure a value pluralism perspective.



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Annexes



Typology of ecosystem services in TEEB (De Groot et al., 2010)

	Main service types
	PROVISIONING SERVICES
1	Food (e.g. fish, game, fruit)
2	Water (e.g. for drinking, irrigation, cooling)
3	Raw Materials (e.g. fiber, timber, fuel wood, fodder, fertilizer)
4	Genetic resources (e.g. for crop-improvement and medicinal purposes)
5	Medicinal resources (e.g. biochemical products, models & test-organisms)
6	Ornamental resources (e.g. artisan work, decorative plants, pet animals, fashion)
	REGULATING SERVICES
7	Air quality regulation (e.g. capturing (fine)dust, chemicals, etc)
8	Climate regulation (incl. C-sequestration, influence of vegetation on rainfall, etc.)
9	Moderation of extreme events (eg. storm protection and flood prevention)
10	Regulation of water flows (e.g. natural drainage, irrigation and drought prevention)
11	Waste treatment (especially water purification)
12	Erosion prevention
13	Maintenance of soil fertility (incl. soil formation)
14	Pollination
15	Biological control (e.g. seed dispersal, pest and disease control)
	HABITAT SERVICES
16	Maintenance of life cycles of migratory species (incl. nursery service)
17	Maintenance of genetic diversity (especially in gene pool protection)
	CULTURAL & AMENITY SERVICES
18	Aesthetic information
19	Opportunities for recreation & tourism
20	Inspiration for culture, art and design
21	Spiritual experience
22	Information for cognitive development



Examples of EKLIPSE indicators for each challenge area (Raymond et al., 2017)

Raymond, C. M., Frantzeskaki, N., Kabisch, N., Berry, P., Breil, M., Nita, M. R., ... & Calfapietra, C. (2017). A framework for assessing and implementing the co-benefits of nature-based solutions in urban areas. *Environmental Science & Policy*, 77, 15-24.

	Challenge	Type	Indicators
1	Climate mitigation and adaptation		Carbon storage and sequestration in vegetation and soil Monetary values: value of carbon sequestration by trees Temperature reduction Energy and carbon savings from reduced building energy consumption
2	Water management	Physical	Run-off coefficient in relation to precipitation quantities Flood peak reduction, Increase in time to peak Reduction of drought risk Increasing ground water availability, Absorption capacity of green surfaces, bioretention structures and single trees Nutrient abatement, abatement of pollutants Ground water quality Increased evapotranspiration measured/modelled Temperature reduction in urban areas
		Economic	Economic benefit of reduction of stormwater to be treated in public sewerage system Reduction of inundation risk for critical urban infrastructures Stage-damage curves relating depth and velocity of water to material damages
3	Coastal resilience	Physical	Shoreline characteristics and erosion protection Soil, temperature, drainage Flooding characteristics
		Economic	Avoided damage costs Changes in property value
		Social and education	Recreation and public access Number of students benefiting from education and research about coastal resilience/amenity
		Biological	Estimates of species, individuals and habitats distribution Invasive and planted species Algal bloom
		Chemical	Concentration of nutrients Salinity, pH
4	Green space management (including enhancing/ conserving urban biodiversity)		Distribution of public green space Recreational or cultural value Accessibility of urban green spaces for population Changes in the pattern of structural and functional connectivity Species richness and composition in respect to indigenous vegetation and local/national biodiversity targets
5	Air quality	Non-spatial indicators	Annual amount of pollutants captured by vegetation Net air quality improvement Share of emissions (air pollutants) captured/sequestered by vegetation
		Spatial indicators	Pollutant fluxes per m2 per year
		Monetary values	value of air pollution reduction total monetary value of urban forests including air quality, run-off mitigation, energy savings, and increase in property values
		health impact indicators	premature deaths and hospital admissions averted per year
6	Urban regeneration	urban green indicators	Index of biodiversity, provision and demand of ecosystem services. Ecological connectivity Accessibility Ratio of open spaces to built-form. Reclamation of contaminated land
		Building efficiency and environmental design indicators	Reclamation of building materials Energy efficiency Incorporation of environmental design Land devoted to roads



