"NAture Insurance value: Assessment and Demonstration"

NAIAD

SC5-09-2016 Operationalising insurance value of ecosystems

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Deliverable 4.1

General framework for the economic assessment of Nature Based Solutions and their insurance value

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1. Edition information

Tableau 1 : Versions

Version	Edition date	Modification	Written by	Controlled by
1	20-09-2018	First edition	N. Graveline	
2	14/11/2017	First complete draft	all Authors	P. van de Keur, M. Manzano, A. Scriescu, J. weinberg, C. Herivaux
3	04/12/2017	Final	N. Graveline	

2. Summary

3. Introduction

3.1. Problem setting

Increasing extreme natural hazards and exposure call for a renewed attention to disaster risk management and specifically the issue of resilience. Nature Based Solutions (NBS) have been advocated as an alternative or complementary approach to traditional grey infrastructure, to manage water related risks, enhance resilience, provide additional co-benefits and increase environmental awareness. NBS are cost-effective options to reduce risk (and damage costs) that simultaneously provide environmental, social and economic benefits and reinforce resilience (European Commission, 2016). It is assumed that NBS provide additional benefits and resilience in combination with grey infrastructure options, or at equivalent levels for some attributes. NBS are also anticipated to be flexible, which is highly valuable in the face of uncertainty. They might be considered as no regret strategies or even having security margins[1]. NBS, at least as a concept, are relatively new and real case demonstrations at various spatial scales across Europe that also explore the risks of these solutions such as transfer of vulnerabilities will be realized in NAIAD.

According IUCN (2016), NBS involve three types of actions, which may be combined at regional and local level: (i) Preserving the integrity and good ecological status of ecosystems, (ii) Improving sustainable management of ecosystems used by human activities; (iii) Restoring degraded ecosystems or creating ecosystems. The concept of **Nature Assurance Schemes (NAS) has been introduced in NAIAD to characterize a basin or (peri) urban, long-term strategy that integrates NBS (it can be a mix of NBS and grey infrastructure)**.

In NAIAD, NBS will be considered in demonstration sites (DEMOs) either in scenarios where they can further reduce risk compared to a business as usual (BAU) scenarios or, where they are substitutes or additions to grey infrastructure (that could be deconstructed) while providing at least the same level of risk reduction. Indeed, we assume that in all DEMOs there is still room for improving risk reduction. NBS can either reduce hazard (e.g. by retention of water) or exposure, and potentially both in some cases.

Real case demonstration can be either ex-post analysis in cases where NBS have been implemented or ex-ante analysis in cases where we can consider the NBS options to be efficient. There is a need for economic analysis to provide economic information to the debate. The ambition is to understand under which conditions NBS contribute to improved risk management and more resilient risk management and how to evaluate their costs and benefits over time in comparison to alternative measures, including fully grey infrastructure development. It aims to identify means to assess the value of multiple benefits to society provided by an NBS over time to better evaluate the cost benefit of public investment in them (particularly when short term costs are high and impacts less immediate).

The questions that economics can contribute to relate to:

1. Is it worth investing in NBS strategies from a public/social point of view?
2. What is the optimal setting of the strategy?
3. Are there additional insurance values (i.e. damage avoided) of NBS/NAS compared to (only) grey infrastructure risk management strategies?
4. Do NBSs enable operation and maintenance costs savings?

[1] A security margins means that the project dimension or technical set up can be adapted after the realization of it. (see e.g. Hallegatte, 2009)

5. Are the transaction costs of implementing NBS higher compared to (only) grey/classic risk management strategies?
6. What are the positive or negative externalities? Are the co-benefits significant?
7. What are the distribution of costs and the benefits (market and non-market) among people, sectors, territories, time? Who are those that loose and that gain?
8. Who will invest, contribute, be willing, or oppose?

3.2. Objective of the note

The main objective of this note is to provide a **general methodological framework** for the assessment of NBS/NAS in the DEMO settings. This note should establish the basis for the other tasks in WP4, and provide the big picture for the economic analysis. It means defining the major cost/benefit components as well as terminology and vision of NAIAD and present the issues that are of importance for an economic analysis. It should also serve as a guide for DEMOs in NAIAD which will implement an economic analysis. Even if the methods will be different a similar approach to the appraisal is necessary as the ambition is to realize a synthesis on all the economic assessment realized (Deliverable 4.5).

Notably, it is assumed that there is no universal economic analysis method that can be applied in each DEMO. As each context is local, methods applied will depend on (1) stakeholders frame of interests (which sets of indicators and questions are relevant from their perspective), (2) regulatory structure, (3) resources available to perform the analysis, (4) data availability and expertise, (5) biophysical context (6) existing infrastructure, (7) previous approaches to managing risk and resilience.

NAIAD aims to operationalize the concept of insurance value of ecosystems, thus the implications and ways to use and handle the economic analysis in practice is also discussed in this report.

The scope has been slightly enlarged compared to the initial title that mentions only the insurance value of ecosystems. This allows for the presentation in a coherent frame of all economic values that could be of interest for NAIAD. They are different perspectives of these values (society, individuals, insurance and sectors) that can be adopted in the economic assessment, which will be reviewed here.

3.3. Positioning in NAIAD

WP4 has a central role in NAIAD as it will help assess the economic rationale for NBS and NAS. One of the challenges is to integrate this analysis with other disciplines and activities in NAIAD. A strong link must be made, within the DEMOs so as to connect the economic analysis to the stakeholders by mobilizing them to infer the values/benefits to be considered and reporting the results of the analysis to make the economic analysis useful for debate, discussion and decision making.

By providing methods it should help the DEMOs realize more or less (according to the resources they can allocate to this task) in depth economic analysis of NAS in given DEMOs. As the insurance value is a central concept of NAIAD, the economic thinking should at least be developed on a qualitative level, but ideally also at a quantitative level in all DEMOs.

WP4 and the economic assessment will benefit from connections and benefit other works and disciplines in particular:

- WP2 which will provide the characteristics of hazards and events that will enable to assess alternative damages and provide data for the ecosystem services assessment and valuation.

Ecosystem Services are the direct and indirect contributions of ecosystems to human well-being (TEEB, 2010). The ecosystem services approach enables to account for the multiple services that environment or natural features provide globally. The ecosystem services are generally classified in (i) provisioning, (ii) regulating, (iii) cultural and (iv) supporting services, their valuation is limited to the first three types as supporting services contributes to the three first from a human point of view. They integration in the economic assessment is detailed in 4.3.

- WP3 which will concentrate on risk perceptions and social networks. Both will have a strong effect on behaviour and valuation. Exchanges between WP3 and 4 will help improve the robustness and relevance of the economic analysis and assessment.
- WP5 with system dynamic analysis and potentially :
    - A discussion on the articulation of economic values (monetary benefits) and others dimensions (multi-criteria analysis)
    - A discussion on the concrete use of the insurance value of NBS in risk (and environmental) governance (that include collective action problems, etc.).
- Provide methods to WP6 to be tailored to and assessed in the DEMOs.
- WP7 by informing how cost and benefits are distributed and thus provide insights on the willingness to pay of different agents and potential project shareholders, how transfers of values should be organized; e.g. inform about business models and economic or financial instruments.
- WP8: should help WP4 account for different policy context, and provide WP8 with results (through the DEMOs) of economic assessment from the public / policy point of view and help feed policy briefs.
- It will produce information and material for WP9 (dissemination).

4. Insurance value of NBS

4.1. Introduction

The insurance value as a concept is increasingly used by research and science support to policy in environmental management. However, the common noun of "insurance" often refers to the financial instrument aspect, rather than to the risk reduction feature (definition 2 in the box below). NAIAD's definition of insurance value includes the latter. Indeed, the insurance value from a collective, economic point of view does not have the same meaning of that of an insurance company, for which the value will be restricted to the insured value. In NAIAD, all different types of damages (see 6.3) will be considered and integrated in the notion of insurance value; which thus go further than the market insurance value.

Definition of INSURANCE (*Collins English Dictionary*)

1.The act, system, or business of providing financial protection for property, life, health, etc. against specified contingencies, such as death, loss, or damage, and involving payment of regular premiums in return for a policy guaranteeing such protection.

- The state of having such protection
- Also called: insurance policy the policy providing such protection
- The pecuniary amount of such protection
- The premium payable in return for such protection

2.A means of protecting or safeguarding against risk or injury

The concept of insurance value for water related risk management aims to raise awareness on the risk reduction potential that ecosystems provide against a number of hazards. It is the opposite concept of the cost of risk. As an illustration, we can state that waterproofing and developing cities in risk prone areas have been risky and this can be estimated (now ex-ante) with the concept of **cost of risks**. At the opposite, or having another perspective, safeguarding or restoring ecosystems with a role in water regulation e.g. NBS have an insurance value. WP4 adopts an economic perspective of the insurance value of NBS, which is discussed here.

The insurance value is not included in the well-known classification of Total Economic Values (TEV) framework. In the TEV framework, the insurance value can be attributed (partly at least) to the quasi - option values[2] which are values that might appear in the future in a context of uncertainty. As such it is not specific to any type of value (or any type of ecosystem service), but reflects the capacity of the ecosystem to provide services in the face of risk and uncertainty.

4.2. Background on insurance value in the literature

Insurance value emerged recently in the ecological economics literature with the objective of formalizing the resilience potential of ecosystems while developing ecological-economic models (e.g. Baumgartner (2007); Baumgartner & Quaas (2010), Augeraud-Véron et al. (2017)). For example, Quaas & Baumgardner (2008) developed a model in which the increase in the level of biodiversity monotonically decreases the temporal and spatial variability of the level at which these ecosystem services are provided under changing environmental conditions.

Our first interpretation of the insurance value is based on this ecological economic literature and relates to the **resilience** capacity of ecosystems as an insurance device. It is consistent with Pascual et al. (2010) and the assumption that a biodiverse ecosystem is more capable of buffering shocks, and in economic terms, that the biodiverse ecosystem might reduce the volatility of income (or ecosystem services production) compared to the next best alternative. This idea is represented on **Figure 1** (we add the distinction between shocks e.g. hazard and stresses e.g. global changes).

Similarly to a financial portfolio, a diverse - species – ecosystem can have a regulation capacity in the face of a damage event. The stabilization effect of a diverse portfolio can be more and more valuable in the face of climate change and increasing human pressures. For example, Augeraud et al. (2017) considers biodiversity as an insurance device for agricultural production, dampening production volatility. It suggests that the processes behind ecosystem functioning enables a guaranteed continued flow of services as long as the ecosystem stays within a given regime. This perspective triggers progressive stresses such as climate change or human pressures that induce biodiversity degradation as well as shocks. Pascual et al. (2010)[3] argues that this insurance value plus the output value (i.e. value attached to direct ecosystem's services and benefits) equals the Total Economic Value of the ecosystem. This is similar to a quasi-option value interpretation of the insurance value of ecosystem.

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[2] Option value refer to the future use of known and unknown benefits (Pascual et al. (2010) which is TEEB, Chap. 5, 2010)

[3] (see Chapter 5, Figure 2).

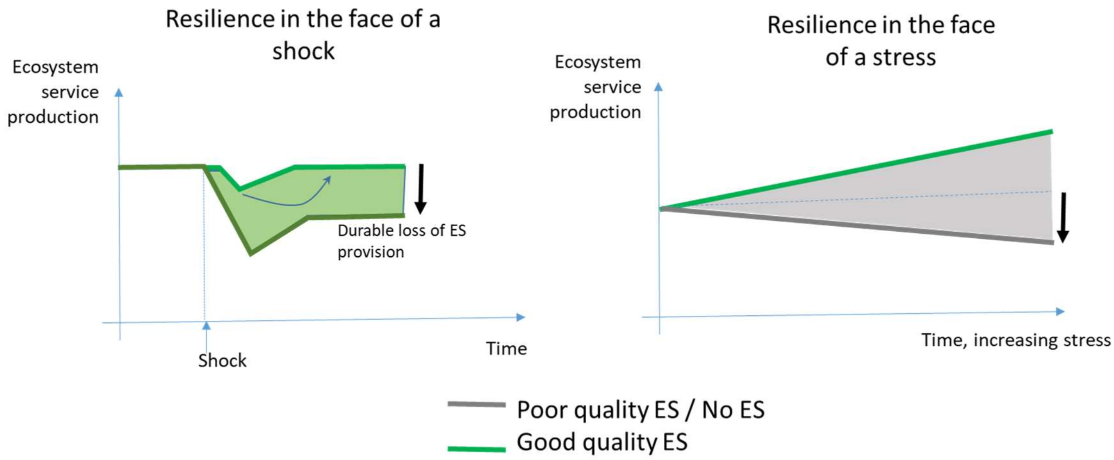

Figure 1 Illustration of the insurance value of ES linked to their resilience capacity. (Source: own elaboration)

For our second interpretation, the insurance value can be conceptualized from the domain of **disaster risk** reduction. Natural hazards characterized as **shocks** will threaten the system under study (territory, city or basin). Any relevant risk reduction strategy has an insurance value. In disaster risk management, the insurance value of a strategy or infrastructure will be the difference in the damage with or without (reference situation), as shown on Figure 2. Here the insurance value of NBS is directly interpreted as a risk reduction potential compared to traditional strategies and baseline scenarios. The term "insurance value" is not really used in the disaster risk literature, with the notion of 'cost of risk' preferred; the term insurance is limited to discussing those risks that are covered by market insurance[4].

[4] Note also that within the insurance sector the idea of "insurable" risks refers to a risks that can be estimated e.g. for which both the insured value and the probability of occurrence can be estimated.

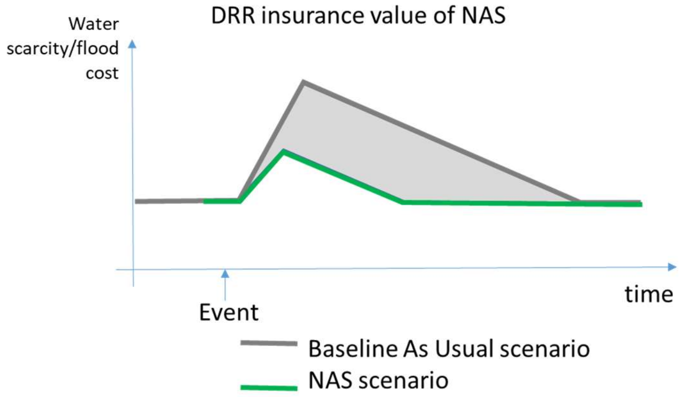

Figure 2 *Illustration of the insurance value* stemming from the risk reduction potential of NBS / NAS. The grey area is the insurance value (Source: own elaboration)

4.3. Conceptual framework for insurance value in NAIAD

In NAIAD we build on these two interpretations of the insurance value, which can be considered as two components of a total insurance value of any risk reduction strategy as illustrated on Figure 3. Opting for NBS exploits on two types of attributes of NBS: (i) **their risk reduction potential and (ii) the sensitivity of the NBS**, which will inform about the capacity of sustained supply of risk reduction potential and co-benefits production in the face of shocks or stresses. The two attributes together form the total insurance value. We consider shocks and stresses as threats. Drivers that evolve slowly like climate change or population increase represent **stresses** for ecosystems and threats to ecosystem services provision. Natural hazards are considered as shocks and we anticipate that their frequency and severity might increase with climate change.

In the context of NBS implementation, we could argue that any action or strategy that would safeguard or improve the state of ecosystems, building on the capacities of the ecosystems can be interpreted as a strategy that boosts or at least safeguards the inherent insurance capacity and value of an ecosystem.

An important remark also while merging the two strands of **Disaster Risk Reduction** and **Ecological valuation or ecological Economics** is that we do not adopt as main methodological framework the perspective of the ecosystem service approach (see 3.3). In theory, the ecosystem service framework includes the services of regulation, among them the services of flood, scarcity or drought regulation. However, in the perspective of NAIAD, where disaster risk reduction is the main motivation, the insurance value with regard to the reduction of water related risks is manipulated as a separate concept, while the **ecosystem services** (valuation) approach will be used to assess some[5] of the **co-benefits** provided by ecosystems (when they service level increase with NBS). The variation in the production of ecosystem services will be either co-benefits (if they increase) either opportunity cost (if they decrease).

One share of these co-benefits will be associated to quasi-option values and will emerge from the **resilience capacity** of ecosystems.

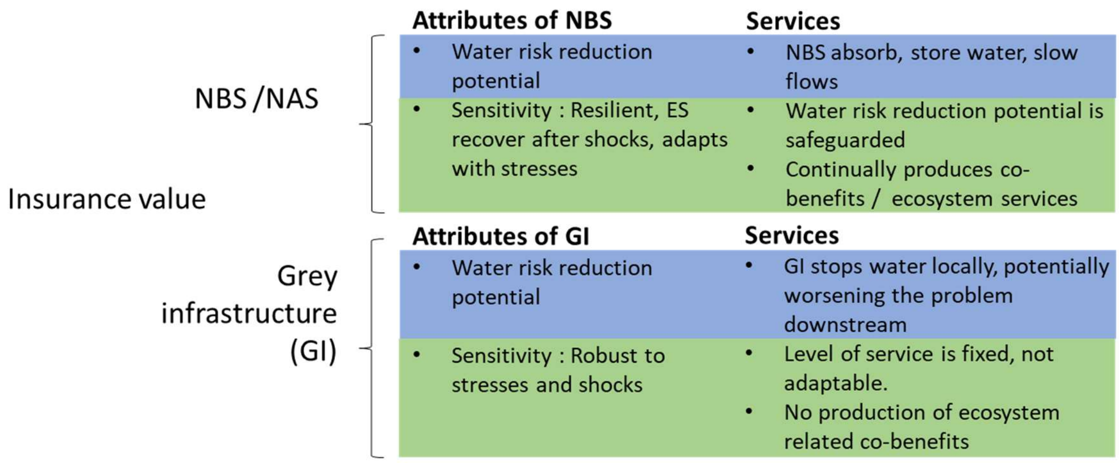

Figure 3 The two components of insurance value: (i) water risk reduction potential, (ii) sensitivity of NBS

In the economic analysis, it is necessary to **compare an alternative to a reference situation** in order to enlighten the relative costs and benefits of each. Typically, defining the risk reduction component of the insurance value will require making the difference in the damages in two situations (at least):

- **The baseline risk reduction strategy is defined as current practice and policy in each DEMO**: Grey infrastructure such as flood control infrastructure have obviously an insurance value in the face of natural hazards and often must be seen in combination with NBS. It can reduce uncertainty, mitigate risks, improve resilience and reduce damage, where NBS can provide additional capacity. However, these infrastructures might not be resilient in the face of other hazards and might have a negative effect of shifting the risk further downstream. It can also provide a number of co-benefits over the short, medium and long term.

[5] Other co-benefits can also be considered such as more social justice.

- **NAS risk reduction strategies** are those that integrate at least one NBS besides grey infrastructure and that can be considered in DEMOs. We think that NAS strategies that offer at least the same level of risk reduction potential than baseline risk reduction strategy must be contemplated. In NAIAD DEMOs more than one should be considered with different levels of "greening" e.g. of share of NBS among all strategies. These strategies can mitigate risks, improve resilience and reduce damage. They can also present a number of co-benefits. Intrinsically the NBS maybe better at enabling a sustained flow of value in the face of hazards or stresses and in non-economic terms it means that we should value the diversity (resilience) of an ecosystem in the set-up and assessment of NBS and NAS. This should be considered at least qualitatively.

Each of these risk reduction strategies are more or less relevant in the face of a natural hazard. The assumption that we adopt in NAIAD is that the insurance value of a NAS strategy outperforms (or at least equals) the baseline strategy. Said differently we should consider only NAS that have a higher insurance value than grey infrastructures that are already implemented. Some will guarantee the sustainability of these insurance relevant ecosystem values over the long term when faced with stresses (e.g. climate change) and shocks (e.g. natural hazards). The insurance value of a NAS will enable resilience and recovery, in the case where it is affect by a hazard or stress, to ensure its long lasting insurance capacity.

5. General approaches to the analysis of alternative strategies

First, the point of view of the analysis is critical. In NAIAD a collective (or public) point of view is adopted because the strategy that are examined are focused on developing collective goods[6] such as nature based solutions which aim is to protect communities; and collective goods need public interventions (Krutilla, 1966).

5.1. Cost-Benefit Assessment (CBA)

Among the most classic economic assessment methods are Cost Benefit Analysis (CBA) and Cost-Effectiveness Analysis (CEA). CBA can compare two or more alternatives accounting for all costs and benefits of different alternatives. Alternatives can be policies, plans or projects at different scales. CEA can be used when the objective is to compare different strategies with regard to a unique type of service that is valued with a quantitative non-monetary metric (the effectiveness) and other benefits are ignored or not considered significant. On the other hand, Cost-Risk Analysis can be used in cases where no benefits (co-benefits) distinguish the scenarios, which is not the case in NAIAD. In the case of NAIAD, CBA is relevant as it will enable to compare

[6] In economics, a collective good is a good that is non-excludable (no one is excluded from receiving benefits) and non-rival (the consumption of the good or service does not reduce the possibility of consumption of another).

strategies with different service levels and account for multiple impacts that will be expressed as benefits such as potential co- benefits linked to ecosystem services. Also CBA helps design optimal strategies as the metrics can be optimized.

The main aim of CBA is support to policy or help to decision making, and was first developed to justify regulations in the US (Pearce et al., 2006). CBA is based on economic welfare theory and should integrate preferences of citizens on basis of their willingness to pay. Realizing a CBA is interesting for other than cost comparison arguments, as the process of identification of the effects makes people and communities discuss on these; which is already a result per se (Shreve & Kelman, 2014).

As NAIAD adopts a social, and public policy perspective, CBA is a relevant choice as it adopts a social (welfare) point of view. The idea of CBA is that the sum of the individual interests and preferences are integrated and aggregated in a unique metric. If the benefits outweigh the cost, even if someone loose something, they could in theory be compensated by the winners that still would have gains left (Kaldor-Hicks Criteria).

The idea of the approach is to compare the sum of all direct and indirect costs and benefits (damages and benefits) that differ between both situations to illustrate the advantage of one respectively to the other. The point of view of institutional economics is to consider also transaction costs in the analysis. Transaction costs are those incurred by the coordination mechanism between agents. Without being able of quantifying these costs, they should be considered when concluding on the CBA (the net benefits should be above the transaction costs).

An overview of the principle of cost and benefit comparison is proposed in Figure 4.

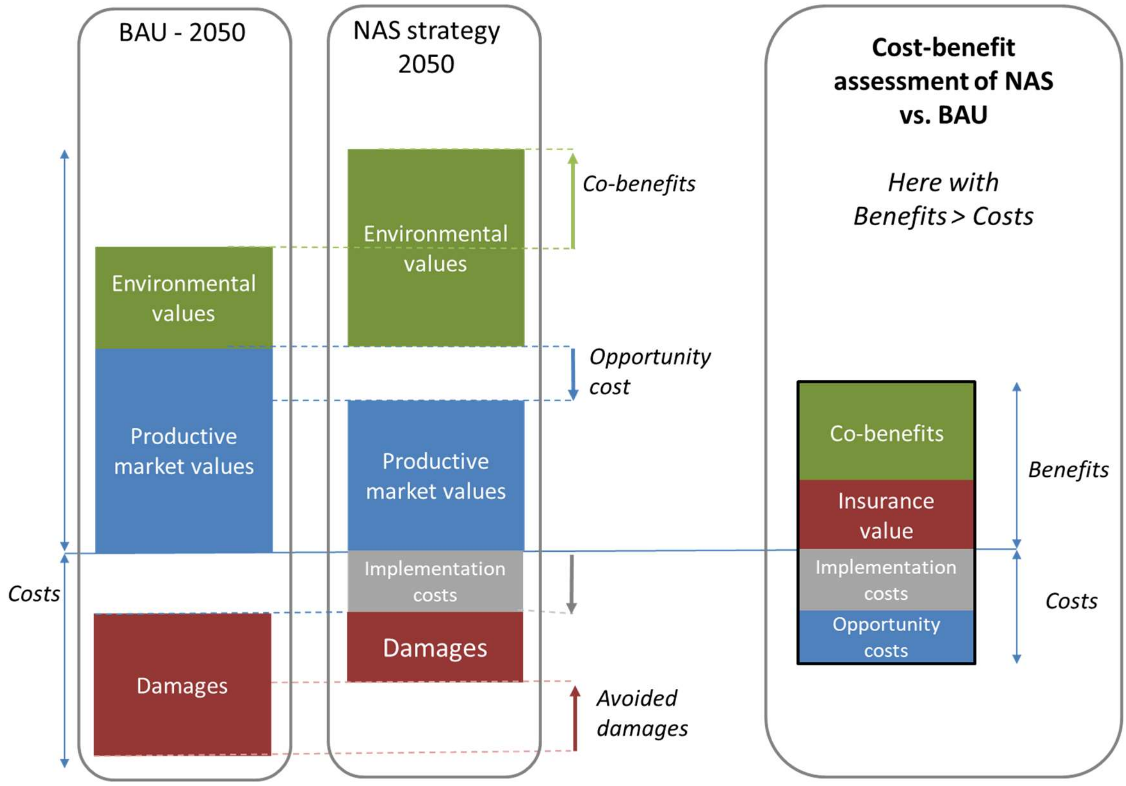

Figure 4 Overview of the different values or costs for different strategies and their translation for a cost-benefit assessment. Nb: in some cases BAU will integrate new grey infrastructure that will need to be accounted for. Here the case where a BAU involves no new infrastructure is illustrated.