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		Socio-cultural indicators	Conservation of built heritage resources
			Land dedicated to pedestrians
			Public transport links
			Access to open space
			Access to cultural facilities
			Access to housing
			Level of devices contributing to the safety of users in the neighbourhood
7	Participatory planning and governance		Openness of participatory processes
			Legitimacy of knowledge in participatory processes
			Social learning concerning urban ecosystems and their functions/services
			Policy learning concerning adapting policies and strategic plans by integrating ecosystem services and possibly their valuation
			Perceptions of citizens on urban nature
			Social values for urban ecosystems and biodiversity
8	Social Justice and Social Cohesion	Social Justice	The availability and distribution of different types of parks and/or ecosystem services with respect to specific individual or household socioeconomic profiles and landscape design
			Access to financial resources, including indicators of income per capita in a given neighbourhood, or urban area
			Bodily integrity: being able to move freely from place to place; to be secure against violent assault, including indicators of crime by time of day
			Senses, imagination and thought: being able to use the senses, to imagine, think, and reason about the environment, informed by indicators of levels of literacy, mathematics and science knowledge
			Emotions: being able to have attachments to things and people outside ourselves; to love those who love and care for us, including indicators of place attachment, empathy and love
			Being able to participate effectively in political choices that govern one's life, including indicators on level and quality of public participation in environmental management
		Social cohesion	Structural aspects: indicators of family and friendship ties; participation in organised associations; integration into the wider community
			Cognitive aspects: indicators of trust, attachment to neighbourhood, practical help, tolerance and respect
9	Public health and well-being	Psychological indicators	(= Relaxation and restoration, sense of place, exploratory behaviour, socializing).
			Reduction in chronic stress and stress-related diseases measured through repeated salivary cortisol sampling and hair cortisol ; use cortisol slope and average cortisol levels as an indicator of chronic stress.
			Cognitive and social development in children: indicators related to improvement in behavioural development and symptoms of attention deficit/hyperactivity disorder (ADHD) related to green space use; questionnaire indicators on sociodemographic and household characteristics, the time spent playing in green and blue spaces, ADHD symptom criteria, such as emotional symptoms, inattention, conduct problems, hyperactivity/inattention, and peer relationship problems; and a strengths subscale for prosocial behaviour.
			Mental health changes measured through Mental Well-being scales asking participants how they have felt over the previous four weeks in relation to a number of items (e.g., feeling relaxed, feeling useful), with responses rated on a 5-point scale from "none of the time" to "all of the time".
		Health indicators related to physical activity	(= Sports and leisure activities including e.g. walking, cycling).
			Number and share of people being physically active (min. 30 min 3 times per week).
			Reduced percentage of obese people and children; reduced overall mortality and increased lifespan.
			Reduced number of cardiovascular morbidity and mortality events
		Health indicators related to ecosystem service provision	(Buffering of noise and air pollution, reduced heat, exposure to microflora)
			Reduced autoimmune diseases and allergies (potentially)
			Reduced cardiovascular morbidity and mortality
			GIS related indicators: NDVI, proximity measures (green space of min. 2 ha within 300m, percentage of green space
10	Potential for Economic Opportunities and Green Jobs		Number of subsidies or tax reductions applied for (private) NBS measures
			Number of jobs created; gross value added
			Change in mean or median land and property prices
			New businesses attracted and additional business rates
			Resource efficiency in the urban system (CO2 emissions per capita, CO2 emissions for transportation per capita)
			Public-sector cost per net additional job
			Net additional positive outcomes into employment
			Net additional jobs in the green sector enabled by NBS projects.
			Gross value added per employees based on full-time equivalent jobs in the green sector.



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			Production benefit: earnings uplift arising from skills enhancement in the design and implementation of NBS.
			Consumption benefits: property betterment and visual amenity enhancement resulting from NBS.