Typically, the insurance value of NAS is one of the advantages we want to illustrate: it will be mainly the difference between baseline risk reduction strategy damages and NAS strategy damages.

Box : Example of the role of the economic analysis in flood control projects state subsidy allocation

In France, the use of the economic analysis is necessary for flood risk reduction projects ("programmes d'actions de prévention contre les inondations d'actions de prévention contre les inondations (PAPI)") to candidate for public subsidies from the Barnier fund. CBA has been extended to account for other indicators in a multi-criteria setting. The exact constraints are as follow :

- A MCA (including CBA) if the cost is above 5 M€ (without tax).
- A CBA if the cost is between 2 and 5 M€ (without tax (si MCA is optional).

- An adapted economic argumentation if the cost is under 2 M€.

The ministry for environment has published two methodological guides, which need to be followed to be eligible to the Barnier fund. The MCA setting enable to account for quantitative indicators relative to (i) human health, (ii) economy, (iii) environment and (iv) local heritage assets and qualitative indicators that are not predefined. More info : https://www.ecologique-solidaire.gouv.fr/levaluation-economique-des-projets-gestion-des-risques-naturels

5.1. Multi-criteria & economic analysis

Multi-criteria analysis (MCA) is used in many situations. For both the private and public sectors (Belton and Steward, 2002), MCA is a formalized process to combine monetized measures with non-monetary indicators, generally with at least one direct quantitative measure contained within the assessment. The approach aims to take into account a number of criteria rather than focusing on a single criterion, such as through Cost-Benefit Analysis (CBA) or Cost Effectiveness Analysis (CEA). It should be noted that the MCA is not a substitute but rather a complement to CBA and CEA analysis. However, unlike CBA, in MCA there is no explicit rationale or necessity that benefits should exceed costs. In MCA, as with the case for CEA, the 'best' option can be inconsistent with improving welfare, so doing nothing could in principle be preferable but may undermine the overall objective.

MCA uses a formalized process to combine monetized measures with non-monetary criteria, generally with at least one direct quantitative measure of ecosystem quality or quality contained within the assessment. A common feature to MCA methods is the performance matrix, where each row is a policy option and each column describes that option's performance along a specific dimension. The performance matrix helps to guide the two necessary stages for combining all attributes: scoring, turning qualitative measures into ranks; and weighting, numerically determining the relative importance of each quality (Dodgson et al., 2009). From there the choice of MCA technique is relatively open, with a key factor being the number of alternatives that need to be evaluated.

Due to the presence of strong assumptions and high levels of uncertainty in MCA (Schoemaker, 1991; Bell et al. 1988), scenario analysis can be used as a complementary tool that allows multiple potential pictures of the future to be presented. Sensitivity analysis, broadly defined, is the investigation of potential changes and errors to parameter values and assumptions of the model and their impacts on conclusions gathered (Pannell, 1997). To check for the robustness of the results a sensitivity analysis can be conducted. Triantaphyllou and Sánchez (1997) highlight that a sensitive decision criterion is the one with the lowest weight if changes are measured in absolute terms and it is the one with the highest weight if weight changes are measured in relative terms (i.e. as a percentage). This suggests that the number of decision criteria is more important to the MCA than the number of alternatives.

5.2. General methodological framework

This section provides a general stepwise approach that we recommend to use in NAIAD DEMOs to implement an economic analysis methodology. This framework should be implemented while having recourse to variable types of methods among which stakeholder implication should be considered and input (data, modelling). Every DEMO will have a different stakeholder implication level, however, we recommend that stakeholders are associated to the work of each step except step 4. Stakeholder implication methods to account for risk perceptions will be developed in WP3 and stakeholder based methods to account for co-benefits will be detailed in D4.4.

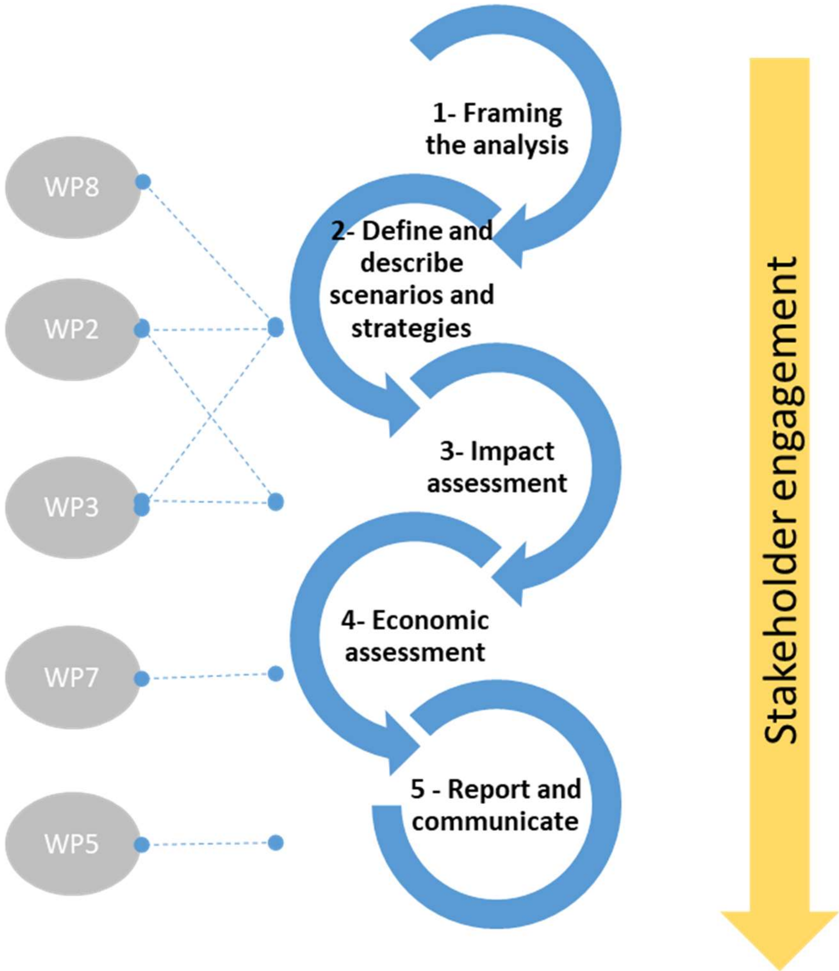

Figure 5 Methodological framework for the economic assessment in NAIAD and links to other WP (means that this step must be realized hand in hand with WP)

1. Framing the analysis

- Define scope and objectives :
    a. define the other specific questions that the analysis should aim to answer above the central question of "Will the NAS strategy provide more net benefits than classical DRR strategy?"
    b. Choose between a ex-post or ex-ante analysis ?

Two different approaches can be adopted a **(i) backward or historical perspective or (ii) a forward looking or future perspective**. This choice will depend on whether NBS have already been implemented in the DEMO or not, and what we want to illustrate. In the case of the historical perspective, the idea is to illustrate how NBS have delivered an insurance value, in the future perspective the idea is to illustrate how NBS could deliver an insurance value. In both cases, the idea is to compare strategies that integrate NBS to situation with grey infrastructures. Defining this question is encouraged to be realized with stakeholders. As the economic analysis have the ambition to serve operationalization of NBS, thus the analysis has to help decision makers take actions. To do that, it is important that we answer their questions in order to lower barriers of stakeholders to invest or accept NBS.

- Define spatial scales: Identify the spatial perimeter and the populations (human and ecosystem) at which impacts will be observed.
    a. Consider a sufficiently large scale to integrate all significant costs, benefits and populations of interest.
- Define the time horizon (see 7.1)

    The time horizon coupled to the discount rate (see 3.5) of the appraisal will have an important impact on the results. NBS may prove efficient in the long run, so a sufficiently far horizon is recommended (at least 2050). Different time horizons can be considered for different calculations (e.g. in a sensitivity analysis) to show the importance of considering the long run to enlighten the benefits of the NAS strategy.

2. Define and describe scenarios and strategies

Scenarios considered in the assessment will be a combination of both strategies and socio-economic scenarios that will represent the evolution of the stakes and of the land uses as shown on Table 1.

The reference situation is the actual situation that can be characterized by a recent past year[7].

- Define Baseline strategy (or policy)

Establish the reference situation or baseline as well as the business as usual scenario. The Business as Usual (BAU) is a scenario that develops without significant policy changes, at least that depicts the trend without policy innovations. It will serve as a reference to compare the NAS strategy. It must be described over time period considered and can be non-static.

- Define NAS strategies (or policy)

Description of the measures that will be combined in the DEMO to form the alternative NAS strategy. It can be a combination of NBS and traditional grey infrastructure measures. If useful, the measures can be classified upon actions that target the reduction of (i) the hazard, (ii) the exposure (e.g. lesser construction in risky areas, stop of soil waterproofing), (iii) vulnerability (other than NBS measures). This strategy should be described over the time period considered.

[7] the current year is often difficult to characterize because of missing data

- Define past scenarios for historical perspectives

Historical scenarios in terms of land use and socio-economic development must be considered to assess the insurance value and co-benefits of alternative strategies in a retrospective perspective. The observed development is the basis for comparing both a none-NBS strategy and a NBS strategy. The advantage of this approach is that no strong assumption have to be done about the scenario as it can be observed.

- Define future scenarios for forward looking perspective

In a forward looking perspective, strategies can only be correctly assessed given future scenarios that will characterize land use, economic development and structuring regulations. Indeed, NAS will have land use and socio-economic implications, so that a strategy needs to be supported by a possible land use development (e.g. it is hard to imagine giving more room for water if the housing market develops without densification around a river). Some land with different potential destination will be affected (agriculture, recreational, protected, residential or business development area) and consequences of land use will condition the future scenario (e.g. gentrification, need for depopulation or reduced growth of inhabitants, etc.). Therefore, a decision on choosing NBS solutions is closely connected to long term development planning, and presence or absence of future development scenarios, that lies in hands of local (or hierarchically higher) administrations. These future scenarios will need to be determined, at least qualitatively, by describing the land use and the trends in demography, economic activities and regulations. See Table 1 as an example.

Note that all combinations of scenarios and strategies are not necessarily realistic. Ideally, coherent combinations of scenarios and strategies are contemplated to illustrate the need for a multiple stake development perspective, such as the idea of "territorial projects"[8] that combine both socio-economic trends and a feasible policy scenario.

Table 1 Examples of scenarios that can be considered. Scenarios are a combination of strategies and future scenarios that describe the socio-economic evolution of the area.

SCENARIOS		Strategies		
		Baseline	e.g. NAS 1	e.g. Ambitious NAS
Future scenarios e.g. 2050	Business as Usual	B/BAU	1/BAU	2/BAU
	Low growth	B/LG	1/LG	2/LG
	High growth	B/HG	1/HG	2/HG

3. Impact assessment

[8] Developed in France to ensure a possible future that accommodated both the constraint of the maximum water use and a viable agricultural development.

- Identify and describe all impacts that will be observed when assessing the alternative(s) scenarios with regard to the reference BAU scenarios. Figure 3 illustrates the idea that NBS will have impact on land occupation and how different effects of hazards can be considered.