CICES classification (Haines-Young and Potschin, 2018)

Haines-Young R, Potschin MB (2018). Common International Classification of Ecosystem Services (CICES) V5.1 and Guidance on the Application of the Revised Structure. EEA. www.cices.eu

Section	Division	Group	Class (or simple descriptor if too complex)		
Regulation & maintenance	Transformation of biochemical or physical inputs to ecosystems	Mediation of wastes or toxic substances of anthropogenic origin by living / non-living processes	Waste bioremediation		
			Waste filtration/ storage		
			Dilution by freshwater and marine ecosystems		
			Dilution by atmosphere		
			Natural processing of wastes		
		Mediation of nuisances of anthropogenic origin	Smell reduction		
			Noise attenuation		
			Visual screening (of unsightly things)		
			Regulation of physical, chemical, biological conditions	Regulation of baseline flows and extreme events	Control of erosion rates
					Buffering and attenuation of mass movement (landslides/ avalanches)
	Hydrological cycle and water flow regulation (Including flood control, and coastal protection)				
	Wind protection				
	Fire protection				
	Lifecycle maintenance, habitat and gene pool protection	Pollination (fruit trees and other plants)			
		Seed dispersal (wild plants)			
		Maintaining nursery populations and habitats			
		Regulating living conditions by the physical environment			
	Pest and disease control	Pest (and invasive species) control			
		Disease control			
	Regulation of soil quality	Weathering processes (ensuring soils form and develop)			
Decomposition and fixing processes (ensuring the organic matter in our soils is maintained)					
Water conditions		Regulation of the chemical quality of freshwater			
		Regulation of the chemical quality of salt water			
Atmospheric composition and conditions		Regulation of chemical composition of atmosphere and oceans			
		Regulation of temperature and humidity, including ventilation and transpiration			
Cultural	Direct, in-situ and outdoor interactions with living / natural physical systems that depend on presence in the environmental setting	Physical and experiential interactions with nat. environment	Using the environment for sport and recreation; using nature to help stay fit		
			Watching plants and animals where they live; using nature to destress		
		Intellectual and representative interactions with natural environment	Researching nature (scientific investigation or creation of traditional ecological knowledge)		
			Studying nature (Education and training)		
			Culture or heritage		
			Aesthetic experiences		
		Indirect, remote, often indoor interactions with living/ natural physical systems that do not require presence in the environmental setting	Spiritual, symbolic and other interactions with natural environment	Symbolic meaning (national or local emblem)	
				Sacred or religious meaning (spiritual importance)	
				Entertainment or representation (films, books)	
			Other biotic characteristics that have a non-use value	Existence value (the things in nature that we think should be conserved)	
Option or bequest value (the things in nature that we want future generations to enjoy or use)					
Provisioning	Biomass			Cultivated terrestrial plants for nutrition, materials or energy	Cultivated terrestrial plants for food
		Fibres and other materials from cultivated plants			
		Cultivated plants grown as a source of energy			
		Cultivated aquatic plants for nutrition, materials or energy	Plants cultivated by in- situ aquaculture for food		
			Fibres and other materials from in-situ aquaculture		
			Plants cultivated by in- situ aquaculture grown as an energy source		
		Reared animals for nutrition, materials or energy	Livestock raised in housing and/or grazed outdoors		
			Material from animals that we can use		
			Animal materials used as a source of energy or for traction		
		Reared aquatic animals for nutrition, materials or energy	Animals that are cultivated in fresh or salt water that we eat.		
			Animals that are cultivated in fresh or salt water that we can use as a material.		
			Animals that are cultivated in fresh or salt water that we can use as a source of energy.		
		Wild plants (terrestrial and aquatic) for nutrition, materials or energy	Food from wild plants		
			Materials from wild plants		



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		Wild animals (terrestrial and aquatic) for nutrition, materials or energy	Materials from wild plants, fungi and algae used for energy
			Food from wild animals
			Materials from wild animals
			Material from wild animals that can be used as a source of energy
	Genetic material from all biota (including seed, spore or gamete production)	Genetic material from plants, algae or fungi	Seed collection
			Plants, fungi or algae that we can use for breeding
			Genetic material from wild plants, fungi or algae that we can use
		Genetic material from animals	Animals used for replenishing stock
			Wild animals that we can use for breeding
			The genetic information that is stored in wild animals that we can use
	Water	Surface water used for nutrition, materials or energy	Surface water for drinking
			Surface water used as a material (non-drinking purposes)
			Freshwater surface water used as an energy source
			Coastal and marine water used as energy source
		Ground water for used for nutrition, materials or energy	Ground (and subsurface) water for drinking
			Ground water (and subsurface) used as a material (non-drinking purposes)
	Non-aqueous natural abiotic ecosystem outputs	Mineral substances used for nutrition, materials or energy	Ground water (and subsurface) used as an energy source
			Minerals in our food
			Natural inorganic materials from nature that we can use
		Non-mineral substances or ecosystem properties used for nutrition, materials or energy	Natural inorganic materials from nature that we can use as an energy source
			Non-mineral substances or ecosystem properties used for nutritional purposes
			Non-mineral substances used for materials
			Wind energy
			Solar energy
			Geothermal