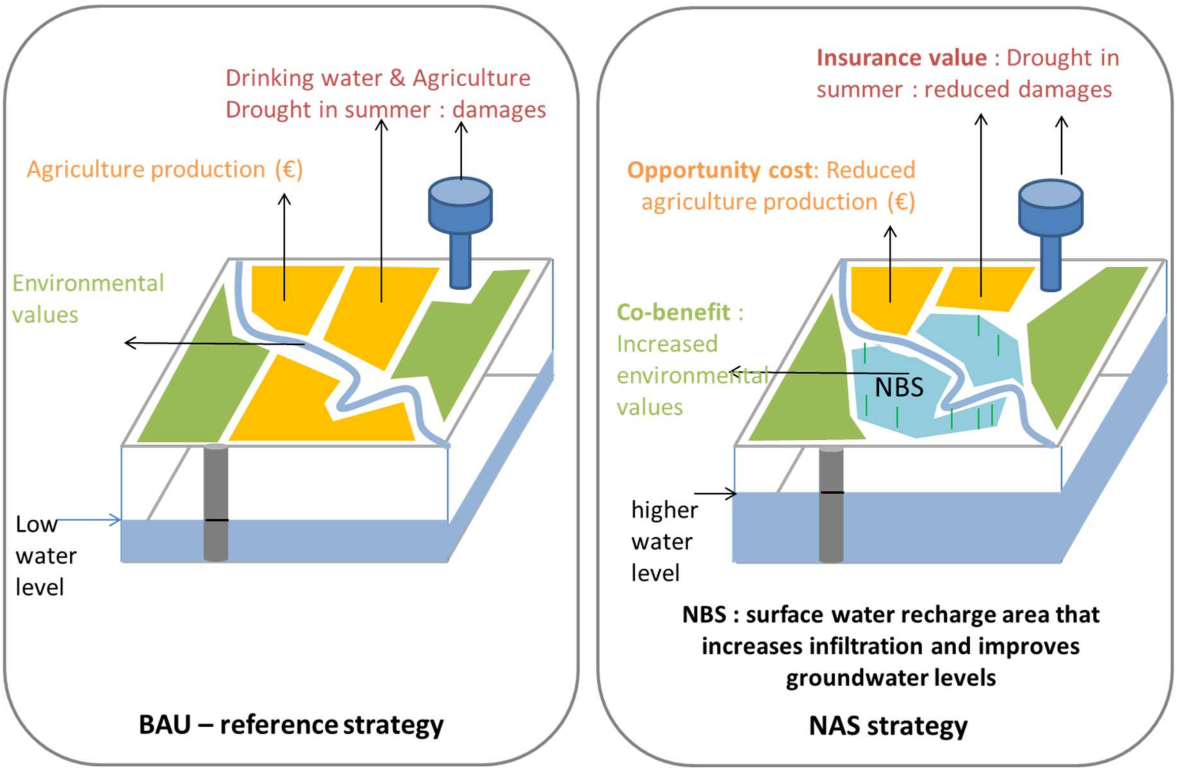

Figure 6 Illustration of the impacts in terms of land use and types of costs and benefits for the BAU and NAS strategy

- Identify the capacities/results of modelling (DEMO models, European scale models) that can help characterize the impact assessment.
- Determine for all impacts if these can be monetized as cost or benefits (that will be aggregated in the CBA indicator) or analysed with another metrics or qualitatively (MCA indicators).
- For those impacts that cannot be monetized, determine the method and realize the qualification or quantification for these indicators. This can be performed by mobilizing stakeholders and experts independently or during a workshop.
- Characterize the uncertainty for each impact, cost, benefit or other indicators assessment.

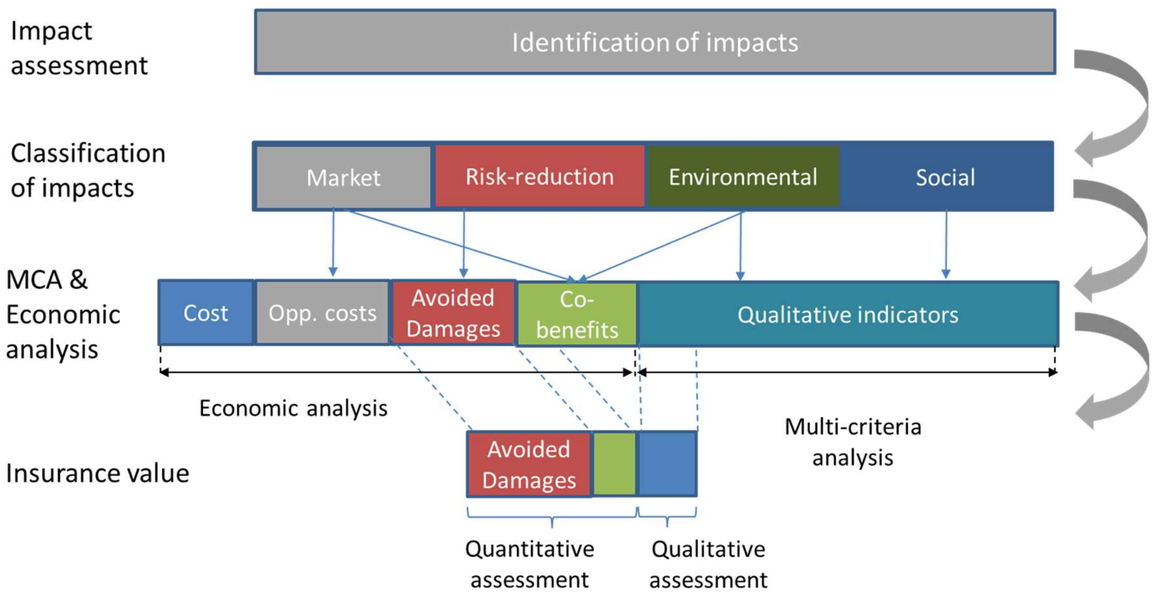

Figure 7 From the impact assessment to the economic analysis and the extraction of the insurance value. As explained in section 2, the insurance value has two components (i) water risk reduction potential (which is an avoided damage), (ii) sensitivity of NBS e.g. their capacity to deliver services in the face of uncertainty. Note that among the qualitative indicators there can be indicators relative to avoided damages and co-benefits.

4. Economic assessment

Details on this part are given in section 6.

- Classify the cost and benefits identified in the cost-benefit typology. The cost-benefit typology as shown on Figure 4 and adopted in NAIAD distinguishes:
  (i) Implementation costs,
  (ii) opportunity costs,
  (iii) avoided damages or damages for each strategy in every scenario,
  (iv) co-benefits

Section 3 details each of these costs and benefits and will give insights on how to assess them.

- Choose a method to assess each type of costs and benefits according to data and resource constraints (time and skills).
- Assess costs and benefits of all strategies.
- Choose a discount rate.
- Choose and calculate indicators. Calculate the indicators by aggregating costs & benefits. Different indicators can be calculated: present value of net benefits (PVNB) and the benefit cost ratio (BCR): the ratio of the present values of benefits and costs and the internal rate of

returns. When PVNB is positive (i.e. BCR is bigger than 1), the benefit is larger than the costs. To focus on the avoided damage of the NAS we also recommend the calculation of the present value of avoided damages and, to enlighten the interest of NBS in terms of risk reduction: the ratio of present value of avoided damage/present value of damage in the reference situation.
The Annex 1 details the calculus of the different indicators.

- Perform a sensitivity analysis or robustness analysis in case of deep uncertainty. (See section 3.9 on uncertainty7.5).

5. Report and communicate

- Combine the CBA indicator(s) in a table with the other indicators (lines= indicators; columns= scenarios)
- Present the sensitivity or robustness analysis and discuss.
- Illustrate the results with graphs that show the evolution in time of costs and benefits with no discounting and discuss them.
- Analysis of the trade-offs between insurance value and co-benefits for the reference and NAS strategies.
- The analysis of the share between insurance value, opportunity costs and co-benefits can inform about the way the project should be set-up (e.g. main driver risk reduction or other advantages?) and the barriers it will face (e.g. if high opportunity costs appear, for instance lost agricultural production, the project might face important opposition). This should also be linked to social acceptability and induced institutional change issues.
- Conclude on the question of the analysis and raise awareness on the limits and uncertainties.

6. Costs and benefits components

6.1. Implementation costs

Deliverable D4.2 assesses the life cycle costs (LCC) of NBS by providing an overview of the temporal and spatial distribution of costs related to NBS and the grey infrastructure counterparts. To characterize the implementation and operation costs of NBS and traditional infrastructure an understanding of the costs of the entire life-cycle is required. LCC entail all costs to be incurred during the lifetime of a product, work or service, including:

- Investment (Purchase price) and all associated costs (delivery, installation, insurance, etc.)

- Operating costs, including energy, fuel and water use, spares, and maintenance
- End-of-life costs, such as decommissioning or disposal

6.2. Opportunity costs / resource costs

The opportunity costs are those costs associated with the foregone alternative, which can be measured by the net benefit foregone because the resources that provide the services cannot be used in their next beneficial use (Tietenberg and Lewis, 2016). For example, the opportunity costs of developing a flood expansion area would be the benefits that would have been generated if it was converted to a profitable land use.

Taking policy actions regarding the land use, such as NBS, imply choosing between alternative uses of land and water resources. Beyond the environmental benefits provided by each alternative, these have implications in terms of opportunity costs (Ruijs et al., 2015).

In addition to the efficiency implications of policies, the assessment of the opportunity costs has great relevance for equity. While many environmental benefits are enjoyed regionally, even globally, the opportunity costs tend to be borne locally (Balmford et al., 2011).

NAIAD analyses the potential of NBS to reduce exposure to flood and drought hazards. The implementation of NBS implies alternative uses of both land and water resources. In such a context, the major opportunity costs to be considered are the net benefits that could have resulted if the land and water resources compromised by the NBS implementation were devoted to other revenue producing activity, such as agriculture, timber production, etc. **As NBS are often considered to need more land than the traditional grey infrastructure alternatives, the opportunity cost of land might be significant, particularly when these are located in cities where land is very valuable.**

The assessment of such opportunity costs can be assessed through the straight calculation of the income lost from such resources not being used in the best alternative available or, in case of loss of property rights, the cost of compensation. For agricultural land, this can be done using data on average income per area of agricultural/forestry land, which is usually available from national agricultural statistics or from the European Farm accountancy Data Network (FADN). For urban land this can be done using the market price of land or as for farm land estimating the value forgone by the next economic activity.

Opportunity costs can also be considered when there is a **foregone use of water** (Note that in the case where drought is the hazard considered then it will be integrated in the damage cost and not as an opportunity cost). Data on the agricultural value of water can be found in the abundant existing scientific literature. Alternatively, it can be calculated from agricultural statistics on farm income and agricultural water use or using different methodologies, such as the residual method, agro-economic models or others (see Young, 2005, for an overview of methods to value water resources).

In the case of agricultural/forestry land, opportunity costs could be also more directly assessed by its market price, which is the capitalized value of the annual income generated by these activities. Water resources are rarely traded in markets, so this would not be an option. However, it must be noted that, where water is a relevant input for agricultural production, its value could be already incorporated in the market price for irrigated land. In parallel, the value of water resources could also be estimated as the difference between the market price of irrigated and non-irrigated land.

In addition to the benefit lost from not using land and water resources for the best available alternative, there might be also opportunity costs derived from the restrictions that the implementation of NBS may impose on other economic activities, such as restrictions on agricultural/forestry practices, fishing or mineral extractions, for example (Kaphengst et al., 2011). In this case, opportunity costs could be assessed through the resulting loss income or through the resulting loss of land value (Lawley and Towe, 2014). If these restrictions could potentially lead to a change of the cropping patterns in the area, this could be assessed using agro-economic models.

In addition to the above, opportunity costs may also take the form of indirect effects on the rest of the economy. For instance, a reduction in land may not only result in a loss of income but also in a reduction of employment and other indirect effects on the economy. This can be assessed using more comprehensive methodologies such as input-output analysis or Computable General Equilibrium models. However, these will be neglected in NAIAD, but can be reported as a qualitative indicator in the MCA.

6.3. Damage costs

6.3.1. Definition and types of damage costs

Natural disasters impose large costs to societies. There is an ample literature on the assessment of the economic value of the damages caused by natural disasters. Following Brémond et al. (2013), we can define “damage” as a negative impact of a natural hazard/disaster on a socioeconomic system and “cost” as the monetary valuation of the damage. The term “loss” would designate damages that result in the destruction or depreciation of some element of the socioeconomic system.

A first step in the assessment of the damage costs of a natural hazard is to clearly define the baseline scenario and the purpose of the assessment, e.g. for the calculation of insurance premiums, for the design of prevention and risk mitigation measures, for the assessment of compensations after the hazard, etc. (Hallegate and Przyluski, 2010). The purpose of the assessment will determine the type of damages to be measured.

The damages from natural hazards, and their costs, can be classified in tangible and intangible (Merz et al., 2010; Meyer et al., 2012). Tangible costs are those resulting from impacts from the hazard that can be easily quantified in monetary terms, while intangible costs are those that are difficult or even impossible to measure, as they comprise non-market values (Merz et al., 2010; Brémond et al., 2013; Balbi et al., 2013). The sum of all tangible and intangible costs resulting from a natural disaster is the total cost of the natural hazard for a socioeconomic system (social cost of the hazard), a concept which is symmetrical to that of total economic value, used in environmental economics to estimate the benefits provided by natural resources (Balbi et al., 2013).

At the same time, damages can also be classified **in direct and indirect** based on their spatial extent. Direct costs would be those resulting from damages produced in the geographical area directly affected by the natural hazard, while indirect costs would correspond to damages that take place outside the area affected by the hazard (Merz et al., 2010). However, the distinction between direct and indirect costs is also used to represent the time scale of the damages (Merz et al., 2010; Brémond et al., 2013): direct damages would take place during the hazard event (damages to human life, infrastructures, capital or production), while indirect damages would occur later in time (production losses due to business interruptions caused by electrical shutdowns, people that cannot go to work, lost or deteriorated production capital such as trees or machines, etc.). That is, indirect damages would be induced by direct damages and would occur, either in time and/or space, outside the hazard event (Merz et al., 2010). In other words, indirect losses would be those "*spanning on a longer period of time, a larger spatial scale or in a different sector than the disaster itself*" (Hallegate and Przyluski, 2010).