Reporting categories of nature's contributions to people (NCP) used in IPBES assessments (Diaz et al, 2018)

Díaz et al. (2018) "Assessing nature's contributions to people." *Science* 359.6373: 270-272.

	Reporting categories of nature's contributions to people	Brief explanation and some examples
1	Habitat creation and maintenance	The formation and continued production, by ecosystems or organisms within them, of ecological conditions necessary or favorable for living beings of direct or indirect importance to humans.
2	Pollination and dispersal of seeds and other propagules	Facilitation by animals of movement of pollen among flowers, and dispersal of seeds, larvae or spores of organisms beneficial or harmful to humans
3	Regulation of air quality	Regulation (by impediment or facilitation) by ecosystems, of CO ₂ /O ₂ balance, O ₃ , sulphur oxide, nitrogen oxides (NO _x), volatile organic compounds (VOC), particulates, aerosols, allergens Filtration, fixation, degradation or storage of pollutants that directly affect human health or infrastructure
4	Regulation of climate	Climate regulation by ecosystems (including regulation of global warming) through: • Positive or negative effects on emissions of greenhouse gases (e.g. biological carbon storage and sequestration; methane emissions from wetlands) • Positive or negative effects on biophysical feedbacks from vegetation cover to atmosphere, such as those involving albedo, surface roughness, long-wave radiation, evapotranspiration (including moisture-recycling) and cloud formation • Direct and indirect processes involving biogenic volatile organic compounds (BVOC), and regulation of aerosols and aerosol precursors by terrestrial plants and phytoplankton
5	Regulation of ocean acidification	Regulation, by photosynthetic organisms (on land or in water), of atmospheric CO ₂ concentrations and so seawater pH, which affects associated calcification processes by many marine organisms important to humans (such as corals)
6	Regulation of freshwater quantity, location and timing	Regulation, by ecosystems, of the quantity, location and timing of the flow of surface and groundwater used for drinking, irrigation, transport, hydropower, and as the support of non-material contributions (NCP 15, 16, 17) Regulation of flow to water-dependent natural habitats that in turn positively or negatively affect people downstream, including via flooding (wetlands including ponds, rivers, lakes, swamps) Modification of groundwater levels, which can ameliorate dryland salinization in unirrigated landscapes
7	Regulation of freshwater and coastal water quality	Regulation – through filtration of particles, pathogens, excess nutrients, and other chemicals – by ecosystems or particular organisms, of the quality of water used directly (e.g. drinking, swimming) or indirectly (e.g. aquatic foods, irrigated food and fiber crops, freshwater and coastal habitats of heritage value)
8	Formation, protection and decontamination of soils and sediments	Formation and long-term maintenance of soil structure and processes by plants and soil organisms. Includes: physical protection of soil and sediments from erosion, and supply of organic matter and nutrients by vegetation; processes that underlie the continued fertility of soils important to humans (e.g. decomposition and nutrient cycling) ; filtration, fixation, attenuation or storage of chemical and biological pollutants (pathogens, toxics, excess nutrients) in soils and sediments
9	Regulation of hazards and extreme events	Amelioration, by ecosystems, of the impacts on humans or their infrastructure caused by e.g. floods, wind, storms, hurricanes, heat waves, tsunamis, high noise levels, fires, seawater intrusion, tidal waves Reduction or increase, by ecosystems or particular organisms, of hazards like landslides, avalanches