Both space and time dimensions are of special relevance when defining the damage costs to be measured. Moreover, the time will differ significantly depending on the hazard in question. For instance, the time scale of a drought is not comparable to that of a flash flood or an earthquake. The spatial and time extent of the assessment must thus be clearly defined (Merz et al., 2010).

In sum, we could distinguish between direct tangible, direct intangible, indirect tangible and indirect intangible costs. However, such classification comes mostly from the flood damage assessment literature, and maybe not fully adequate for other hazards, such as droughts.

Meyer et al. (2012) propose a different classification: 1) Direct costs; 2) Business interruption costs in the area affected by the hazard, of the enterprises directly affected by the hazard; 3) Indirect costs induced by direct damages or business interruption both inside or outside the area affected by the hazard; 4) Intangible costs; and 5) Risk mitigation costs (which can be direct, indirect or intangible). Similarly, Rose (2004) distinguishes between direct effects, business interruption effects and "higher-order effects" (those impacts beyond the physical destruction or deterioration of assets and the resulting loss of production due to business interruption). The latter would include intangible or non-market effects. However, Hallegate and Przyluski (2010) propose that business interruption losses be included in the direct costs to avoid consistency problems when dealing with drought hazards, which span over a long period of time.

The relevance of each type of damage is case-specific. For instance, the non-market effects of hazards depend on the level of impact on the environment, on the cultural or historical heritage, on public health and on human life. Indirect effects are directly related to the duration of the reconstruction pace after the hazard, but also to the spatial extent of the assessment (Hallegate and Przyluski, 2010; Molinari et al., 2014).

NAIAD focuses on flood and drought hazards, which are very different events in terms of their duration and spatial extent of both the hazard and of the associated damages (OECD, 2013). Although they are basically related, respectively, to the excess or deficit or rainfall compared to "normal" or "reference values", floods result from sudden events while droughts are cumulative events over much longer periods of time and that normally affect larger areas.

Both floods and droughts may result in severe economic impacts both in rural and urban areas as a consequence of the loss of households and productive assets, loss of production, depreciation of the market value of physical assets (land, buildings, etc.), water restrictions, etc. A classification of flood damages in urban areas can be found in Jonkman at al. (2008), while Brémond et al. (2013) propose an exhaustive classification of flood damage costs in the agricultural sector based on a review of studies on this topic. Classification of drought impacts can be found in Kallis (2008), Wilhite (1993, 2000) and Logar and van den Bergh (2012).

6.3.2. Methods to assess damage costs

Different methods can be used to monetarily assess the impact of damages. The choice of method will mostly depend on the availability and quality of data. Two types of approaches to damage cost assessment can be distinguished: ex-post and ex-ante (Rose, 2004; Meyer et al., 2012). The first relies on the analysis of recently occured events, while the second is based on the estimation of the impacts of scenarios that have never been observed. The first method is used for events for whom the hazard or damage data is available. This second method is generally used to simulate:

- Extreme events that have never occurred (or occurred in the distant past before data recording);
- Stochastic event sets to have an exhaustive knowledge of the exposure.

Similarly, damages can be assessed by the independent expert (insurance experts, construction experts) who provide loss estimates based on experience eon real observed events (building damage curves or providing mean losses per type of exposed risk) or can be estimated in virtual/simulated cases based on models (that might be calibrated on real observed data) or simplified estimation. In this later case, strong assumptions will be made.

The so-called “real” value of damages may sometimes be estimated using different sources:

- Insurance claims (generally extrapolated to complete missing data, an example can be found on the CCR website https://erisk.ccr.fr/faces/erisk-accueil.jsp
- Post-event feedback, particularly for non-insured losses to have a global economic loss asset.

Regarding the **assessment of direct damage costs**, it is important to take into account that these are directly related to the assets’ susceptibility to the hazard intensity (Merz et al., 2010). A common feature of most studies assessing the direct damages is the use of stage-damage functions (or susceptibility functions), which represent the relationship between the characteristics of the hazard (or other properties in a multivariate correlation) and the damage caused by the hazard to a specific type of asset (called element at risk). Damage functions can be developed for each type of element at risk either empirically using data on real damage losses or hypothetically from “what-if analysis” using expert assessment (Merz et al., 2010; Cammerer et al., 2013). The alternatives are using previously estimated damage functions for the area of study or transferring damage functions developed for other areas or for larger spatial extents (Cammerer et al., 2013). Merz et al. (2010) review studies assessing direct damages for flood hazards, while Brémond et al. (2013) review those studies that focus specifically on the agricultural sector. An alternative to using real damage losses data is to use insurance data (policies and claims) to develop damage functions.. Some recent examples are the studies by Moncoulon et al. (2014), Naulin et al. (2016) and Watson et al. (2016). In most cases, the hazard characteristics used to develop damage functions are model outputs. For example for flood, the level of water for the elements at risk are simulated using a hydrological model. In this case, the hazard characteristics present the uncertainties of the hydrological models. Thus, such a calibrated damage curve is strongly connected to the hazard model that was used for calibration. This specificity make it difficult to use a given damage curve with a different hazard model (or a flood prone area for example).

Flood damage models allow assessing the damage caused by specific or hypothetical flood events that are characterised by one or more parameter, but they can also be used to assess the risk of flood or flood exposure. Flood risk assessment consists in combining the hazard characteristics and the damage evaluation with the probability of occurrence of the hazard event to obtain damage –probability curves or flood-risk curves (Rojas et al., 2013; Foudi et al., 2015). Damage-probability curves relate flood damages with their probability of occurrence. In addition to the parameters characterising the hazard intensity, the event should be also characterised in probabilistic terms. In the case of NAIAD, damage-probability curves will be the basis for the calculation of the insurance value of NBS. Some examples of flood risk/exposure assessments at different spatial scales can be found in Rojas et al. (2013), Jongman et al. (2014), Moncoulon et al. (2014), Foudi et al. (2015) and Naulin et al. (2016).

In the case of droughts, the most common approaches to assessing direct costs are those based on market valuation techniques, such as the avoided cost approach, the replacement cost approach, the production function method and the market price method (Logar and van den Bergh, 2013). The market price method can be also used to estimate damage functions, which relate the economic impact of drought with the drought intensity (Gil-Sevilla et al., 2010; Gil et al., 2011; Lopez-Nicolas et al., 2017; Giannikopoulou et al., 2017). In the case of the direct impacts of drought on agriculture, another commonly used method is simulating them using agro-economic models, which, as damage functions, can be integrated in an hydrohydro-economic framework (Iglesias et al., 2007).

The **estimation of indirect damage costs** is more complex, as it requires accounting for the induced economic impacts of the hazard beyond the area directly affected by it. In addition to the collection of data on past events (from statistical data or ad-hoc surveys), the major methodological approaches rely on input-output, computable general equilibrium or econometric models (Cochrane, 2004; Merz et al., 2010). Rose (2004), Okuyama (2007) and Hallegate and Przyluski (2010) provide critical discussions of each approach. Although these methods provide more comprehensive assessments, the number of empirical applications in the case of floods and droughts is very small compared to studies assessing direct damage costs. Examples of the estimation of indirect damages are Jonkman at al. (2008), Carrera et al. (2015) and Schulte in den Bäumen et al. (2015) for flood events and Pérez y Barreiro-Hurlé (2009) and Martin-Ortega et al. (2012) for drought events.

The methods to assess **losses due to business interruption** are different from those used for direct and indirect damages (Meyer at al., 2012). In the case of floods, the most common approaches are using sector-specific reference values (e.g. daily lost added value per worker, per area, etc. depending on data availability) or directly estimating the production losses as a percentage of direct costs. In the case of droughts, the most common approach is the analysis of specific events by comparing the annual value of production ex-ante and ex-post (see Martin-Ortega et al., 2012, for an example).

It is worth to mention that risk mitigation costs are sometimes integrated in the damage cost assessment (in a perspective of global risk cost calculation, e.g. Meyer et al. 2012) by computing the direct costs of their implementation (Bouwer et al., 2014). In NAIAD these costs are considered as implementation costs (see section 3.1).

Last, while numerous studies on the field of the economic analysis of natural disasters have assessed tangible costs (Meyer et al., 2012), the issue of **assessing intangible or non-market damages** has barely being addressed (remaining cleaning or damages that individuals have to cope with, psychological traumas…). Markantonis et al. (2012) and Meyer et al. (2012) review methods for the economic valuation of the intangible damages of natural hazards. For those impacts that can be valued and quantifies, these are the same methods used for the economic valuation of environmental goods and services, which will be used in NAIAD for the assessment of the co-benefits of NBS (section 3.4).

6.4. Co-benefits

6.4.1. Introduction

The existence of co-benefits of NAS is one of the most powerful assumptions for the additional benefits of NBS compared to classic grey infrastructure risk reduction strategies. We assume that these co-benefits, on top of the insurance value of NBS, will make the NBS and NAS viable and bankable. They will be necessary to build the business case for NAS. This first argument is also related to the legitimacy aspect. Understanding the production of co-benefits is relevant because alternative policies can be leveraged to meet multiple restoration goals efficiently and may be important in promoting voluntary compliance *(*Wainger et al., 2013).

A dedicated deliverable (D4.4) will be dealing with the estimation of co-benefits with ecosystem services and natural capital accounting. As such, the following paragraph is an introduction to the topic and DEMOs will benefit from an in-depth guidance with D4.4 to design DEMO adapted methodologies for the valuation of co-benefits.

We define co-benefits as benefits of NAS or NBS that arise beyond the primary benefits of damage reduction of NAS and NBS. Thus, the co-benefit term is very specific to the NAIAD perspective that has as a main aim the disaster risk reduction and associated climate change impacts. For instance, even if reforestation has a higher benefit in timber valuation than for flash flood prevention, timber production will be considered as a co-benefit to the insurance value of the NBS (here reforestation).

The recent literature on "Ecosystem-based adaptation" (EbA) to climate change impacts has also drawn upon this notion. As Munang et al. (2013) put it, "EbA is the use of natural capital by people to adapt to climate change impacts, which can also have multiple co-benefits for mitigation, protection of livelihoods and poverty alleviation". They even insist that "the main advantage that EbA has over others adaptation approaches are that it can deliver multiple co-benefits" since it can achieve multiples policy objectives: climate change adaptation and mitigation; socio-economic development; environmental protection and biodiversity conservation. Two main advantages of EbA, vis-à-vis "hard engineering solutions", have been identified in the related multidisciplinary literature: (i) EbA might be more cost-effective since it is more adaptable and it has multiple co-benefits; (ii) it is more amenable to "community-based adaptation" that would foster "learning by doing".

The word co-benefit can have different meaning depending from the perspective of the study or policy area. For instance, the notion of "ecosystem services co-benefits" grown out from the concern of biodiversity conservation to suggest a potential of 'win–win' situations where biodiversity conservation and the delivery of ecosystem services overlap (see e.g., Karousakis, 2009). Most studies in this field deal with the role that species-led management for the benefit of biodiversity can play in the delivery of wider ecosystem services.

6.4.2. Identification of co-benefits

We suggest building on the Eklipse framework (Raymond et al., 2017) for the identification of the types of co-benefits, as it suggest a systematic analysis of NBS impact. The **Eklipse framework** provides an analytical framework for assessing the synergies and co-benefits provided by NBS in urban areas[9] as shown on Table 2. They argue going beyond the ecosystem service approach (e.g. European MAES[10] framework) while considering also the socio-ecological context when valuing NBS. Eklipse suggests considering 10 challenge areas against which NBS can be assessed. The work also suggests different types of indicators detailed in the Annexes of the report that will be of help in NAIAD. The 10 challenges areas are:

- Climate mitigation & adaptation
- Water management
- Coastal resilience
- Green space management
- Air quality
- Urban regeneration

[9] Even if this has specifically been developed for urban areas it might accommodate most cases where NBS are developed

[10] MAES (Mapping and Assessment of Ecosystem and their Services) framework (http://ec.europa.eu/environment/nature/knowledge/ecosystem_assessment/)

- Participatory planning and governance
- Social justice and social cohesion
- Public health and well-being
- Economic opportunity and green jobs

Some challenge areas refer to environment and can be assessed with the lens of the ecosystem service approach, other refer to society and will need other approaches to be assessed.