10	Regulation of detrimental organisms and biological processes	Regulation, by organisms, of pests, pathogens, predators or competitors that affect humans (materially and nonmaterially), or plants or animals of importance for humans. Also the direct detrimental effect of organisms on humans or their plants, animals or infrastructure.
11	Energy	Production of biomass-based fuels, such as biofuel crops, animal waste, fuelwood, agricultural residue pellets, peat
12	Food and feed	Production of food from wild , managed, or domesticated organisms, such as fish, bushmeat and edible invertebrates, beef, poultry, game, dairy products, edible crops, wild plants, mushrooms, honey Production of feed (forage and fodder) for domesticated animals (e.g. livestock, work and support animals, pets) or for aquaculture, from the same sources
13	Materials, companionship and labor	Production of materials derived from organisms in cultivated or wild ecosystems, for construction, clothing, printing, ornamental purposes (e.g. wood, peat, fibers, waxes, paper, resins, dyes, pearls, shells, coral branches) Live organisms being directly used for decoration (i.e. ornamental plants, birds, fish in households and public spaces), company (e.g. pets), transport, and labor (including herding, searching, guidance, guarding)
14	Medicinal, biochemical and genetic resources	Production of materials derived from organisms (plants, animals, fungi, microbes) used for medicinal, veterinary and pharmacological (e.g. poisonous, psychoactive) purposes. Production of genes and genetic information used for plant and animal breeding and biotechnology
15	Learning and inspiration	Provision, by landscapes, seascapes, habitats or organisms, of opportunities for the development of the capabilities that allow humans to prosper through education, acquisition of knowledge and development of skills for well-being, information, and inspiration for art and technological design (e.g. biomimicry)
16	Physical and psychological experiences	Provision, by landscapes, seascapes, habitats or organisms, of opportunities for physically and psychologically beneficial activities, healing, relaxation, recreation, leisure, tourism and aesthetic enjoyment based on the close contact with nature (e.g. hiking, recreational hunting and fishing, birdwatching, snorkeling, diving, gardening)
17	Supporting identities	Landscapes, seascapes, habitats or organisms being the basis for religious, spiritual, and social-cohesion experiences: • Provisioning of opportunities by nature for people to develop a sense of place, belonging, rootedness or connectedness, associated with different entities of the living world (e. g. cultural, sacred and heritage landscapes, sounds, scents and sights associated with childhood experiences, iconic animals, trees or flowers) • Basis for narratives, rituals and celebrations provided by landscapes, seascapes, habitats, species or organisms • Source of satisfaction derived from knowing that a particular landscape, seascape, habitat or species exists
18	Maintenance of options	Capacity of ecosystems, habitats, species or genotypes to keep options open in order to support a good quality of life. Examples include: • Benefits (including those of future generations) associated with the continued existence of a wide variety of species, populations and genotypes. This includes their contributions to the resilience and resistance of ecosystem properties in the face of environmental change and variability • Future benefits (or threats) derived from keeping options open for yet unknown discoveries and unanticipated uses of particular organisms or ecosystems that already exist (e.g. new medicines or materials) • Future benefits (or threats) that may be anticipated from ongoing biological evolution (e.g. adaptation to a warmer climate, to emergent diseases, development of resistance to antibiotics and other control agents by pathogens and weeds)