Table 2 Illustration of the Eklipse approach to NBS assessment for a tree cover

	Challenge area	**Expected impact of NBS**	**Indicators**	**Metrics**
Example	Climate Mitigation	E.g Carbon sequestration in soil	Carbon storage and sequestration in vegetation and soils	T of carbon stored per unit area

Different scales (local, meso and global) are distinguished particularly for the climate and water challenge. Note that in NAIAD´s perspective, we exclude some of the co-benefits of Eklipse that are the main aim of NBS for which they will be implemented, such as flood regulation capacity or drought management capacity, and that will be accounted for in the damage valuation (insurance value).

Concerning the ecosystem services which will be an important part of the co-benefits, further research will be done in Task 4.4 on co-benefits assessment that will review the different methods to characterize the co-benefits and see on which of the latest developments NAIAD should base its methodological approach to ecosystem services valuation (IPBES, CICES…).

6.4.3. Valuation

Box : Monetisation ? where do we stop ?
There is a debate on the relevance of monetization of ecosystems services (e.g. Raymond et al. 2017). Obviously, monetization enables to assess the importance of a project or policy in comparison with another and provides a rational to support decision. Choices made without monetary valuations are not necessarily inferior, as long as there is clear specification of the conditions of disclosure over what is worth valuing (Vatn and Bromley, 1994). The primary argument against non-monetary type methods is that they provide no way to ex-ante assess the relative performance of alternative scenarios. So while most allow you to judge which policies are preferred to others, there is rarely any measure of by how much and what the effects are of marginal changes, which is the case of monetary methods. These will be discussed further in 6.4.

Once we have defined what we want to value, we can begin to concern ourselves with how it should be valued. **Table 3** provides a list of frequently used methods, not limited to economic methods, grouped according to the manner in which they determine value. *Direct market methods* focus on a particular relationship between human activity and the environment, using existing market prices and behaviour as the contextual starting point from which a value for ecosystem services can be determined. Two such methods are adjusting existing market prices for non-market factors or estimating a production function to determine a shadow price for an environmental input or output. These methods are often data intensive and results can be sensitive to assumptions about functional forms and estimation procedures.

Table 3 Methods of Valuing Ecosystem Services (not necessarily economic)

Methodology	**Approaches**
Direct market	Market analysis (e.g. hedonic pricing), production function estimation
Indirect market	Replacement cost, avoidance cost, travel cost, recovery cost, resource cost, environmental cost
Measuring Attitudes	Surveys about attitudes, preferences, intentions; focus groups; behavioural observations; choice experiments; stated preference economic methods (e.g. contingent valuation)
Benefits transfer	Application of outside study values to a site of interest
Civic Valuation	Referenda and initiatives; citizen valuation juries
Decision Science	Choice modelling, multi-criteria analysis
Non-economic Quantity-based	Habitat equivalency analysis, ecological production functions
Ecosystem indicators	Indicators of economic and/or community-based values
Biophysical Ranking	Ecological footprint, embodied energy analysis

Sources: US EPA (2009) and Graves, et al. (2009).

Indirect market valuation is used when no relevant market exists, determining hypothetical market values based on observed behaviour, so-called revealed preference methods. These methods establish either the willingness to pay (WTP) for a service or the willingness to accept (WTA) a loss in services. They include a variety of cost-based approaches, which estimate costs to either replace, avoid or mitigate a specific type of damage, and hedonic estimation methods, which use price differentials in related markets to estimate the value of specific ecosystem services, e.g. the effect of access to parks on real estate prices.

Cost-based approaches estimate costs to either replace or to avoid/mitigate a specific type of damage. Edens and Hein (2013) argue that the two most relevant measures for measuring ecosystem services in monetary terms are restoration value (ex-ante, hypothetical value) and loss due to a degradation in the capacity to supply (ex-post, realized damages). Conceptually, these methods are related to the ideas of WTP and WTA, without margins for the incidental inclusion of non-use valuations because they are rooted in the economic concept of opportunity costs. This feature may or may not be desirable, depending upon the context of the ecosystem service(s) being evaluated.

Many ecosystem services also create value that is not and sometimes cannot be captured by markets, as is argued of many non-use values (Krutilla, 1967). In this case, the only option is to *measure attitudes* and directly ask users how they interact with the environment, most commonly through individual surveys, individual discussions, or group discussions. The economic versions of these methods are called **stated preference methods**, and they use questions aimed at obtaining value in monetary terms, either directly or indirectly through revealing the willingness to accept changes in bundles of ecosystem services. They are most commonly used to assess individual valuations, which can then be combined to determine an aggregate value. Reliance on survey data has its issues, particularly the well-discussed issue that WTP measures are generally higher than WTA measures in environmental applications such as ecosystem services valuation (Haab, et al., 2013). Shogren et al. (1994) provide evidence that this is due to the lack of available substitutes that is inherent for many public goods, with the solution of avoiding WTP measures whenever possible.

Such methodologies can also be used without the focus on monetary values, instead eliciting information about attitudes, preferences or intentions. Perceptual involve assessments stemming from detailed presentation of scenarios with verbally presented descriptions and images, while conjoint surveys require choices between multiple bundles that offer various attributes and reveal relative preferences for these trade-offs. Qualitative analyses without a monetary component tend to be interpreted either as ordinal rankings or interval-scale measures of differences in assessed values. Such methods are desirable when monetary terms are likely to be viewed as not ethically appropriate or there is difficulty expressing values of interest in such terms.

Civic valuation can also involve stated preferences or revealed behaviour. Stated preferences again take the form of surveys or representative groups (sometimes referred to as citizen's juries), if they are used to assess collective benefit values, e.g. through asking about the WTP or WTA of society instead of that of individuals. Revealed behaviour takes the form of community decisions such as voting behaviour on public referenda and initiatives.

Decision science methods attempt to determine the reasons behind the observed choices of individuals, usually without making the question of value explicit. These methods are particularly useful when there is significant uncertainty. One common method is choice modelling, where individuals are asked to repeatedly choose between two bundles of goods. Relative preferences over trade-offs are estimated by looking at how changing the bundle of goods, in terms of quantity, quality or their attributes, affects which option is preferred. These types of analyses, e.g. Koundouri et al. (2014) and Hanley et al. (2006), can use stated or revealed preference data, and can determine values of trade-offs in monetary or non-monetary terms.

Biophysical ranking methods focus on quantification of levels of quantities of biophysical indicators that underlie ecosystem services production, e.g. biodiversity or use of energy or materials. Similar to these measures are *ecosystem benefit indicators*, which focus on benefit specific measures rather than ecosystem-based physical quantities. Examples of such indicators include distance to the nearest vulnerable human community, users of a service (potentially within a given area), or amounts of specific land types or uses in the target area.

These can be used to inform group valuation methods or transformed into indices to allow for changes in levels of different variables to be ranked and compared. A subset of this combines a number of key tracking variables into indices which are then evaluated relative to a defined baseline or target level. This is not an economic method, because it does not implicitly provide trade-offs (e.g. between income and ecosystem services), instead focusing on levels of ecosystem services and their benefits. The lack of information about trade-offs is the major shortcoming of these methods.

The simplest index is a discrete rating, e.g. from 1 to 5, where higher numbers represent higher ecosystem quality, benefits, or better governance, e.g. use of hydrological indices and measures of river integrity to determine a four level ranking of the conservation status of major rivers in South Africa (Nel et al., 2007). A more sophisticated option is a standardized index, e.g. a rating that can take any value from 0 to 1 or a normalized index with 100 as the baseline, where differences between values reflect relative differences in ecosystem qualities. An example of use of this methodology is Vorösmarty et al. (2010), who use 23 stressors (drivers) grouped into four major themes to develop scores between 0 and 1 to assess relative levels of threat to biodiversity and water security, measured for 30' (latitude x longitude) cells across the globe.

An important category are *non-economic quantity-based* methods. Habitat equivalency analysis is used to determine compensation for ecosystem losses, and can be set up in monetary terms, although it is most commonly used for service-to-service trade-offs rather than to produce value measures (King, 1997). Carpenter et al. (2009) mention more scientific quantitative modelling and analysis of trade-offs as major steps needed to move beyond the MEA. Indeed these types of methodologies became a major point of focus in recent work, e.g. the InVEST model (Sharp et al., 2016), an open source suite of software that can currently account for eighteen distinct ecosystem services processes.

Benefits transfer involves the application of outside study values to a site of interest, the policy site. The costs of gathering site-specific data are the main reasons this technique sees relatively frequent use. The primary concern with such methods is accuracy, stemming from the appropriateness of applying the study site(s) to the policy site. The solution to this is careful analysis of what services potential policies affect, how these policies interact with the ecosystem and who the beneficiaries are (Plummer, 2009). Of particular interest for our purposes is ecosystem services mapping, which uses valuation estimates that can be linked to a desired ecosystem service and can be linked to a particular type of landscape. Then, for each ecosystem service and type of landscape, total value estimates are converted into marginal values (value per unit, e.g. $/hectare). The marginal value is multiplied by the amount of that landscape type for the policy site, adding up the value of all applicable ecosystem services to get the total value of a landscape type. The sum of the value of all landscape types gives the total value of the policy site (Plummer, 2009).

Benefits transfer can also be used to determine a value function rather than directly calculating a value. Often this is accomplished via a meta-analysis, using a large number of existing studies as data in a regression to determine the effects that observed characteristics of the site and human activity have on the measured value of the ecosystem service(s). Data from the policy site can then be input into the model to determine the predicted value. An example of this is given by Brander et al. (2012), who determine a value function for European wetlands and use it to estimate the changes in value due to climate change. Despite significant progress in recent years, there is still a big debate on how to increase accuracy and reliability of benefit transfers with an acceptable level of error (e.g. Bateman et al., 2011; Martin-Ortega et al., 2012; Glenk et al., 2015).

Towards assessment in the DEMOs

In NAIAD, potential NAS co-benefits will be DEMO specific. While certain services like e.g. carbon storage in forest will not depend on site specific characteristics, others will be DEMO specific. For those which supply and demand will not depend from local specific characteristics, the EcoActuary tool of KCL will be able to characterize some large scale properties (1 km – 10 km spatial resolution) of the services, e.g. by using remote sensing data. For the latter, their valuation will need an in-depth understanding of the use and non-use values of environmental resources in this specific setting (e.g. how will a recreation area be used by citizens; see e.g. Herivaux et al., 2017) and the results of detailed field work will be necessary.

Two different approaches to value co-benefits of NAS can be distinguished: (i) original studies will analyse and try to reveal environmental values for the specific DEMO, (ii) benefit transfer methods that will make use of existing valuation studies and adapt it to the specific case (see Box 2). In NAIAD, methods will necessarily be adapted to both DEMO specificities and data and time to collect missing data. The role of stakeholder engagement is encouraged at all the steps of the methodology, thus implying stakeholders in the valuation process is recommended. The role of local **stakeholder knowledge** and thus the role of stakeholder participation in the assessments of NAS co-benefits is a condition for both robust global assessments of NAS options and their acceptability by policy makers and end users. For example, Pagellab et al. (2013) present "Polyscape" – a GIS mapping framework providing a landscape scale valuation of multiple ecosystem services. They insist on the need for a simple classification system to identify and communicate synergies and trade-offs, while the standard economic assessment approaches (and even multi-criteria analysis) are said to introduce, for now, additional layers of complexity and require assumptions that can be difficult to communicate. Thus, in NAIAD a challenge for a good co-benefit assessment is both to engage stakeholders to balance and value the respective values of differing or converging ecosystem services. These methods will be developed in D4.4.

A particular reflection should happen in NAIAD and in DEMOS on limits on monetization of non-marketable values. This question will be considered from a stakeholder point of view, a technical (data) point of view and a qualitative/scientific point of view (is it worth to monetize under uncertainty and how to make uncertainty transparent to the stakeholders)?

7. Issues at stake for the economic assessment in the DEMOs

7.1. Time

The time horizon of the economic assessment will be critical, as it will induce lesser or more integration of future long-term effects (costs and benefits observed after the time horizon). NBS are supposed to show specific cost dynamics compared to classic grey infrastructures risk reduction strategies as shown on Figure 5. For instance, NBS based on vegetation will take long before the efficiency of the measure is significant, and their benefits (services) will span over a very long period of time without need of reinvestment or high maintenance costs as in several classic grey infrastructure. Efficiencies of NBS based on infiltration (groundwater recharge) are time dependent according to the local and regional groundwater dynamics and other physical features. For instance, active groundwater management might be efficient as soon as a heavy rain will occur, but their co-benefits might increase in the future.

In NAIAD, the time horizon of the economic assessment should be decided and discussed carefully. We need to ensure that a sufficient large time span will enable to account for all benefits that will arise in the future. Therefore, we cannot recommend a uniform duration as this will vary according specific NBS and services observed. In addition, economic evaluations are also expected to vary with future climate and land use conditions and these must be included for long time climate adaptation and NBS implementation planning.

In France, for flood risk management project 50 years' time are considered while in the UK 100 years are used or, if the time horizon is less, then a residual value of projects are integrated (CGSP, 2013).

The consequences of benefits appearing in the future is a potential lesser willingness to adopt of stakeholder that do not want to wait for a strategy to be efficient. The political time is short and the efficiency must be quick. There is a communication challenge to explain that to stakeholders and make them accept and understand how delayed benefits are accounted for in economic assessment. The discount factor will account for that.

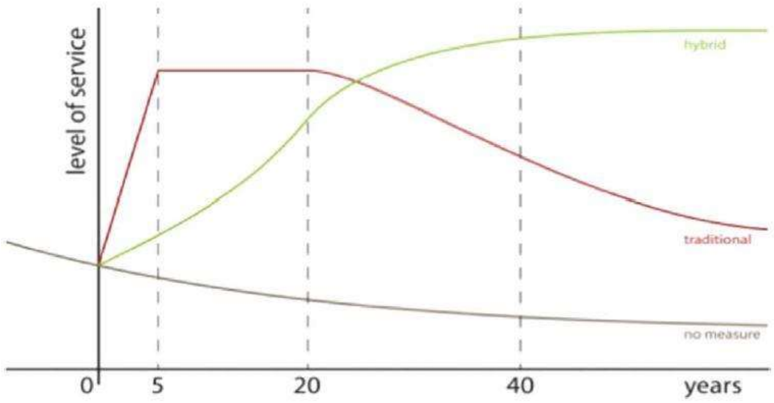

Figure 5 Grey versus Green infrastructure qualitative natural capital dynamics.
(Altamirano et al., 2013)

- Discount factor

Discounting illustrates the preference for the present, as it put less value on future pay offs and costs. The higher the discounting factor the more the preference for the present and lesser care for intergenerational equity issues. The choice of the appropriate discount rate remains one of the most controversial issues in literature. One argument for a high discount factor (e.g. 10%) is that future generations will be better off. However, in environmental CBA the trend is towards lower discount rate to give weight to long-term negative impacts (e.g. climate change, pollution). The Dutch treasury argues for 4% discount rate (Brouwer & Van Ek, 2004). The report Quinet for public infrastructure in France advocates for a rate of 2,5% and 1,5% from 2070 on (CGI, 2017; Quinet, 2013). In NAIAD, we suggest working with the Quinet assumption described before, and if possible providing a sensitivity analysis on it. (see the Annex 1 for the use of the discount factor in the CBA).

7.2. Spatial scale issues

Similarly, benefits or damages could extend over larger scale than the initial perimeter of the case study: for instance, the improvement of flows in a river might have benefits over the initial perimeter and reversely improvements at one place may have negative effects over the initial perimeter. Benefits and costs may differ across different spatial scales and the population affected. The net effects of disasters will vary across the scales of aggregation: individuals, firms, communities, regions and nations. Accurately identifying stakeholders and communities affected directly and/or indirectly by the intervention is necessary to providing the best estimates of socio-economic costs and benefits.

The assessment will need to state clearly the geographic scale of the study, and the population affected to determine what it will account for and what it will not and argue why. The data issue should not prevent from accounting larger areas if costs or benefits are located there. It is better to estimate than neglect completely a source of costs or benefits and affected people or communities. They might be indirect effects or effects that are far from the initial perimeter that will not be worth quantifying. However, for these effects the possibility to refer to it on a qualitative basis and consider it next to the CBA and monetized effects is interesting so that the decision maker understands the limits of the assessment.

7.3. Point of views

The social or collective point of view meaning accounting for all cost and benefits that society will face without accounting for transfers within different groups is the reference point of view adopted for CBA. All costs and benefits will be aggregated to form the net benefit.

The analysis should provide the base for communication towards citizens to adjust their perceptions of the risk and the opportunity of NAS. However, this presentation masks individual situation of losers and winners. The distribution of costs and benefits among different stakeholder will help design and assess business model possibilities (who will invest, contribute, be willing, oppose). The distribution issue also enable to address the equity question.

The CBA can be recalculated for different point of views to help enlighten distribution and equity issues. There is no reason to assume that NAS will be a priori consensually accepted as effective and legitimate solutions against natural hazards. NAS might imply choices that let some worse off in terms of resource allocation and land-use planning. In this context, CBA can be viewed as a pragmatic tool designed to demonstrate the allocative efficiency and to ensure the legitimacy of any finally chosen policy option.

Other point of views can be adopted for specific analysis. For instance, the insurance companies might run a CBA accounting only for their exposure and consequence avoided damages without being interested in co-benefits.

7.4. Risk perception & aversion

Natural hazard and climate related risks are complex and affect differently stakeholders under diverse governance regimes. Even within single political jurisdictions, different groups of interest may have differing visions for the future and perceptions of what constitutes threats to their livelihoods or activities. Those differences can be the origin for diverse policy responses to the same threat but also for conflicting responses to climate change or risk. Therefore, understanding how people perceive a risk and how do they develop aversions towards particular risk management are important factors contributing to successful adaptation to potential future risks.

When it comes to innovation, as NBS might be, capturing the perception of the possible risks that might influence the acceptance or rejection of a NBS measure is key if we want to avoid high transactions costs. Risk perception plays an important role when reacting to hazards (Renn, 1990), and can be influential in determining how people choose to mitigate the risk of those hazards (Martin et al., 2009). However, perceived risk does not always correspond with scientific analysis views (Science Communication Unit, 2014), because risk perception is a combination of social, cultural, political and emotional factors as well as of innate influences (Renn and Rohrmann, 2000). In cases in which people have a limited perception of risk, their reaction might be less appropriate, proportionate, or even harmful (e.g. building houses in flood prone areas). In other cases in which the perception of risk is shaped by historical and social events, the reaction to the hazards and disasters might be more appropriate to the event at hand and could reduce possible harm (e.g. areas with institutions that have a long history of dealing with hazards).

When designing policies or measures including NBS, an important aspect should be the inclusion of an evaluation phase that would capture the positioning of the population and actors affected towards the NBS suggested regarding their capacity to mitigate the risk and towards the "new" risk level. This will serve twofold: on the one hand and as already said, reducing the transaction costs[11], and on the other hand, providing metrics for analysis, in cases in which the aversion is so high that might trigger implementation.

[11] Transaction costs can be found at all levels of policy implementation and are, among other, related to the efficiency of the policy implementation. For a policy or management action to be implemented, its benefits should be greater then its costs. Transaction costs can be understood therefore, as all costs incurred until implementation and as consequence of implementation such as pre-implementation actions (information collection, legislation development, hiring and training staff, lobbying, contracting) or post-implementation actions (reporting, monitoring, and enforcement). Those indirect costs are fundamental for being able to assess the efficiency of the management measure or policy. Transaction costs should be reflected in appropriate metrics that allow for a fast assessment of a possible policy implementation failure (see for more information under http://ec.europa.eu/environment/integration/research/newsalert/pdf/210na3_en.pdf)

In theory, well-being or utility functions that describes how actors and population are well off with a risk reduction strategy can integrate different risk aversion coefficient of these agents. For instance, it will be much more critical for small holders to absorb a shock than for large companies that will be affected less by a variation in their utility. Different utility functions could be used as a combination of aggregated utilities with different risk aversion parameters characterizing different types of groups or actors, as suggested for instance by Pearce et al. (2006). Another way of dealing with the different risk aversion is to define thresholds under which the damage is critical to the continuation of the activity (e.g. farming).

7.5. Uncertainty and sensitivity analysis

'Uncertainty[12]' denotes a cognitive state of incomplete knowledge that results from a lack of information and/or from disagreement about what is known or even knowable (epistemic uncertainty). It can also result from natural variability of physical and social systems (ontological) in which case it is irreducible. Finally, uncertainty can also result from ambiguity, i.e. different, but equally valid framing of views and situations. Sources of uncertainty are important in the analysis of NAS, because we are dealing with (i) natural hazard in the context of climate change and (ii) ecosystems which processes are relatively not well known, and also with large inherent uncertainty (ontological). The time horizon chosen will also play on the uncertainty. We can distinguish 3 types of uncertainty that are described below:

- Risk level uncertainty
- NBS efficiency uncertainty which is related to the effective risk reduction potential of NBS.
- Co-benefits uncertainty linked to the ecosystem services valuation uncertainty

Risk level uncertainty

Risk is a product of hazard, vulnerability (incl. assets) and exposure. The uncertainty is particularly linked to the hazard (rainfalls, evapotranspiration) and the evolution of the assets depending on the evolution of the land use and the values of the assets.

This is an inherent uncertainty, it cannot be reduced with any data acquisition (ontological). The hazard uncertainty is increased by climate change. Climate change increases the variability and intensity of weather events. There is also a great deal of uncertainty in future (model) projections of climate change impacts on the precipitation and runoff. Thus, assumptions need to be included on return periods of hazards, which will be essential for the effectivity and

[12] It is different from risk: risk refers to a situation where the occurrence of future outcomes can be probabilized while uncertainty refers to a situation where no probabilities can be characterized for the different outcomes.

efficiency of NBS, in case these are sensitive to events and climate change (which is usually the case).

Efficiency uncertainty

In NAIAD, models will be used to represent the effect of NAS on risk. This uncertainty will also rely on the models conceptual quality and uncertainty, i.e. in how far models capture relevant physical processes and describe them well, and also dependent on the data quality.

Strategies where the impacts are more certain will be preferred to those where they are highly uncertain, so the estimation of benefit will need to take account of this. Where the uncertainty is always high, some form of robustness analysis of the different intervention options (as employed in the evaluation of flood risk management options) will need to be employed to establish which interventions are robust to the high level of uncertainty. This implies that a wide portfolio of interventions will need to be considered which also take into account the uncertainty about climate and socio-economic drivers that translate into uncertainty about farming futures, as well as the uncertainty in predicting catchment scale impacts on various parameters. Simple decision support tools which present multiple interventions in a risk based framework will be required for policy makers to assess the effectiveness of programmes of measures for given catchments.

Co-benefits uncertainty

The valuation of co-benefits will be particularly challenging when they belong to environmental or social impacts. Using the ecosystem service approach will help adopting a general framework to account for environmental values. However as illustrated by the Spanish experience on groundwater ecosystem valuation (UNEP, 2017) the training, e.g. capacity of the analyst to realize a relevant assessment can be considered as a source of uncertainty. According to Brander et al. (2010) there are three sources of uncertainty in the valuation of ecosystems and biodiversity. First, we may face uncertainty related to the nature of the ecosystem services. Science is starting to shed light about the role of biodiversity in terms of the delivery of supporting services, and robust information is still lacking on how biodiversity contributes to the ecological functions that translate into tangible benefits for society. The flow regulation deals a lot with uncertainty. Second, we may be uncertain or/and ignorant about the way people form their preferences about ecosystem services, i.e., the way they subjectively value changes in the delivery of ecosystem services and biodiversity. Valuation studies often assume that respondents know their preferences with certainty. Empirical evidence in the stated preference literature suggests, however, that respondents are uncertain about their responses (e.g. Akter et al., 2008). Lastly, another layer of uncertainty exists regarding the application of valuation tools. Valuation studies using various techniques can suffer from technical uncertainty due to accuracy problems or biases. Examples being: (i) the potential, hypothetical or strategic, biases that arise from the design of questionnaires in stated preference methods (Bateman et al., 2002), (ii) the effect of assigning probabilistic scenarios in production function based approaches, and (iii) the influence of unstable market prices of substitutes or complements to natural resources in revealed preference methods (e.g., travel cost approach).

There is no consensus about which method is more appropriate for measuring preference uncertainty in stated preference methods. There are three main approaches to deal with this kind of uncertainty in CVM. One is to request respondents to state how certain they are about their answer to the WTP question (e.g., Loomis and Ekstrand, 1998). Another one is to introduce uncertainty directly using multiple bounded WTP questions or a polychotomous choice model (e.g., Alberini et al., 2003). The third option is to request respondents to report a range of values rather than a specific value for the change in the provision of an ecosystem service (e.g., Hanley et al., 2009).

Box : Robust decision making
In highly uncertain system, the robust decision making approach is interesting as it explores all sources of uncertainty and prefer a safe strategy that performs well over all future scenarios rather a strategy that performs best on average. See for instance Lempert et al. (2013).

No regret strategies are typically those that will be a good choice whatever the future reveals to be. NBS seem to be good candidates. Indeed, they will have positive environmental outcomes even if their risk reduction potential is not significantly above grey infrastructure. For instance, restoring a floodplain can easily be considered as a no regret option.

Strategies that are flexible and that can be adapted in the future are also interesting strategies to support in the face of strong uncertainty. Adaptation pathways approach developed by Haasnoot et al. (2013) are particularly interesting in the face of deep uncertainty.