Examples and brief description of ecosystem services valuation methods (non-exhaustive list) (Jacobs et al., 2018)

Jacobs, S., Martín-López, B., Barton, D. N., Dunford, R., Harrison, P. A., Kelemen, E., ... & Kopperoinen, L. (2018). The means determine the end—Pursuing integrated valuation in practice. *Ecosystem Services*, 29, 515-528.

Biophysical valuation methods	MapNat App	The MapNat application enables users to map the ecosystem services and ecosystem disservices they are using or experiencing. MapNat can be interpreted as a form of participatory GIS, as it employs either online maps on which users map the position and type of ecosystem service use, rank the importance and frequency of use and comment, or it makes use of the GPS functionality of the smartphone for mapping. In addition to contributing to increase scientific knowledge, users have access to the records and comments of all other users via a server e.g. to identify spots of interest.
	Spreadsheet-type methods	Spreadsheet-type methods (also known as matrix methods) are a quick and simple way to get an overall spatially-explicit picture of the ecosystem service in case study areas. The method is based on the idea of linking tabular spreadsheet data and spatial data together. The spreadsheet format data can be collected, for example, as expert evaluation or construct from indicators or statistics. A modified, transdisciplinary version of spreadsheet-type methods is GreenFrame, which makes benefit of an extensive set of spatial datasets grouped into themes (instead of using solely land use and land cover data) combined with both scientific experts' and local actors' scorings.
	State and transition models	State-and-transition models (STMs) are conceptual models of ecosystem dynamics after disturbances based on alternate state theory. In contrast to succession theory, which predicts that ecosystems recover from disturbances and return to a reference (undisturbed) state, alternate state theory maintains that disturbances may trigger a regime shift in critical processes (e.g. population recruitment, nutrient cycling) that will maintain the ecosystem in a state that differs from the reference state. In this sense, STMs can be understood as a tool to operationalise, gain a common understanding of, and communicate the importance of ecological functions and processes that underpin the provision of ecosystem services.
Socio-cultural valuation methods	Cards game method	The card game is a method developed to capture the sociocultural dimension of values of ecosystem services. During the card game, the interviewee is asked to score ecosystem services cards in function of importance or desirability. It serves a double purpose: on one hand, a quantitative ranking of ecosystem services is obtained; on the other hand, it encourages interviewees to explain why an ecosystem service is (not) important/desirable to him/her.
	Narrative method	Narrative methods aim to understand and describe the importance of nature and its benefits to people with their own words. By using narrative methods we allow the research participants to articulate the plural and heterogeneous values of nature through their own stories (both verbally and visually). Narrative methods usually collect qualitative data from individuals, but they can be also suitable to measure some aspects of human-nature relations in quantitative or semi-quantitative terms.
	Participatory mapping method	Participatory mapping of ecosystem services consists of assessing the spatial distribution of ecosystem services according to the perceptions and knowledge of different stakeholders. In this sense, it allows the integration of perceptions, knowledge (local-based or technical) and values of different stakeholders and presents the outputs in the form of a map of ecosystem services.
	Photo-elicitation survey	This method aims to translate the people's visual experiences and perceptions of landscapes in terms of ecosystem services. It has been particularly used to explore how landscape multifunctionality (defined as the capacity of ecosystems to provide ecosystem services to society) is related with public perceptions toward landscapes and ecosystem services. This is based on the idea that visual stimuli could be understood as a socially shared communication channel, and then, with potential to identify and analyse social perceptions of ecosystem services.
	Photo-series analysis	Photo-series analysis can be used to revealed social preferences towards cultural ecosystem services and spatially explicit determine the locations where these services are actually provided and used. Photo-series databases are obtained from photosharing websites such as Flickr, Panoramio and Instagram. The analysis of community contributed photos can be used as a complementary technique of interviews, questionnaires or focus groups to assess preferences of cultural ecosystem services, assuming that visitors are attracted by the location where they take photographs. The method allows to identify the cultural ecosystem services which are perceived as most important by people who take the picture and to map their distribution.
	Preference assessment survey	Preference assessment surveys aim to analyze perceptions, knowledge and associated value of ecosystem services demand or use (or even social motivations for maintaining the service) without using economic metrics. It could also be used to understand which ecosystem services are perceived as the most vulnerable, or which make the greatest contribution to human wellbeing. Data is collected through surveys using a consultative approach with different variations, such as free-listing exercises, ecosystem service ranking, rating or ecosystem service selection. It is generally used with an emphasis on individual perceptions (but collective preferences could be also gathered).
	Time use method	Time use study is an innovation of the conventional stated preference techniques, in particular taken from the contingent valuation approach. In this case, the payment vehicle is expressed in labour hours rather than monetary units (use in the classical willingness to pay studies, see below stated preference methods). Willingness to give up time (WTT) creates a hypothetical scenario to estimate the value of ecosystem services by directly asking people how much time they would willing to invest for a change in the quantity or quality of a given ecosystem service or conservation (or restoration) plan. Besides being an appropriate approach in scenarios where people can invest time for particular activities related with nature; this approach is also useful in areas with income constraints.
Monetary valuation methods	Benefit/value transfer	Benefits transfer (BT), or more generally - value transfer (VT) - refers to applying quantitative estimates of ecosystem service values from existing studies to another context.
	Cost-based methods	Cost-based valuation methods are a group of 'exchange-based' techniques that use the cost of actual measures to maintain ecosystem service provision as a proxy for the value of avoiding, mitigating or restoring the loss of services ecosystems provide.
	Hedonic pricing	Hedonic pricing is the study of multi-correlation between environmental characteristics of a good and its sales price. The Hedonic property pricing (HPP) method can be used to estimate monetary values for ecosystem services that directly affect 'amenities' of properties which in turn are reflected in property prices. The HPP method requires large data sets of property sales statistics with physical characteristics of the property itself - and particularly for ecosystem services - e.g. characteristics of green infrastructure in the neighborhood.
	Production function method	The production function approach relate the output of marketed goods to the inputs of ecosystem services through the use of econometric techniques. The challenges involved in its application are that data on the relationships between the services (regulation and provision services) and on other non-environmental inputs are often difficult to obtain.
	Shadow pricing method	The shadow price is the marginal price society 'puts' on the provision of non-marketed ecosystem services through setting environmental targets. The costs incurred to reach the agreed level of a particular environmental target can be used as data reflecting the implicit valuation of the ecosystem service(s) involved. The shadow price is the cost of obtaining an additional unit of ecosystem service obtained by