In NAIAD DEMOs, we recommend therefore to define all the factors contributing to uncertainty and for those that have a significant uncertainty, provide a sensitivity analysis. If uncertainty is supposed to be very deep a robust decision making approach can be realized.

7.6. Implication for governance

Environmental governance can be defined as an evolving process of "establishment, reaffirmation or change of institutions [rules, conventions, etc.] to resolve conflicts over environmental resources" (Paavola, 2007). Conflicts refer to conflicts of interests (within a context in which parties share common values) and/or values (a context in which antagonisms between parties refer to values) among the involved parties, and arise from interdependence among agents (Paavola and Adger, 2005). CBA is developed as a tool for conflict resolution, via its "efficiency" and "legitimacy" attributes.

- CBA and environmental governance

The allocative efficiency argument is often questioned by the inability of CBA to be exhaustive: part of the costs and the benefits associated to each option are not measured due to the lack of data, multiple uncertainties, inability to impute a monetary value to some "(dis-)amenities", etc. Things become problematic when incompleteness is due to explicit rejection of monetary assessment of ecosystem services' values by some parties (Vatn, 2009). The legitimacy of the whole process of governance is questioned.

The legitimacy of any environmental governance process largely depends on its capacity to *"take into account"*, in some ways, all the "languages of valuation" (Martinez-Alier, 2008) that are expressed by stakeholders in a given context. These languages may exhibit incommensurability between them and environmental decision necessarily results in a "hard choice" and not simply a trade-off, i.e. in a choice in favour of some values over others. In that case, the "moral force" of cost–benefit arguments is questioned because "those whose interests/values are frustrated by an environmental decision are hardly persuaded about its legitimacy by demonstrating that it was optimal … to endorse and realize the interests/values of others" (Paavola, 2007). This may explain why CBA and monetary valuation are not so much concretely used in environmental decision-making.

OECD (2001) notes that "although fairly common in the environmental economics literature, valuation techniques have remained somewhat peripheral to environmental policy-making on major issues". In fact, there may well be a gap between the ambitions of ecosystem services valuation and its concrete achievements in terms of influencing decision-making. Few studies have seriously tried to understand if and how ESV is or is not used by decision-makers. The work of Laurans et al. (2013) is of special interest since it provides a clear and useful typology of expected uses of ecosystem services economic valuation (UESV):

- **Decisive** UESV (for a specific decision):
    - ESV for trade-offs (strict decision-making structure in relation to a set of alternative projects/options and the goal is to optimize social well-being; **this is NAIADs perspective.**
    - Participative ESV: CBA become a "negotiation language, i.e. it is rather seen as a basis for discussion: "through an open debate on ESV parameters and assumptions, stakeholders negotiate and define a project that is adjusted and enhanced in terms of compromise and the sum of interests."
- **Technical** UESV (for the design of an instrument)
    - ESV for establishing levels of damage compensation (ES degradation and it provides guidance for administrative decisions or court rulings that determine the amounts to be paid out).
    - ESV for price-setting (PES schemes or other schemes).
- **Informative** UESV (for decision-making in general)

- **ESV for awareness-raising:** the vector for a broad message concerning the preferences that should be mainstreamed into society, particularly to ensure that ecosystem services considerations are integrated into public and private choices.
- **ESV for justification and support**: ESV is used by a stakeholder to promote a given course of action, as opposed to ESV for trade-offs where valuations are deemed neutral and inform an optimal choice. It may be *a priori or a posteriori* to demonstrate the economic rationality of the measures envisaged or already made. This type of studies will help policy makers in resolving the conflict for the development or conservation of an ecological zone.
- **ESV for producing accounting indicators**: it involves situations where valuation is designed to allow decision-makers, or the public opinion, to remain informed of the state of the natural capital and to integrate this information into their decisions in general.

The UESV remains an open and challenging issue, not really invested both by the academic literature and by policy-makers. It is important that each NAIAD's demo considers "ex ante" this issue and positions itself in the aim and ambition of the CBA. Most of them will be, more or less, at the crossroads between "decisive" and "informative" uses. From a clear positioning will follow an enlightened choice of methods and course of actions.

- NAS: conflicts of interests and diverging risk perceptions

NAIAD's context is about natural hazards mitigation and the efficiency of NAS in that regard. Clearly, the governance challenge does not concern conflicts of values, in the sense of conflicts between antagonistic and irreducible motivations/values towards an object/a resource. All the stakeholders will agree on the need to mitigate flood/other risk(s). However, we may observe cases of philosophical opposition between concrete "hard/grey" solutions with an engineering "all control" view and a NBS perspective. We can thus consider that "means" may carry meanings - i.e. specific languages defining epistemic communities and specific authorities. Conflicts will concern the choice between alternative means and will thus oppose different interests (material, financial, recreational, conservation, etc.). Parties in conflicts will also exhibit diverging perceptions of risks (cf. WP3 and 6.4). Understood in this way, the fields seem less concerned by strong opposition to monetary valuations and conflicts between incommensurable "languages of valuation". Most of the contexts already exhibit a kind of "positive bias" for many actors towards NAS and the remaining challenge seems to come close to "participative" ESV /ESV "awareness-raising"/"justification and support" ESV, rather than to ESV for trade-offs.

- The potential contribution of CBA to the "common good".

From this discussion, it follows than the core issue surrounding CBA applied to NAIAD's NAS is to ensure its contribution to the construction of the "common good", which must always be conceived as a negotiated compromise. In this context, CBA has a twofold purpose: (i) demonstrating – a priori – the socio-economic relevance of NAS applied to natural risk mitigation; (ii) ensuring its acceptability by having played the role of a “community CBA” (co-construction of avoided damages -insurance value- and co-benefits).

An information constraint relates to the relevance, interpretation and application of existing and emergent practices; norms and principles influences the basis of dialogue and are as such another category of factors that can add to uncertainty, such as the relevance, interpretation and application theories of distributive justice and other global paradigms (Beyene & Wadley, 2004; Lautze et al., 2005).

8. Conclusion

This report presents a framework for the assessment of NAS in NAIAD. Because of the different issues at stakes and specificities of the DEMOs no detailed general method is proposed and the analyses in each DEMO will need to consider methodological choices (e.g. cost and benefits to be included, how to value, time and spatial scale) that will be determinant for the quality and feasibility of the assessment. However, this report delivers a general framework that DEMOs need to follow as far as possible. Economic assessment is a critical part in the context of this report that is difficult to carry out and often yields a share of subjectivity of the analyst. For this reason it is of outmost importance that all the steps of the assessment are reported in detail and the assumption are clearly argued for and reported. The support of DEMOs in this task will be provided by the economists within NAIAD to ease the process and ensure a coherence among the different assessments in the perspective of mainstreaming the results and comparing them (even if the results per se are obviously not transferrable).

The methodological work will be an ongoing process in NAIAD as we will learn and confront ideas to real application in the DEMOs. Methodological challenges in NAIAD with respect to the economic analysis remain:

- Link the risk perception research work (WP3) and the economic assessment of NAS;
- Assess, qualify or quantify the co-benefits provided by Nature Based Solutions (task 4.3);
- Accommodate and trade-off various types of knowledge, such as economic indicators and qualitative information in the MCA for a comprehensive help to decision making;
- Investigate how the economic analysis can help and interact with stakeholders to contribute to the operationalisation of the insurance value of nature-based solutions; in other words, that economic analysis integrates governance.

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Annex 1 – Economic indicator calculation for CBA

Parameters and indexes:

- T, the year index ; n the duration of the project considered in the analysis
- “0” index holds for the reference scenario
- “sc I” index holds for the I scenario (alternative NBS scenario)
- Implementation costs : C
- Opportunity cost : OC
- Damage cost of year t : D(t)
- Avoided damage for year t : AD(t)
- Mean Annual Damage : MAD
- Mean annualized Avoided Damage : MAAD
- Co-benefits : CB
- p the return period or frequency of a hazard scenario x
- r : discount rate that depends on t

Estimation of thee mean annual damage (MAD) and Mean annualized Avoided Damage MAAD (Erdlenbruch et al. 2008) as shown on Figure 8

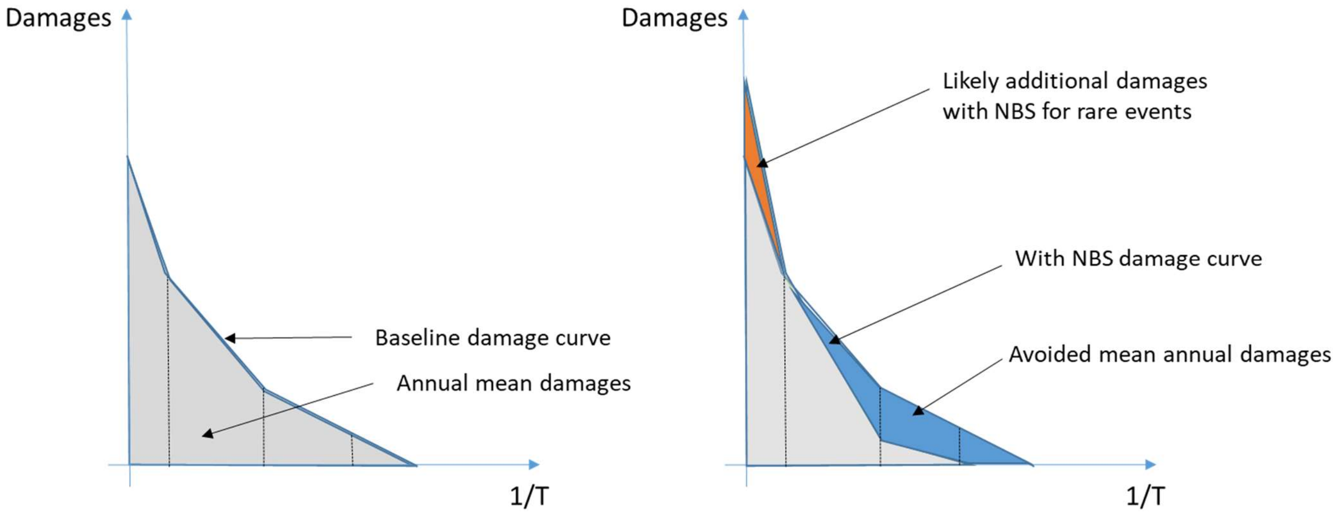

Figure 8 Representation of the calculation of mean annual damages and avoided mean annual damages (redesigned from Erdlenbruch et al. 2008)

This indicator is calculated for both reference (0) and with strategy scenarios (Sc).
It correspond to the weighted sum of the damages, weighting being the flood occurrence probability p(T).

$$MAD = \int_{\boldsymbol{T=Td}}^{\infty} \boldsymbol{D(T) * p(T) dT}$$

With Td the return period of the smallest flood that shows damages. D(T) the damage for the event of return period T.

Or with simplification, when we estimated x hazard scenarios with a return period or frequency of p :

$$MAD = \frac{\boldsymbol{1}}{\boldsymbol{x}} \sum_{\boldsymbol{x}} \boldsymbol{D(x) * p(x)}$$

$$MAAD_{sc\ i} = MAD_0 - MAD_{sc\ i}$$

MAAD is an interesting indicator to handle a mean annual value. However, it can not be used as such in a long term calculation as it is not distributed over time and can thus not be discounted. The use of these indicator is promoted in french system but this does not enable to account for benefits that develop with time (here co-benefits linked to the provision of eecosystem services).

A series of damages have to be simulated to integrate damages and avoided damages in the PVNB equation. Damages ,D(t) for each of the scenarios have to be simulated with modeling or estimated.

- **Avoided damages in year t:**

$$AD_t = D_{t,0} - D_{t,sc\ i}$$

- **The present value of avoided damages**

$$PVAD_i = \sum_{t=0}^{n} \left[\frac{\mathrm{AD}_t}{(1+r)^t} \right]$$

- **The present value of net benefits**

$$PVNB_{sc\,i} = \sum_{t=0}^{n} \left[\frac{C_t + OC_t + CB_t}{(1+r)^t}\right] + PVAD_{sc\,i}$$

or

$$PVNB_{sc\,i} = \sum_{t=0}^{n} \left[\frac{C_t + OC_t + \mathrm{AD}_t + CB_t}{(1+r)^t}\right]$$

- **The Benefit Cost ratio**

$$BCR_i = \frac{\sum_{t=0}^{n} \left[\frac{AD_t + CB_t}{(1+r)^t}\right]}{\sum_{t=0}^{n} \left[\frac{C_t + OC_{t_t}}{(1+r)^t}\right]}$$

- **Internal rate of return is IRR such as the PVNB equals 0,:**

$$PVNB_{sc\,i} = 0$$

Need to solve the following to infer IRR:

$$\sum_{t=0}^{n} \left[\frac{C_t + OC_t + CB_t}{(1+IRR)^t}\right] + MAAD_{sc\,i} = 0$$