		implementing the environmental target which restricts production of marketed goods. The shadow price society ‘pays’ for provision of the ecosystem services is the loss from not obtaining the value from producing marketed goods.
	Stated preference method	Stated preference valuation is a family of techniques which use individual respondents’ statements about their preferences to estimate change in utility associated with a proposed increase in quality or quantity of an ecosystem service or bundle of services. Respondents are presented one or more hypothetical policy or project scenarios describing a project or policy that will lead to a specified environmental change compared to a baseline situation. The answers respondents give, in the form of monetary amounts, ratings, or other indications of preference, are scaled following an appropriate model of preference to yield a measure of value of the proposed ecosystem service change. This value is often monetary in the form of people’s willingness to pay (WTP).
	Travel cost method	The travel cost methods is based on the observation that recreational services can only be realised through physical access to nature. This implies that individuals seeking to enjoy the ecosystem service will need to spend resources (time and money) to travel to the site. The travel activity is a reflection of the use value this service has to people.
Synthesising valuation methods	Bayesian belief networks	Bayesian Belief Networks visualise the modelled system with nodes representing information and links representing dependencies. The links are defined by probabilities so assessment of supporting evidence (measured by uncertainty) is inherent. Complex networks combining ecological and socio-economic data show how a variety of pressures affect multiple ecosystem services values and allow the balancing of a plural value decision process.
	Deliberative valuation	Deliberative valuation is based on the assumption that valuation is a social process in which values are discovered, constructed and reflected in a dialogue with others. Therefore, deliberative valuation invites stakeholders and citizens (the general public) to form their preferences for ecosystem services together through an open dialogue, which allows consideration of ethical beliefs, moral commitments and social norms beyond individual and collective utility.
	Multicriteria decision analysis	Multicriteria decision analysis (MCDA) is an “umbrella term to describe a collection of formal approaches which seek to take explicit account of multiple criteria in helping individuals or groups explore decisions that matter” (Belton and Stewart, 2002). The basic idea of MCDA methods is to evaluate the performance of alternative courses of action (e.g. management or policy options) with respect to criteria that capture the key dimensions of the decision-making problem (e.g. ecological, economic and social sustainability), involving human judgment and preferences.
	Scenario planning method	Scenarios are defined as “a plausible, simplified description of how the future may develop, based on a coherent and internally consistent set of assumptions about key driving forces” (Priess and Hauck, 2015). Scenarios can be developed by the help of expert input or wider public participation, and can take various shapes, including qualitative or quantitative scenarios, exploratory or anticipatory scenarios, and baseline or policy scenarios. In scenario planning various tools and techniques are applied (often in combination) to develop plausible and internally consistent descriptions of alternative future options. Although scenario planning is not a de facto valuation tool, scenarios can be used to explore how ecosystem services might change in the future and how these changes can influence human well-being. Therefore by comparing and evaluating scenarios we can also reveal the value of related ecosystem services.